Resource efficiency of wood-plastic composites

Identifying secondary material substitution potentials and an environmentally sound end-of-life treatment

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Resource efficiency of wood-plastic composites: Identifying secondary material substitution potentials and an environmentally sound end-of-life treatment

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Abstract

In the drive towards a sustainable bioeconomy, there is growing interest in the development of composite materials made of plastics compounded with wood particles, known as wood-plastic composites (WPC). The main use for WPC is as outdoor decking material. The market share of WPC decking compared to solid wood decking is small, but due to their promoted customer-friendliness in the use stage, WPC decking are likely to increase their share on the outdoor decking market. New products may soon be introduced to new markets as injection moulding and 3D printing of semi-finished WPC compounds or filaments are currently in an early stage.

Recycling of post-consumer materials is promoted by political strategies and concepts, such as the European Bioeconomy Strategy and the Circular Economy to ensure a sustainable resource supply in the context of resource efficiency. The goal of the thesis was to identify resource efficiency potentials of WPC, in terms of the projected increase in WPC consumption and the resulting rise in post-consumer WPC in our society. The focus was on characterizing WPC produced from secondary materials from specific waste streams as well as assessing the environmental parameters of the product and end-of-life (EoL) stages of WPC. An extensive selection of methods was chosen comprising material flow and economic analysis, physical characterization of laboratory scaled WPC specimens, and life cycle assessment (LCA). Application considerations in context of normative standards and policy frameworks completed the holistic approach to be able to identify secondary material substitution potentials and an environmentally sound EoL treatment.

Post-consumer mixed waste wood and recycled particleboard (both A II) could substitute virgin wood particles or wood co-products, but insufficient sorting of abrasive materials prevents their utilization in WPC. From an ecological point of view, wood co-products should be considered instead of recycled wood but, these materials in particular are facing a competing demand in context of energy purposes. From a physical perspective, however, using recycled wood in WPC achieved comparable results to WPC produced from virgin Norway spruce particles.

Post-consumer thermoplastic polyolefins from packaging waste as well as polystyrene and acrylonitrile-butadiene-styrene from electronic waste were identified as substitutes for the virgin thermoplastics currently used in WPC. These secondary materials are readily available, as recyclers looking for new markets of their secondary materials, achieve comparable physical results and would benefit the environmental profile of WPC. Recycling to yield secondary WPC was identified as the environmentally best alternative compared to incineration with energy recovery by means of a system LCA.

Kurzfassung

Im Zuge der Umsetzung der Bioökonomie Strategie der Europäischen Union wird eine steigende Nachfrage auf die Ressource Holz prognostiziert. Auch der Konsum von Produkten auf Basis fossiler Rohstoffe steigt weiter an. Holz-Kunststoffverbundwerkstoffe – *wood plastic composites* (WPC) – erfreuen sich aufgrund proklamierter Wartungsfreiheit in der Nutzungsphase steigender Beliebtheit im Terrassendielenmarkt. Aufgrund einer anhaltend steigenden Nachfrage nach WPC-Dielen und einem noch nicht ausgeschöpften technischen Potential an WPC-Produkten aus dem Spritzguss- oder 3D-Druckverfahren, kann in Zukunft mit einem vermehrten Aufkommen an WPC gerechnet werden. Dies wird sich sowohl auf die Rohstoffbereitstellung als auch auf das Abfallaufkommen auswirken und ökologische Herausforderungen mit sich bringen.

Die Wertschöpfung von sekundären Rohstoffen als auch die Kreislaufwirtschaft sind wesentliche Aspekte der Bioökonomie um eine ressourceneffiziente, nachhaltige Wirtschaft zu realisieren und die Abhängigkeit von Rohstoffimporten zu reduzieren. In dieser Dissertation wurde daher untersucht, welche potentiellen sekundären Rohstoffen in spezifischen Abfallströmen stecken (Verpackung, Elektroaltgeräte), die sich als Substitute in WPC eignen könnten. Dazu wurden verschiedene Methoden herangezogen, um zunächst die mengenrelevanten Abfallfraktionen und Sekundärrohstoffen mit Hilfe einer Stoffstromanalyse zu identifizieren, die weiter zu WPC verarbeitet werden könnten. Ein Augenmerk wurde dabei auf ökonomische Wechselbeziehungen gelegt. Im Technikum wurden anschließend WPC-Prüfkörper hergestellt, die physikalisch charakterisiert wurden. Ökologische Fragestellungen wurden anhand von Produkt- und Systemökobilanzen beleuchtet. Umsetzungsbezogene Fragestellungen hinsichtlich rechtlicher Rahmenbedingungen und Produktnormen wurden hinzugezogen.

Gemischtes Gebrauchtholz der Altholzkategorie A II sowie gebrauchte Spanplatten (A II) eignen sich als Substitute für die derzeit verwendeten frischen Sägenebenprodukte oder aufbereiteten Industriehölzer von Nadelhölzern. Die Ergebnisse der Produktökobilanz und der Elementaranalyse zeigten, dass jedoch Sägenebenprodukte verwendet werden sollten. Diese stehen allerdings im immer größer werdenden Wettbewerb zur energetischen Nutzung. Wie Ergebnisse der physikalischen Materialcharakterisierung zeigten, ist die Altholzverwertung für WPC möglich. Sekundäre thermoplastische Polyolefine von recycelten Verpackungsabfällen sowie Acrylnitril-Butadien-Styrol und Polystyrol von recycelten Elektro- und Elektronikabfällen eignen sich als Substitute. Diese sind in ausreichend qualitativer, guter Verfügbarkeit erhältlich und ökologisch vorteilhaft. Die Systemökobilanz zeigte, dass stoffliches Recycling von WPC die ökologisch sinnvollere Alternative gegenüber der Verbrennung mit Energierückführung ist.

Content

List of abbreviations

ABS	Acroloynitrile-butadien-styrene
ALCA	Attributional life cycle assessment
As	Arsenic
AP	Acidification potential
ADPE	Abiotic depletion potential for non-fossil resources
ADPF	Abiotic depletion potential for fossil resources
bbl	Barrel (159 litres)
Co	Cobalt
Cd	Cadmium
Cr	Chromium
Cu	Copper
CCA	Chromated-copper-arsenate
CLCA	Consequential life cycle assessment
CML	Institute of Environmental Sciences
CML-IA	Database containing characterisation factors for LCIA
$energy_{el}$	Electricity
$energy_{th}$	Thermal energy
EEG	German Renewable Energy Act
EoL	End-of-life
EU	European Union
EP	Eutrophication potential
EPD	Environmental product declaration
EPS	Expanded polystyrene
FU	Functional unit
GER	Germany
GHG	Greenhouse gas
GWP	Global warming potential
HDPE	High density polyethylene
KrWG	German Waste Management and Product Recycling Act
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
mc	Moisture content
MAPE	Maleic anhydride polyethylene
MAPP	Propylene-maleic anhydride graft copolymer
MFI	Melt flow index
MDF	Medium-density fibreboard
MoE	Modulus of elasticity
NCV	Net calorific value
Ni	Nickel
ODP	Ozone depletion potential
Pb	Lead
PE	Polyethylene
PHA	Polyhydroxyalkanoate
PLA	Polylactide
POCP	Formation potential of tropospheric ozone photochemical oxidants

PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
UF	Urea formaldehyde
USD	US-dollar
RoHS	Restriction of Hazardous Substances Directive
SEM	Scanning electron microscopy
SMA	Styrene maleic anhydride copolymer
TPS	Thermoplastic starch
TRACI	Tool for reduction and assessment of chemical and other environmental impacts
UV	Ultraviolet
WEEE	Waste electrical and electronic equipment
WPC	Wood-plastic composite

1 Introduction

The introduction of fossil hydrocarbon resources into modern society was an irreversible turn in technological development. It enabled economic growth, mass production of goods, and individual mobility; but also lead to growing waste streams and severe environmental issues on local, regional, and global scales (IPCC 2014). The global demand for engineering materials has quadrupled in the past 50 years. Engineering materials originate from crude oil (i.e., plastics), ores (i.e., metals, ceramics) and biomass (i.e., timber) (Allwood et al. 2011). This resulted in shortage of economically feasible exploitable hydrocarbon fossil resources due to constant increasing exploitation expenses. However, unforeseen technological leaps and political circumstances are likely to change the market price of crude oil dramatically (Neumayer 2013). Crude oil prices have been dropping since 2014 and continue to drop in 2016. On Jan 21, 2016, the crude oil price was 28 USD/bbl. In June 2008, the peak of oil price was 137 USD/bbl (Inflationdata 2016). It is questionable how the prices of crude oil will develop over long run. Yet, it is given that

- industrialized and emerging economies depend almost completely on crude oil.
- The wealth of our societies depends on a non-renewable resource, which
    - faces great uncertainty in terms of availability and price development due to political and technical circumstances,
    - causes hazardous impacts to the human health and the environment during its life cycle,
    - results in increasing synthetic, non-naturally degradable waste.

Up to now, about 50% percent of the total global crude oil consumption is utilized as an energy carrier for transport purposes, 32% is used for heating, and 8% for electricity. The remaining 10% is used for products (5% chemicals, 5% polymers) (Franke et al. 2014). Therefore, a tremendous amount of this valuable resource is lost in the transformation to energy.

Based upon the sustainable development goals (UNCED 1992), the European Commission released a communication report in 2010 aiming at a sustainable and inclusive growth of the European Union – EU (EC 2010). This communication report promoted the use of renewable resources as bioenergy, which is highlighted as the climate change and energy target for the year 2020 and beyond. Further on, The European Bioeconomy Strategy (EC 2012b) highlights the use of renewable resources instead of fossil resources in products as the cornerstone of sustainable economic growth.

1.1 The bioeconomy

The bioeconomy encompasses a strategic vision of the European economy to become independent from hydrocarbon fossil resources and foster the economy with domestic renewable resources. Foremost, this is based on the afore mentioned uncertainties in prices, exacerbated by the EU's massive and increasing reliance on imports to meet its resource needs and its resulting political and industrial dependence on export countries (EEA 2015). This has led to a renaissance in bio-resources for the production of energy (heat, electricity and fuel) in the EU over the last decade.

Increased use of biofuels for land and air transportation will help reduce fossil carbon consumption and mitigate its contribution to anthropogenic GHG emissions (IPCC 2014; Laqua 2015). The so-called "1st generation biofuels" from biomass are derived from starch or sugar crops, such as corn, maize, or vegetable oil, which can be categorized as energy crops, resulting in biodiesel, biogas, and bioethanol. Because biofuel production is highly subsidized and has short rotation cycles (< 1 year), farmers increasingly began switching their agricultural land use to biofuel production instead of food or fibre (Baumann 2015). The transformation of land-use occurs on de-harvested areas, but also on forest areas. This has raised concerns about land use competition, direct and indirect land use change effects, and loss in biodiversity (Alexandratos & Bruinsma 2012; Lauri et al. 2014; Searchinger 2013; Searchinger et al. 2015).

Unused residues and co-products from agricultural land, forest activities and organic waste are promising alternatives to the 1st generation biofuels in terms of cultivable land in the EU Member States. They are classified as the "2nd generation biofuels" (Searle & Malins 2016). Nonetheless, biofuels from woody biomass also raise environmental concerns (Valin et al. 2015). The net carbon fluxes – carbon sequestration and carbon losses – may influence the total carbon balance of a forest and its products (Klein et al. 2015). A non-sustainably managed forest or a forest transformed to agricultural land with less carbon storage in the long term, will lead to more GHG emissions, expressed as biogenic CO_2 (Brandão & Milà i Canals 2013; Searchinger et al. 2009).

Currently, market leaders in the European forest and wood-based industries are seeking new business opportunities for products based on wood, such as wood-plastic composites (WPC), lignin-based products, biochemicals and afore mentioned biofuels (Laqua 2015). This highlights the multiple possibilities of using wood and opening new markets in the near future. The use of wood in various constructive and non-constructive applications, i.e., as a renewable energy carrier and as a renewable precursor for the chemical industry and biofuel industry led to a rising demand for wood resources (Härtl & Knoke 2014). In the long term, this competition is likely to result in a potential shortage of wood resources from sustainably managed forests (Lauri et al. 2014; Mantau et al. 2010; UNECE et al. 2011).

As a result, the EU is facing a competing demand for wood resources as it strives toward the bioeconomy and to maintain sustainable economic growth. The aim of the bioeconomy is to strengthen the production of renewable biological products and resources and the conversion of these resources and waste streams into value-added products by:

- ensuring biodiversity and environmental protection,
- reconciling demands for sustainable agriculture and fisheries, food security, and the sustainable use of renewable biological resources, for industrial purposes and
- striving to be an innovative, resource efficient and low-emission economy (Allen 2015).

1.2 Circular economy

Recycling is one of the cornerstones of the bioeconomy. The term "circular economy" has become popular by a report published by the Ellen MacArthur Foundation and its founding partners from the automotive, IT, energy, and consulting industries (Ellen MacArthur Foundation 2013). The concept is based on different scholarly ideas, such as industrial ecology, biomimicry and cradle-to-cradle (McDonough & Braungart 2002; Singh & Ordoñez 2015; Yuan et al. 2006). The concept was firstly proposed and introduced by Chinese authorities in 2002 as a new development strategy aiming to reduce the contradictions between rapid economic growth and the shortage of raw materials and energy. The circular economy links the afore mentioned scholar ideas with policy implementations and economical aspects with ecological recycling activities embedded in a transnational strategy. For example, the European strategy of circular economy refers to several previous European policy actions, such as the "Roadmap to a Resource Efficient Europe" (EC 2011b) and "Innovating for Sustainable Growth: A Bioeconomy for Europe" (EC 2012b; EP 2015).

Besides circular economy, the term "cascading" has been on the rise, which is also manifested in the circular economy and bioeconomy strategy, but more focused on bio-based materials. Cascading of resources can be linked to substitution and using secondary resources in terms of resource efficiency. Sirkin & Houten (1994), as one of the first authors to research cascading of resources, discussed resource quality loss and quality gain through recycling efforts, which consumes energy, labour, and additional resources. The authors stated, "*resource cascading has been utilized as a method for achieving resource conservation in contexts where resources have been regarded as precious or vital.*"

In Europe, the concepts of circular economy and cascading of resources are implemented to some extent in the Waste Framework Directive 2008/98/EC (EC 2008). Historically, the waste hierarchy was formalized as an ordered system of preferred management options in 1989. In 2008, the European Parliament adopted the waste hierarchy in the Waste Framework Directive 2008/98/EC, which Member States must introduce into national laws (Williams 2015), such as the German Waste Management and Product Recycling Act – KrWG (German Government 2012). These policies priories a cascaded treatment of waste by the so-called "waste hierarchy": (1) prevention, (2) preparing for re-use, (3) recycling, (4) other recovery (i.e., incineration with energy recovery) and (5) disposal (i.e., incineration without energy recovery, landfilling).

All mentioned concepts point at the "polluter pays principle", adopted by OECD in 1972 (OECD 1992), that is manifested in the extend producer responsibility (EPR) (OECD 2001). The EPR emerged in Germany and Sweden in the early 1990s. It was set up as a policy strategy to be able to create incentives for eco-design of packages and products, internalizing the costs of waste management from municipalities and taxpayers to firms and consumers (Lifset et al. 2013). The OECD published a policy guidance for governments in 2001, how to implement EPR in national policy making (OECD 2001) and is now integrated in the Waste Framework Directive 2008/98/EC Article 8 (EC 2008). The EPR is a an established policy instrument for resource efficiency and addresses the resource supply chain as well as the downstream markets of a company's products (OECD 2016).

1.3 Resource efficiency

The "Roadmap to a Resource Efficient Europe" (EC 2011b) stated the ambitious goal, that by 2050 the EU's economy will be driven by sustainably managed resources. The way to achieve this vision is called "resource efficiency". A brief look at a global perspective, international initiatives have been established to support polices with scientific assessments in context of a sustainable resource use, such as the Resource Panel of the United Nations Environment Program (UNEP 2016). US policies have focused on energy efficiency, with Japanese and German policies and strategies focusing more on material cycles, such as the ProgRess I in Germany (BMUB 2015b; Huysman et al. 2015). In 2016, the German Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety published ProgRess II, which now satisfies the energy efficiency perspective as well (BMUB 2015a).

The EC will stimulate the secondary materials market and demand for recycled materials through economic incentives. Further, The European Commission stated (EC 2015), "*Optimising the use of resources is essential to ensure that growth is green and inclusive. Starting next year* [2016 – author's note]*, we will implement an Action Plan on the Circular Economy to create a single market for the re-use of materials and resources, supporting the move away from a linear* *economy. This will require action in all parts of the economic cycle, from sourcing to production, consumption, waste and recycling and innovation to harness economically and environmentally efficient business opportunities.*" Recycling is therefore a profound aspect of resource efficiency. It reduces the demand for virgin resources and thereby mitigates related energy use and environmental impacts (Klinglmair et al. 2014). Changing the material structure of a product by substitution of secondary materials through recycling activities is thus a potential way towards resource efficiency on a product level (Woidasky & Hirth 2012).

How can resource efficiency be measured or assessed? A multitude of resource efficiency indicators are represented in the "Resource Efficiency Scoreboard" of the EU (eurostat 2016) respectively in the scoreboard database on eurostat. For example, the indicator "resource productivity" reflects the ratio of economic output (GDP) to domestic material consumption (DMC) (Equation 1). Domestic material consumption estimates the amount of raw materials (measured by mass) directly used by an economy, including both materials extracted from domestic territory and net inflows of goods and resources from abroad (EEA 2015).

$$Resource\ productivity = \frac{\text{Gross Domestic Product (GDP)[€]}}{\text{Domestic material consumption (DMC)[kg]}}$$

Equation 1. Resource productivity according to Resource Efficiency Scoreboard used on eurostat

Although the resource productivity has increased in the EU-28 by 29% from 1.34 €/kg in 2000 to 1.73 €/kg in 2012, the European consumption patterns remain resource intensive (EEA 2015). Therefore, it can be argued that resource productivity is more focused on macro-economics than integrating ecological concerns on a product level. Huysman et al. (2015) stated, that indicators measuring resource efficiency can be categorized in (i) resource use-oriented indicators and (ii) environmental impact-oriented indicators which both can be assessed by the life cycle assessment (LCA) methodology.

1.4 Life cycle assessment

The ISO TR 14062: Environmental management (ISO 2002) provides a general outline of integrating environmental aspects into product design, where LCA should be used to gain ecological information in the conceptual design stage of a product. Two major LCA approaches need to be distinguished to assess the ecological impacts of products or services: "attributional" and "consequential". These two approaches differ significantly in their goal, scope, system boundary and data acquisition (EC et al. 2010; Ekvall & Weidema 2004; Guinée et al. 2011; Rajagopal 2016; Zamagni et al. 2012). If the scope is to describe the potential environmental impacts of a product, the suitable approach would be probably the attributional LCA (ALCA), as described by Heijungs (1997). If the aim is to assess the environmental dependencies of changes by using resource X or Y in a product, the suitable approach would be the consequential LCA (CLCA) approach. CLCA aims at modelling the consequences of one additional unit of output (marginal unit) rather than the average consequences of a product (Curran et al. 2002; Finkbeiner et al. 2014). The result provides information about how an individual decision will influence the environment and whether the purchase of a supposed environmentally friendly product is likely to lead to a reduction in overall environmental impacts or not (Frischknecht 2006).

At first sight, a thoroughly conducted environmental analysis should follow the CLCA approach to be able to deal with questions of resource efficiency and derived resource substitution potentials. However, conducting a CLCA leads to severe shortcomings that will be further explained. Beforehand, existing CLCA studies and used models rarely provide high levels of accuracy, completeness or precision. Case studies just reflect a few effects, as CLCAs cannot describe the full consequences of a change (Ekvall 2002; Finkbeiner et al. 2014).

In detail, the provision of quantitative environmental information is the unique feature of LCA. Data acquisition for the life cycle inventory (LCI) is therefore extremely important for a quantitative data driven assessment. While the ALCA is a rather mature approach (Rajagopal 2014), data acquisition in CLCA is a rather complex issue that needs knowledge in the identification of the marginal change and economic modelling, such as, i.e., equilibrium modelling (Ekvall 2002; Ekvall et al. 2016; Ekvall & Weidema 2004; Plevin et al. 2014b; Weidema 2003). EC et al. (2010) published The International Reference Life Cycle Data System – known as the ILCD handbook – for further guidance of good-practice LCA beyond the ISO standards 14040ff. The ILCD handbook gives guidance on applying both ALCA and CLCA approaches. However, Ekvall et al. (2016) published a discussion paper about the revision of the ILCD handbook due to inconstancies among the application of those approaches in LCA studies, which have been interpreted not consistently (before and) after the publication of the ILCD handbook. See for example the discussion between Brandão et al. (2014), Dale & Kim (2014), Anex & Lifset (2014) and Suh & Yang (2014) in response on the work of Plevin et al. (2014b) and the reply again by Plevin et al. (2014a).

Further, Rajagopal (2016) stated that ALCA is often used to understand the potential loads and benefits of replacing one activity or service with a substitute. However, the consequences of substitution generate spill over effects on the production and consumption which suggests the use of CLCA. These effects can in turn also be seen as unintended, market-mediated or indirect effects. Therefore, the bigger questions lie in issues surrounding forecasting and assessing unintended consequences. Mortimer (2016) stated that "*unintended consequences are distinct from risks and trade-offs in that they are unforeseen at the time of planning and implementation.*"

LCA is well suited to assess the environmental performance of a product today or ex ante, but it is less suitable for prospective questions (World Economic Forum et al. 2016). The latter inherently implies consequences. The future is naturally uncertain and limits exist to comprehensibly describe future consequences of a change (Ekvall 2002), such as rebound effects, technological shifts, price changes and political circumstances. This becomes noticeable at the core of the LCA methodology as good-LCA practice only concerns “potential” environmental impacts and not the “actual” environmental damage or health risks. As a result, carrying out an LCA is a steady state analysis (Wrisberg et al. 2002) which is best shown with the ALCA approach rather than the dynamic CLCA approach. Last but not least, conducting an LCA is trying to abstract the reality in a model. Therefore, it is not possible to account for all information but rather to identify the system’s main objectives, functions, and parameters and their relationship to each other (Ausberg et al. 2015).

Finkbeiner et al. (2014) proposed assessing the changes by means of a baseline using ALCA and different scenarios instead of using CLCA until more robust and consistent methods and case study experiences are available. This is further supported by studies that suggest viewing both approaches as complementary rather than substitutable (Anex and Lifset 2014; Dale & Kim 2014; Rajagopal 2014, 2016; Suh & Yang 2014). The LCA community refers to the ILCD handbook as a guide for good LCA practice and as a starting point for further methodological progression concerning CLCA. Therefore, a revision of the ILCD handbook can be expected in the future (Ahlgren et al. 2015; Ekvall et al. 2016; Laurent et al. 2014).

2 Wood-plastic composites: status quo

WPC are wood particles incorporated in thermoplastic matrices and produced in a one-step or two-step production process. The EN 15534-1 (CEN 2014a) defines WPC "*as a material or product made thereof being the result of the combination of one or several cellulose-based material(s) with one or several thermoplastics, intended to be or being processed through plastic processing techniques.*" WPC are mainly applied in outdoor applications. Water uptake and biological durability of the composites are considered superior to some wood species due to the use of hydrophobic thermoplastics (Clemons 2002; Glasser et al. 1999; Müller 2011). The water uptake can be further improved through modification of the wood particles (Krause & Grahl 2014; Müller et al. 2012). The density of WPC is higher than of solid soft wood decking, i.e., Scots pine (*Pinus sylvestris*) 0.47 g/cm^3 (Dunky & Niemz 2002), and of neat plastic, i.e., polyethylene (PE) 0.94–0.97 g/cm^3 (Klyosov 2007). The stiffness and heat stability of thermoplastic matrices are improved by adding wood particles (Valle et al. 2007; Wolcott & Englund 1999). Although wood-derived fillers are not as popular as inorganic fillers, they have advantages to the latter such as low density (which reduces weight) no–low damage potential to the processing equipment and low cost (Kaseem et al. 2015).

The amount of European produced WPC is small compared to China and USA, but steadily increasing according to market data from nova-Institute (Carus et al. 2015a; Carus et al. 2015b; Partanen & Carus 2016). Table 1 and Figure 1 present current market data on WPC production techniques and application fields in the EU. Germany comprises almost half of the share of European produced WPC (Eder 2013). Up to now, the main applications are extruded (i.e., outdoor decking) or compression moulded (i.e., car-door panels). WPC decking comprise a market share of 6% (Türk 2014). These products are becoming more and more attractive to customers due to their advantages in the use stage, such as low maintenance as well as better swelling and shrinking behaviour than solid wood decking. The price of WPC decking ranges in the middle price range for solid wood decking from domestic sources and of tropical or thermo-modified wood.

Table 1. WPC production in Europe.
Data based on Eder (2013), Carus et al. (2015a)

WPC production technique	Share	10^3 t
Extrusion	81%	210
Compression moulding	13%	34
Injection moulding	6%	16
Others	< 1%	1
Total	**100%**	**261**

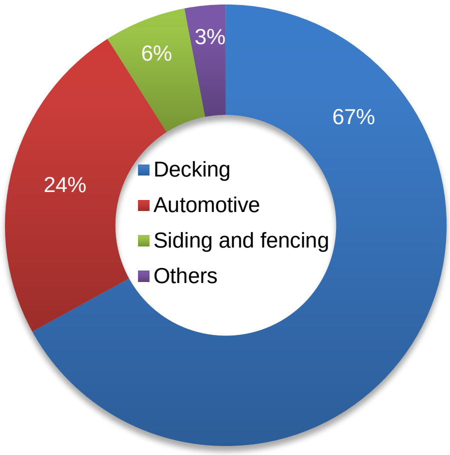


Figure 1. WPC applications (Eder 2013)

Despite of the small size of the WPC market compared to other composites, the potential applications of WPC have yet to be tapped up with regard of wood particle modification and additives for improvements of composite properties. Injection moulding and 3D printing of WPC compounds and filaments are in technically early states (Grujovic et al. 2016), but expand the application possibilities of wood materials compared to conventional wood-based products like particleboards or fibreboards.

Nevertheless, from an environmental point of view, WPC faces challenges embedded in the afore mentioned sections of the bioeconomy, circular economy and resource efficiency. They can be addressed to the upstream (raw material supply) and downstream (product applications, end-of-life) processes.

2.1 Raw material supply for existing WPC products

2.1.1 Wooden lignocellulosic particles

In Germany, the yearly roundwood consumption is about 75 x 10^6 m^3. The main distribution channels for roundwood as a material are saw logs (45%), engineered wood products (11%) and pulpwood (8%). About 36% of roundwood is directly burned as an energy carrier without having a material utilization first. If the amount of waste wood and wood residues is added, about 50% of the national timber stock is used as an energy carrier (Bioökonomierat 2016; Seintsch & Weimar 2013). According to Mantau (2012a), more wood was used for energy than for material purposes in Germany in 2010. This can be mostly related to the doubled fuel wood demand from private households between 1994 and 2005 which increased competition for wood resources, especially for lower quality timber grades (Härtl & Knoke 2014; Wimmer et al. 2013). In addition, political incentives such as the German Renewable Energy Act – EEG (German Government 2014) support the use of fuelwood.

Thermal energy ($energy_{th}$) from fuelwood of virgin sources is likely to remain an important alternative for private households for heating purposes. In terms of resource efficiency and the shift from softwood to hardwood in context of changing forest management, softwood particles should today be efficiently used by resource cascading (Bioökonomierat 2016; BMEL 2016; Polley et al. 2015). According to Wimmer et al. (2013), alternative natural fibres such as hemp, flax or sisal may be seen as substitutes to wood particles in WPC but these natural fibres are much higher in price. The intensified utilization of waste wood and wood residues is a potential solution to a value-added, competitive wood cascade (Bioökonomierat 2016).

About 80% of the produced WPC decking in Germany are in accordance with the "Quality and test specifications for production control of decking" of the Qualitätsgemeinschaft Holzwerkstoffe e.V (Qualitätsgemeinschaft Holzwerkstoffe e.V. 2016). The wood content must be minimum 50% of dry mass and derived from certified, sustainably managed forests and/or waste wood of classification A I according to the German Waste Wood Directive – AltholzV (German Government 2003), in order to obtain the quality label "Qualitätszeichen Holzwerkstoffe". Therefore, waste wood of minor quality in context of higher concentration of impurities such as A II shall not be utilized for WPC. Other WPC products without this label may consist of 20–80% wooden lignocellulosic particles (Carus et al. 2015b).

2.1.2 Thermoplastic polymers

Plastics have become one of the essential materials of the modern economy (Allwood et al. 2011; World Economic Forum et al. 2016). They are responsible for the economic benefits of the sectors packaging, transportation, healthcare, and electronics in context of their low cost, versatility and durability (Andrady & Neal 2009). Despite their great success, the extraction of the hydrocarbon feedstocks faces political insecurities (Rajendran et al. 2012). The conversion to plastic products is highly energy intensive and the products thereof may cause severe impacts to the ecosystems and human health if not properly handled after use (Essel et al. 2015; Rochman et al. 2015). In addition, German plastic converters have concerns regarding the supply of plastics in the near future due to the weak euro exchange rate to the USD and Chinese renminbi (INVERTO 2015).

The most commonly used plastics in WPC are the commodity polyolefins plastics [polypropylene (PP), polyethylene (PE)], and polyvinyl chloride (PVC). They are low in price compared to other plastics such as acrylonitrile-butadiene-styrene (ABS) and have a low processing temperature < 200 °C, for what they are suitable to be mixed with wood in context of the thermal degradation of wood (Borrega & Kärenlampi 2008; Klyosov 2007). Also bio-based polymers such as Bio-PE and polylactide (PLA) can be used (Partanen & Carus 2016). However, the currently high price of bio-plastics prevents further use (Kim 2014). According to the afore mentioned quality specifications of German produced WPC decking, the plastics have to be derived from 100% virgin sources or homogenous post-industrial plastics from primary plastics manufacturing (Qualitätsgemeinschaft Holzwerkstoffe e.V. 2016).

2.1.3 Utilization of secondary materials for WPC

Historically, WPC was invented based on the recycling approach to produce value added products from waste (Klyosov 2007) what is in contrast to the quality label of German WPC decking. The technical feasibility of substituting secondary materials for WPC has been intensively studied in the research community (Balasuriya et al. 2003; Chen et al. 2006; Gozdecki et al. 2014; Krause et al. 2013; Migneault et al. 2014; Zimmermann & Zattera 2013). All these studies focused only on a technical perspective by characterizing the laboratory scaled composites by means of physical testing and the like. Often, secondary materials were declared "waste", and instead post-industrial products or simply co-products, such as trimmings were used (Boeglin et al. 1997; Nourbakhsh et al. 2010). Degradation of polymers due to ultraviolet (UV) wavelengths is of interest for the quality of new products made from secondary materials. Kazemi-Najafi et al. (2013) concluded that degraded HDPE affected negatively the processing of composites, but mechanical properties were similar to the WPC with a virgin HDPE matrix. An overview of the impact of recycling and accelerated weathering on the processing properties of secondary polyolefins for WPC is provided in Table 2.

Table 2. Impact of recycling and accelerated weathering on processing properties of secondary polyolefins for WPC (Kazemi-Najafi 2013)

Processing properties	Description	Authors/further reading
Melting point	Impurities and additives may raise or reduce the melting point	Achilias et al. (2008), Kazemi-Najafi et al. (2009)
Immiscibility	Additional costs due to separation of recycled plastic compartments, Different degradation of polymers during processing and service life can affect the compatibility	Mantia et al. (1992), Goodship (2007), Waldman & Paoli (1998),
Rheology	Increase or reduction of melt flow index (MFI) of recycled PP and PE (decrease in melt viscosity) due impurities	Klyosov (2007), Kazemi-Najafi (2013)
Crosslinking	Crosslinking is affected by weather exposure and degradation and may have negative impact on processability	Kazemi-Najafi et al. (2013)
Crystallinity	Crystallinity of recycled PP and PE is usually less than that of virgin (which can be related to crosslinking)	Valadez-Gonzalez et al. (1999), Gulmine et al. (2003), Kazemi-Najafi et al. (2013)
Polarity	Formation of polar groups in recycled plastics (especially in PE and PP) may improve compatibility between plastic and wood Sometimes the formation of polar groups is accompanied with crosslinking	Valadez-Gonzalez et al. (1999), Gulmine et al. (2003), Kazemi-Najafi et al. (2013)

2.2 End-of-life of WPC

The EN 15534-4 (CEN 2014b) states that "*WPC materials are recyclable materials which can be treated in a material recovery process intended to save resources while minimising harmful emissions into air, water and soil as well as their impacts on human health.*" From a technical point of view, the challenge of WPC recycling is that the composites are complex regarding the material matrix (Kazemi-Najafi 2013; Schirp & Hellmann 2013). Feedstock recycling in terms of resource cascading of the neat resources is technically challenging. During the compounding process of WPC, the thermoplastic matrix is heated above the crystalline melting point (T_m). Then, wood particles are added to the melted thermoplastic and mechanically, irreversibly bonded to the plastic-matrix. The WPC matrix is cooled until the thermoplastic molecules solidify, which is known as the glass transition temperature (T_g) (Klyosov 2007). Additionally, WPC seeks a high grade of plastics purity in context of molecular immiscibility (Brogaard et al. 2014; Christensen & Fruergaard 2010; Schalles 2004; Soccalingame et al. 2013; Winandy et al. 2004). Degradation of WPCs due to repeated processing cycles and environmental exposure also may complicate recycling (Winandy et al. 2004) as described previously in Table 2 for secondary polyolefins.

Schirp & Hellmann (2013) demonstrated that the flexural properties and water absorption of WPC were positively affected by using 20% secondary WPC and 66% "fresh" WPC. A high amount (14%) of stabilizers had to be added. Other studies based on long-term study design reported the technically feasibility of recycling WPC to secondary WPC based on physical, mechanical and biological results (iVTH 2015a, 2015b, 2015c).

According to the quality label of WPC decking, secondary re-grinded WPC profiles are only allowed to be used from the manufacturer's own WPC system, which are withdrawn from the market (Qualitätsgemeinschaft Holzwerkstoffe e.V. 2016). This implies that WPC profiles need to be collected separately from other waste collection systems. If post-consumer WPC are collected through conventional bulk waste collection and therefore mixed with other products, laboratory-scaled results showed, that near infrared (NIR) spectroscopy is suitable to sort different WPC based on the plastics' molecular structure into homogenous fractions (PE-WPC, PP-WPC etc.) (Meinlschmidt et al. 2014). It is possible to determine the plastics content in WPC with dynamic scanning calorimetry, which was tested on PP-WPC (Jeske et al. 2011). Li et al. (2015) showed that a combination of fourier transform infrared spectroscopy and partial least squares regression is promising in the context of determining the share of wood and plastics in post-consumer WPC. Such automated sorting techniques are currently not economically feasible for post-consumer wood (Meinlschmidt et al. 2013), and only applied in plastics recycling, but an economic evaluation for post-consumer WPC is lacking. It is also questionable if incineration (with energy recovery) of WPC would be environmentally preferable to recycling.

2.3 Environmental assessments of WPC

In terms of ecological aspects of WPC a literature review was performed to evaluate the status quo of the WPC which were assessed, and how the LCA study was conducted. *Science Direct*, *Springer Link*, *Wiley Online Library* databases were used as well as *Google Scholar* and the conventional Google search engine*.* The literature search was conducted according to the following criteria:

- Published studies starting from the year 2006 until 2015
- Peer-reviewed articles and LCA studies in English and German languages
- The analysis is in accordance with ISO 14040 and 14044 as minimum standard requirements
- Search key words: "life cycle assessment", "ecological footprint", "environmental assessment" and "wood–plastic composites" including their abbreviations
- Where lignocellulosic softwood and/or hardwood particles were used, excluding biomass from short rotation coppice and plants

The matched studies were descriptively analysed by using the following evaluation criteria that were grouped in two categories:

I) Product(s) under study

1) Product category: Which parameters were used as the functional unit?
2) WPC formulations: Which materials were used to produce the WPC under study?
3) What was/were the foreground WPC processing technique/s
4) What was the $energy_{el}$ demand for WPC processing?

II) Applied LCA methodological approaches

5) Functional unit: What parameters were used as the functional unit?
6) Life Cycle Inventory (LCI): Which kind of LCI approach was applied (i.e., ALCA, CLCA)?
7) Allocation: Which allocation method/s was/were applied?
8) Post-consumer scenario: How was the end-of-life (EoL) assessed?
9) Life Cycle Impact Assessment (LCIA): Which LCIA method was applied?
10) How was resource efficiency addressed?

The first search results indicated 39 research articles dealing with LCA of WPC. Among those articles, studies focusing on natural fibres were not selected, such as flax fibre Yan et al. (2014), hemp fibre (Schmehl et al. 2008), China reed fibre (Corbière-Nicollier et al. 2001), kenaf fibre (Wang et al. 2013), jute fibre (Alves et al. 2010) and rice husks (Vidal et al. 2009). As a result, seven studies matched the requirements for the literature search: Thamae & Baillie (2008), Xu et al. (2008), Bolin & Smith (2011), Stübs et al. (2012), Bergman et al. (2013), Mahalle et al. (2014), and Qiang et al. (2014). These studies are listed in Table 3. In 2015, two environmental product declarations (EPD) were published for WPC decking and claddings (IBU 2015a, 2015b). These studies are not shown in Table 3 because the EPD of WPC decking can be linked to a great extent to the study of Stübs et al. (2012).

Table 3. Results of literature review of LCA of WPC: Products under study; *waste wood or secondary wood

Product category	WPC alternatives	Wood source: Virgin wood	Wood source: Co-products	Wood source: WW*	Plastics source: Virgin	(%)	Plastics source: Secondary	(%)	Additives	(%)	WPC processing	Authors
Outdoor decking	1	–	50%	–	HDPE	25	HDPE	25	–		?	Bolin & Smith (2011)
Outdoor decking	2	–	50%	–	PE	40	–		Coupling agent (PE) Filler (talc) Lubricant (Polsyster) Biocide (borax) Thermostabilizer (TiO_2)	2 2 2 2 2	Compounding – extrusion	Bergman et al. (2013)
					–		PE	40				
Outdoor decking	2	70%	–	–	PE	27	–		Coupling agent (MAPE) UV stabilizer (TiO_2) Pigments	1 1 1	Compounding – extrusion	Stübs et al. (2012)
		50%			PVC	47	–		Thermostabilizer (phenol) Lubricant (wax) Pigments	1 1 1		
Automo-tive part	1	40%	–	–	PP	60	–		–		Injection moulding	Thamae & Baillie (2008)
Transport material	2	20%	–	–	PLA	80	–		–		Extrusion blending – injection moulding	Qiang et al. (2012)
						55			PHA	25		
Lab-scale prototype	1	30%	–	–	PP	70	–		–		Compression moulding	Xu et al. (2008)
Lab-scale prototype	2	30%	–	–	PLA TPS	35 35	–		–		Injection moulding	Mahalle et al. (2014)
					PLA	70						

2.3.1 Products under study

Ad 1) Product categories

Accordingly to the main applications of WPC (Carus et al. 2015b; Eder 2013), most LCA studies focused on WPC outdoor decking. The three LCA studies dealing with outdoor decking compared WPC decking to solid wood decking of deep-pressure treated pine (*Pinus sylvestris*) and tropical Bilinga (*Nauclea diderrichii*) (Stübs et al. 2012); California redwood (*Sequoia sempervirens*) (Bergman et al. 2013); alkaline copper quaternary treated lumber of unknown species (Bolin & Smith 2011). Auto interior parts were considered in one study. Thamae & Baillie (2008) investigated the replacement of glass fibre reinforced PP car door panel with wood particles. Three LCA studies were conducted without a relation to the current application fields of WPC. Qiang et al. (2014) assessed the environmental performance of WPC transport pallets based on their laboratory scale study PLA-based WPC (Qiang et al. 2012). Xu et al. (2008) and Mahalle et al. (2014) studied the environmental performance of WPC preforms and prototypes made under laboratory conditions.

Ad 2) WPC formulations

The wood content of the analysed WPC decking were in the range of 50–70% from virgin roundwood (Stübs et al. 2012) and co-products from wood processing industry (Bergman et al. 2013; Bolin & Smith 2011). Other LCA studies of WPC focused solely on wood particles from industrial roundwood with a wood content range of 20–40%. Waste wood or wood residues from forest management were not considered. The wood species were spruce or pine. No hardwood wood species were analysed. All studies used different nomenclature for the wood content such as wood flour, wood powder, and wood fibre. No study included a particle analysis, such as provided in Benthien et al. (2016). It can be assumed that the geometries can be best described as *wood particles*.

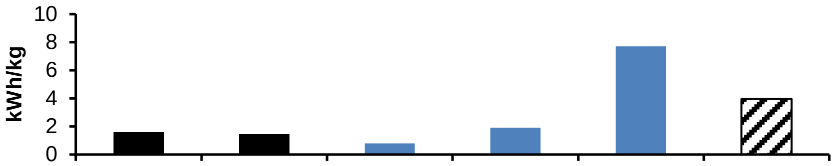
On the plastics side, PE and PVC matrices were analysed in the reviewed studies of WPC decking. PP is missing. The LCA studies focused on the use of post-consumer HDPE (Bolin & Smith 2011) and PE from plastic bags (Bergman et al. 2013) were both performed in N-American context. Other studies focused solely on virgin thermoplastics. Qiang et al. (2014) and Mahalle et al. (2014) studied WPC blends of biodegradable PLA and thermoplastic starch (TPS).

Two of three LCA studies of WPC decking (Bergman et al. 2013; Stübs et al. 2012) considered additives in varying amounts. Qiang et al. (2014) evaluated polyhydroxyalkanoates (PHA) what was case specific for the PLA-WPC blends which were mechanically characterized in a previous study (Qiang et al. 2012). As a result, additives were rather poorly represented in the reviewed studies although they are crucial for the bonding between polar wood and the non-polar plastics (Klyosov 2007).

Ad 3) Foreground WPC process techniques and ad 4) Energy demand

Table 4 presents the specific energy demand for foreground manufacturing techniques. The energy demand for extruded WPC were almost the same: 1.4 kWh/kg (Bergman et al. 2013) and 1.6 kWh/kg (Stübs et al. 2012). Energy demand for injection moulded WPC ranged from 0.8 kWh/kg for a car door panel (Thamae & Baillie 2008) to 1.92 for a transport pallet (Qiang et al. 2012), and to 7.71 kWh/kg for a prototype (Mahalle et al. 2014). Energy demand for manual hydraulic heated press was 3.95 kWh for 90 sheets with a sheet geometry of 127 x 127 x 2 mm^3 (Xu et al. 2008). Bolin & Smith (2011) did not mention the manufacturing process. Considering the total energy input amount of WPC production, upstream processes were identified with a share of 92% of the total energy demand (Qiang et al. 2014). Mahalle et al. (2014) detected the foreground bio-composite production process as most energy intensive (90%) regarding cradle-to-gate whereas both studies were based on the same PLA from NatureWorks LLC. The sources of energy were not discussed in the studies.

Table 4. Energy demand (kWh/kg) of foreground WPC manufacturing processes
*Energy demand for 90 preform sheets

		Products under study					
Process step	**Process**	*Decking*		*Car door panel*	*Trans-port pallet*	*Prototype*	*Preform sheet**
Raw Material preparation	Wood fibre drying	–	–	–	–	–	1.17
Intermediate step	Compounding	1.11		–	0.44	5.59	–
	Heating-cooling mixer	0.12	–	–	–	–	–
	Grinding	–	0.01	–	–	0.25	–
	Drying	–	–	–	–	0.52	–
Conditioning	Extrusion	0.37	1.42	–	–	–	–
	Injection moulding	–	–	0.8	1.48	1.35	–
	Compression moulding	–	–	–	–	–	2.78
Cumulative Energy demand cradle-to-gate		1.6	1.4	0.8	1.9	7.7	3.9
Authors		Stübs et al. (2012)	Bergman et al. (2013)	Thamae & Baillie (2008)	Qiang et al. (2014)	Mahalle et al. (2014)	Xu et al. (2008)

2.3.2 Applied LCA methodologies

Table 5. Results of literature review of LCA of WPC: Applied LCA methodological approaches. *cradle-to-gate analysis

	Outdoor decking	Outdoor decking	Outdoor decking	Preform sheets	Car door panel	Transport pallet	prototype
Functional unit	1,000 board feet	Production, 15 years use and disposal of 1 m^2 decking (incl. the substructure)	100 ft^2 (9.29 m^2) of decking material with a service life of 25 years and the thickness depending on material selection	a) kg/m^3 b) Material service density	A car door panel of volume 992 cm^3 for service life of 200,000 km	1,000 kg of a transport pallet	1 kg of biocomposite (prototype)
LCI approach	ALCA	ALCA	ALCA	ALCA	ALCA	ALCA	ALCA
LCI geographical data	N-America (USA)	Germany	N-America (USA)	Australasia (AU + NZ)	Europe	China	N-America (USA and Canada)
Allocation approaches	–	Economical values	–	–	Energy recovery	–	–
Post-consumer EoL scenario and data	Landfill based on assumptions	Incineration based on LCA background dataset, Assumptions for material recycling	Landfill based on literature information	–*	10% incineration, 90% landfill based on newsprint LCA background dataset	–*	–*
LCIA method	TRACI 2009	CML 2001	TRACI 2.1	Eco-Indicator 99	Eco-indicator 95, Eco-Indicator 99, CML 2001, EPS 2000	TRACI 2009 + Multi Criteria Decision Tool	TRACI
Authors	Bolin & Smith (2011)	Stübs et al. (2012)	Bergman et al. (2013)	Xu et al. (2008)	Thamae & Baillie (2008)	Qiang et al. (2014)	Mahalle et al. (2014)

Ad 5) Functional unit

As shown in Table 5, all studies used different functional units (FU). Thamae & Baillie (2008), Stübs et al. (2012) and Qiang et al. (2014) integrated time-related conditions in the FU. Only Stübs et al. (2012) studied the biological degradation behaviour of WPC decking in outdoor conditions to calculate the life expectancy of a WPC decking. The technical life expectancy was investigated through experimental laboratory tests and climate tests under real conditions. Questionnaires were carried out among WPC-customers to gain additional information of the product's use stage. The authors also compared hollow WPC decking to solid WPC decking by including the substructure in the FU.

Xu et al. (2008) considered next to a mass-related FU (kg/m^3) the "material service density" which considers the requirement of the wood-fibre-reinforced PP composite to withstand a given mechanical load, specifically the tensile load – expressed as material service density. To withstand the same tensile load, 81% additional PP in mass was needed.

Ad 6) LCI approach

All reviewed studies had the goal to compare the ecological footprint of specific WPC formulations to each other or compare WPC products to other products, such as solid wood decking (Bergman et al. 2013; Bolin & Smith 2011; Stübs et al. 2012) or reinforced plastics with non-renewable fibres (Thamae & Baillie 2008). By substituting secondary plastics in the WPC matrix, the authors stated the environmental advantages when comparing to WPC made of virgin resources.

As discussed earlier in Section 1.4 (LCA), such statements need macro-economic analyses to deal with consequences or evaluate the application considerations, such as done by Xu et al. (2008). None of the other studies included a CLCA approach, nor clearly stated the use of ALCA data, though they obviously used data of the latter. Plevin et al. (2014b) provided a good example for interpreting the LCA results based on the ALCA approach: "*We estimate that the ALCA rating of product X is Y% lower than that of product Z, though this does not imply that producing more of X results in a Y% reduction. To infer the actual climate impact of an action affecting the use of X and Y requires a change-based (consequential) analysis.*"

Ad 7) Allocation and ad 8) Post-consumer scenario

Allocation in foreground WPC manufacturing (gate-to-gate) was not considered in most cases due to the internal recyclability of WPC residues. Allocation in upstream processes (supply chain) were solved by the use of LCA background databases. Studies which did not use LCA databases for upstream processes, used data from laboratory data (Xu et al. 2008) or from literature (Qiang et al. 2014) and allocated the environmental burdens on physical parameters.

Concerning post-consumer WPC, waste management and recycling activities leading to multi-functional outputs for allocation methods are well discussed (Frischknecht 2010; Klöpffer & Grahl 2012; Nicholson et al. 2009; Rigamonti et al. 2009; Sandin et al. 2014; Schrijvers et al. 2016b). Frischknecht (2010) stated that the choice of the allocation approach depends on the LCA practitioner's affiliation. Choosing the appropriate allocation method tends to be a subjective decision with great impact on the LCA results. Scientific efforts have been made to develop a single-formula for all recycling situations (Manfredi et al. 2015; Pelletier et al. 2014) but this is heavily criticized (Schrijvers et al. 2016a) and is likely to supports biased LCA (Finkbeiner 2014).

Among the WPC studies, Thamae & Baillie (2008) applied the "avoided burden" approach in the EoL stage. Incineration of wood fibre and PP content was closed-loop modelled so that the recoverable energy by incineration of WPC replaces primary energy within the system boundary. This means, that derived potential credits of energy recovery were subtracted from the environmental loads of WPC production. Stübs et al. (2012) modelled the EoL according to the available technology in the year 2012, which comprises bulk waste collection and incineration of PE-/PP-/PVC-WPC in German facilities with energy recovery. Additionally, a potential material recycling scenario was assumed. In this scenario, 50% of post-consumer WPC were recycled to substitute virgin WPC. The other 50% were incinerated with energy recovery.

Ad 9) Life Cycle Impact Assessment (LCIA) and ad 10) Resource efficiency

The CML-IA method (Guinée 2002), also referred as CML 2001, was used by Stübs et al. (2012). CML-IA is a midpoint LCIA method and specifically developed for the European context. Endpoint indicators, such as the Eco-indicator for example, are weighted and aggregated LCIA results. Klöpffer & Grahl (2012) stressed that weighting and aggregation of LCIA results are scientifically questionable practices and suggest avoiding this issue by documenting the weighing process of each indicator. For instance, Qiang et al. (2014) used the un-weighted TRACI method first, then the multi-criteria decision tool "attribute hierarchy model" by showing the pairwise significance of the LCIA of WPC.

Resource depletion or resource efficiency indicators were not applied in the reviewed studies. Klinglmair et al. (2014) in accordance with, Heijungs et al. (1997) and Finnveden et al. (2009) stated in a review about the assessment of resource depletion in LCA, that no LCIA methodology provides full coverage of these issues. Assessing abiotic resource efficiency and especially biotic resource depletion potential has not been well or at all established in the LCIA methods.

3 Aim and scope of the thesis

Based on the literature reviews and status quo of raw material utilization for WPC, the EoL situation and ecological considerations, a thorough feasibility study of substituting secondary materials as well as the identification of an environmentally sound EoL pathway is missing. The aim of this thesis is to highlight the feasibility of substituting secondary materials by identifying potential secondary materials from specific waste streams, their availability and price situation within the German system boundary. The identified secondary materials are further processed to WPC specimens, which are physically characterized and environmentally assessed by means of LCA. The structure of the discussion of the thesis reflects the end-of-waste criteria (Table 6) as outlined in Article 5 (1) of the KrWG (German Government 2012), which is essential for the identification of substitution potentials of secondary materials as well as for the EoL of WPC.

Table 6. End-of-waste criteria in context of substitution potentials and EoL

End-of-waste criterion	**Reflects**	**Examples related to WPC**
The substance or object is commonly used for specific purposes.	Qualitative aspects	Do secondary plastics and wood achieve comparable physical properties?
There is an existing market or demand for the substance or object.	Market mechanisms	Which potential secondary materials are available for WPC and is the substitution economically feasible?
The use is lawful (substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products).	Legislative and normative aspects	Is the content of hazardous substances below the threshold values of the requirements of normative product standards?
The use will not lead to overall adverse environmental or human health impacts.	Ecological aspects	What is the ecological profile of WPC from secondary materials?

Thereof, the overarching research questions are derived:

(1) Which secondary materials from which waste streams can be identified to substitute primary (virgin) materials in WPC production?

(2) What are the differences in physical and mechanical properties of WPC produced from secondary materials in comparison to their virgin counterparts?

(3) What obstacles need to be considered in terms of applicability?

(4) What is the difference of secondary vs. virgin materials in WPC based on LCA?

(5) What is the ecological preferable EoL pathway of the composites

Peer-reviewed paper I

Sommerhuber, P.F., Welling, J., and Krause, A. 2015. Substitution potentials of recycled HDPE and wood particles from post-consumer packaging waste in wood-plastic composites. *Waste Management* 46:76–85.
DOI:10.1016/j.wasman.2015.09.011.

Overarching research questions reflected in this paper:

(1), (2), (3)

Specific research questions discussed in this paper:

- Which **legislative frameworks** and **market constrains** need to be considered if post-consumer **waste wood of category A II** and **high-density polyethylene (HDPE)** from packaging waste substitute virgin materials in WPC?
- What is the difference in **physical and mechanical properties** of WPC from these secondary materials in comparison to WPC from virgin materials?
- What are the differences in **colour** of the composites in terms of consumer acceptability?

Abstract

The market share of wood-plastic composites (WPC) is small but expected to grow sharply in Europe. This raises some concerns about suitable wood particles needed in the wood-based panels industry in Europe. Concerns are stimulated by the competition between the promotion of wood products through the European Bioeconomy Strategy and wood as an energy carrier through the Renewable Energy Directive. Cascade use of resources and valorisation of waste are potential strategies to overcome resource scarcity. Under experimental design conditions, WPC made from post-consumer recycled wood and plastic (HDPE) were compared to WPC made from virgin resources. Wood content in the polymer matrix was raised in two steps from 0% to 30% and 60%. Mechanical and physical properties and colour differences were characterized. The feasibility of using cascaded resources for WPC is discussed. Results indicate the technical and economic feasibility of using recycled HDPE from packaging waste for WPC. Based on technical properties, 30% recycled wood content for WPC is feasible, but financial and political barriers need to be overcome.

Peer-reviewed paper II

Sommerhuber, P.F., Wang, T., and Krause, A. 2016. Wood-plastic composites as potential applications of recycled plastics of electronic waste and recycled particleboard. *Journal of Cleaner Production* 121:176–185.
DOI:10.1016/j.jclepro.2016.02.036.

Overarching research questions reflected in this paper:

(1), (2), (3)

Specific research questions discussed in this paper:

- Can WPC be produced from secondary **acrylonitrile-butadiene-styrene (ABS)** and **polystyrene (PS)** from waste of electrical and electronic equipment (WEEE)?
- What is the difference in physical and mechanical properties of ABS- and PS-WPC with **recycled particleboard (A II)** in comparison to Norway spruce?
- What content of **inorganic potentially hazardous substances** can be expected and how does the content affect the applicability of the composites in terms of **legal frameworks** and **product standards**?

Abstract

Wood-plastic composites were injection-molded from recycled acrylonitrile-butadiene-styrene and polystyrene from post-consumer electronics in the interest of resource efficiency and ecological product design. The wood content was raised in two steps from 0% to 30% and 60%. Reinforcement performance of recycled particleboard was compared to virgin Norway spruce. Styrene maleic anhydride copolymer was used as the coupling agent in the composites with a 60% wood proportion to investigate the influence on interfacial adhesion. The composites were characterized by using physical and mechanical standard testing methods. Results showed increased stiffness (flexural and tensile modulus of elasticity), water uptake, and density with the incorporation of wood particles to the plastic matrices. Interestingly, strength (flexural and tensile) increased as well. Wood particles from Norway spruce exhibited reinforcement in terms of strength and stiffness. The same results were achieved with particleboard particles in terms of stiffness, but the strength of the composites was negatively affected. The coupling agent affected the strength properties beneficially, which was not observed for the stiffness of the composites. The presence of cadmium, chromium, copper, arsenic and lead in the recycled resources was found by an elementary analysis. This can be linked to color pigments in recycled plastics and insufficient separation processes of recycled wood particles for particleboard production.

Peer-reviewed paper III

Sommerhuber, P.F., Wenker, J.L., Rüter, S., and Krause, A. Life cycle assessment of wood-plastic composites by applying product and system methodological approaches. *Resources, Conservation and Recycling* (Under Review).

Overarching research questions reflected in this paper:

(4), (5)

Specific research questions discussed in this paper:

- What is the **ecological difference** of WPC made from virgin vs. secondary resources?
- How can **ecological parameters be linked to physical parameters** in terms of substitution potentials?
- What is the **environmentally preferable EoL** option of the composites?

Abstract

In the drive towards a sustainable bioeconomy, a growing interest in the development of composite materials made of plastics compounded with wood particles, known as wood-plastic composites (WPC), can be observed. Wood is seen as one of the cornerstones for sustainable economic growth, while the use of thermoplastics from hydrocarbon fossil resources and additives for WPC potentially cause severe environmental impacts along the entire life cycle. In this study, the life cycle stages of raw material supply and end-of-life pathways of WPC were assessed environmentally from different perspectives with life cycle assessment (LCA). The utilization of alternative raw materials reflected the WPC producer's point of view. Harmonized product LCA standards were applied and combined with physical parameters of actually produced composites to give credit to substitution potentials in terms of resource quality. The downstream pathways of post-consumer WPC products reflected the recycler's perspective. A system LCA approach was needed where systems with equal functions were generated to secure a comparison of end-of-life treatment systems. Results showed that WPC produced from secondary materials is the ecologically and technically superior alternative. Recycling of the composites would be the ecologically preferable pathway, but is limited in application due to current recycling directives and markets. The share of virgin WPC to secondary WPC in new WPC is a sensitive issue.

Case study: KET ECOLIFE® – WPC window profile

Overarching Research Questions reflected in this case study:

(4)

Specific research questions discussed in this case study:

- What are the environmental hotspots of ECOLIFE® from cradle-to-gate?
- What is the environmental contribution of WPC in a product system?

Abstract

The aim of this LCA was to assess the environmental hotpots for the manufacturer's internal decision support of a window system made of a co-extruded WPC profile. The assembled window is called ECOLIFE®, manufactured by Kappes Environment Technology (KET) in China. The implementation of the LCA in this thesis has multiple reasons. Firstly, although the product is manufactured in China, the quantitatively relevant materials (wood, plastics, and additives) are imported from Germany and Europe. Secondly, KET obtains the same sort of secondary HDPE granulates from packaging waste that were used for the laboratory-scaled WPC in peer-reviewed paper I, which were further analysed with LCA in peer-reviewed paper III. Thirdly, the laboratory-scaled studied secondary plastics and WPC itself are obtained for an actually manufactured product. The LCI is, therefore, based on primary data acquisition. Fourthly, the application field is new for WPC. The product design and category differs significantly from mass-produced and already environmentally assessed WPC decking as the product is assembled in more manufacturing stages with other semi–finished products including packaging materials.

4 Material and methods

4.1 Material flow analysis

Addressing the overarching research question:

(1) *Which secondary materials from which waste streams can be identified to substitute primary (virgin) materials in WPC production?*

In the first step, the availability of secondary materials was identified by simplified material flow analysis, which could potentially substitute the currently used virgin materials in WPC manufacturing. This was done by consulting statistical data of waste flows and primary production using the databases from Destatis and eurostat as well as reports, like market data on German (waste) wood flows (Mantau 2012b, 2015; Mantau et al. 2012; Seintsch & Weimar 2012, 2013) and recycled plastics (Consultic 2014; Plastics Europe 2015; Villanueva & Eder 2014). The geographic system boundary was Germany with a broader view on Europe.

Production data of WPC were taken from nova-Institute (Carus et al. 2015a; Eder 2013; Partanen & Carus 2016). German prices of waste wood, co-products, virgin, and secondary plastics were gathered from different volumes of EUWID *Recycling- and Entsorgung* and EUWID *Holz und Holzwerkstoffe* starting from the year 2005 until 2015.

The software ifu e!sankey (ifu Hamburg 2015) was used for visualization of material flows where applicable.

4.2 Experimental design

4.2.1 Materials

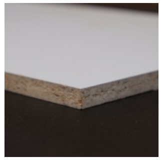
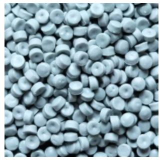
Addressing the overarching research questions:

(2) *What are the differences in physical and mechanical properties of WPC produced from secondary materials in comparison to their virgin counterparts?*

(3) *What obstacles need to be considered in terms of applicability?*

The materials used for experimental design as well as for the ecological characterization are shown in Table 7.

Table 7. Materials used in experimental design and LCA. *measured, **not specified in data sheet and not measured

	Norway spruce	*Mixed waste wood (A II)*	*Particleboard (A II)*	*SABIC® HDPE*	*ALBA recythen® HDPE*	*WEplast PS*	*WEplast ABS*	*MAPE CO/UL EP*	*Licocene PP MA 7452*	*SMA 3000 P*
										
Source	Virgin	Post-consumer mixed AII	Post-industrial (AII)	Virgin	Post-consumer (packaging)	Post-consumer (WEEE)	Post-consumer (WEEE)	Virgin	Virgin	Virgin
Description	Kiln-dried, planed board, without bark	Mix of transport pallets (Norway spruce, Scots pine, European beech) and engineered wood products	18 mm, melamine coated, 8.5% UF	HDPE, white-transparent	Mix of PE-variants with high HDPE ratio, recycled, extruded, grey	Recycled (crushed, sorted, washed) colourful	Recycled (crushed, sorted, washed) colourful	Coupling agent, white-transparent	Coupling agent, yellow	Coupling agent, white
Shape at delivery	Boards	Crushed, >200 mm particle size	Boards	Ø 5 mm granulate	Ø 5 mm re-granulate	Ø 6-8 mm shredded grading	Ø 6-8 mm shredded grading	Ø 5 mm granulate	Ø 5 mm granulate	Powder
Mc-% at delivery	9%	33%*	9%*	–**	0.1%	0.1%	1%	–**	–**	–**
MFI (g/10 mm)	–	–	–	1.8 @ 190 °C/2.2 kg	2 @ 190 °C/5 kg	7 @ 200 °C/5 kg	25 @ 200 °C/5 kg	2 @ 190 °C/2.2 kg	–**	–*
Impurities	–	Particles of plastics, nails and glass	Likeliness of presence of recycled wood particles in core-layer	–	–	Wood (0–0.1%) Rubber (0–0.2%) Flame retardants (0–0.1%) PVC (0–0.1%) PS, PPO (0.5–1.5%)		–	–	–
Presence in peer-reviewed paper Nº.	I, II, III	I, III	II	I, III	I, III	II	II	I, III	–	II

4.2.2 Composite preparation

Figure 2 presents the composite preparation by showing the applied WPC production techniques and utilized materials. The secondary materials were derived from post-consumer packaging sources. They were compounded in a laboratory-internal mixer, grinded in a Retsch grinding mill with a mesh size of 8 mm, and compression moulded in a Siempelkamp computerized hydraulic hot press. The specimens had to be cut from sheets to the geometries as called in the testing standards for physical and mechanical characterization (Table 8).

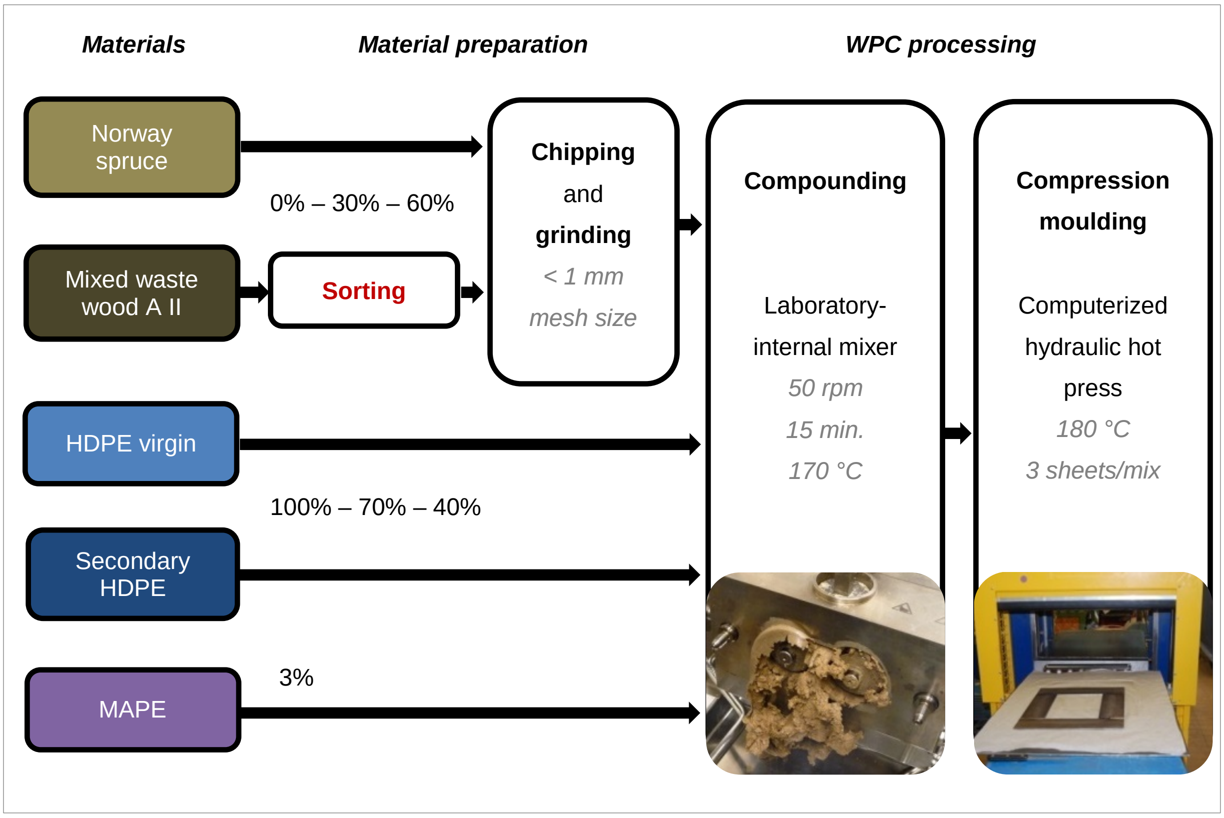


Figure 2. Experimental design of WPC made of post-consumer HDPE and waste wood from post-consumer packaging. Peer-reviewed paper I

Extrusion compounding and injection moulding were used for WPC from secondary plastics from WEEE and particleboards (Figure 3). The specimens were directly produced by the injection moulding tool. Only secondary plastics were chosen for peer-reviewed paper II. This was based on the decision that ABS products are very heterogeneous in context of its molecule ratios which affects processing parameters as the copolymer consists of three different monomers: acrylonitrile, butadiene, and styrene. Styrene maleic anhydride (SMA) was used only in composites with 60% wood proportion to test its influence on the physical and mechanical performance on WPC.

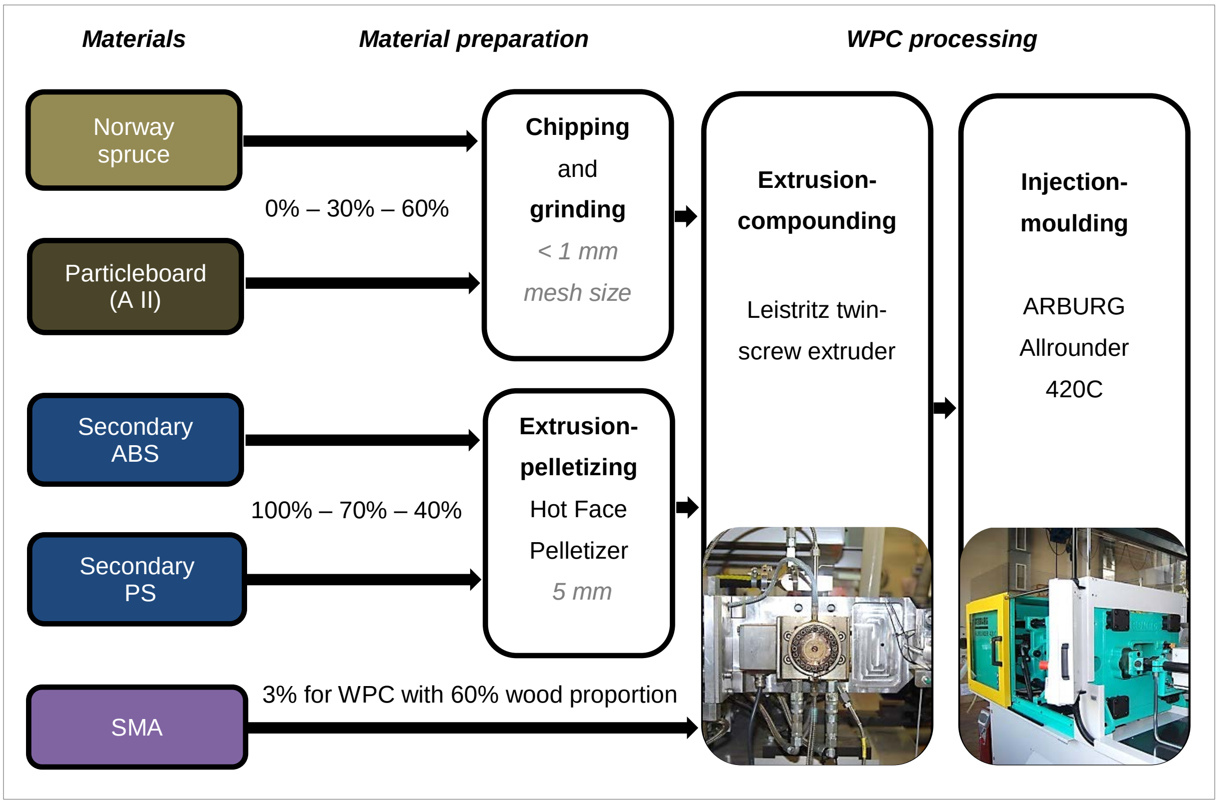


Figure 3. Experimental design of WPC made of post-consumer ABS and PS from WEEE and recycled particleboard. Peer-reviewed paper II

The composites produced with PP-matrix from post-consumer packaging were produced as Figure 5 with two exceptions. The recycled PP, called ALBA PP recythen®, provided by INTERSEROH Dienstleistungs GmbH, Germany, were already homogenously delivered in pelletized geometry and were directly useable for extrusion compounding. The MAPP, called Licocene PP MA 7452 (Clariant), with a density of 0.93 g/cm^3 was used as a coupling agent.

4.2.3 Composite characterization

Physical and mechanical testing was conducted in accordance with material characterization standards as shown in Table 8. Water absorption was calculated by differential weighing of the specimens for each period (at days 0, 1, 2, 4, 7, 14 and 28). Density was calculated according to Archimede's principle based on the assumption that the water used for immersion has a density of 1 g/cm^3. Flexural tests (3-point bending) and tensile tests (Makroextensometer) were conducted using a Zwick/Roell Universal testing machine. A Zwick/Roell HIT5.5P was only used for un-notched Charpy impact strength test for HDPE based WPC in peer-reviewed paper I

Table 8. Composite characterization. *for neat plastics ** for WPC

	Standard	Peer-reviewed paper Nº.	
		paper I	*paper II*
Sample size (n)		12	10
Physical characterization			
Density	DIN EN 15534-1	$Density\ (g/cm^3) = \frac{W_0}{V_0}$	
Moisture absorption	DIN EN 15534-1	$Water\ absorption\ (\%) = \frac{W_t - W_0}{W_0} * 100$	
Mechanical characterization			
Flexural tests	ISO 178		
Crosshead speed (mm/min)		2	5
Load cell (kN)		5	5
Tensile tests	EN ISO 527-3		
Crosshead speed (mm/min)		10* 1**	1
Load cell (kN)		10	5
Charpy impact strength (un-notched)	ISO 179		Not tested
Energy of pendulum (J)		1	

Scanning electron microscopy (SEM) was conducted to investigate morphology characteristics of WPC specimens and to evaluate the wood and polymer matrix distribution. Specimens for SEM were prepared by cutting a square of 3–5 x 2 mm^2 from tested specimen. Vapour coating of the SEM samples was done in a BioRAD SC 510 SEM Coating System. The surface microstructure was analysed in a Quanta FEG Type 250.

According to published work of, i.e., Schlummer et al. (2007), Dimitrakakis et al. (2009), Oguchi et al. (2013), heavy metal content in secondary materials was expected. The Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) method was used to analyse the following elements: arsenic (As), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), and lead (Pb) in secondary HDPE, PP, ABS, PS, mixed waste wood A II, and particleboard. The samples were separately ground and dissolved in Aqua regia with multi-element standard solution IV Certipur to determine the previously mentioned elements. The analysis was conducted with an inductively coupled plasma atomic emission spectroscopy iCAP 6300 duo ICP/OES.

4.3 Life cycle assessment

Addressing the overarching research questions:

(4) *What is the difference of secondary vs. virgin materials in WPC based on LCA?*

(5) *What is the ecological preferable EoL pathway of the composites*

LCA was applied in peer-reviewed paper III and in the case study of a window profile called ECOLIFE®. The ALCA method was used in all studies. The studies differ in their goal and scope for what product LCA and system LCA methodologies were needed.

4.3.1 Product LCA

This thesis comprises two product LCA studies. First, the laboratory-scaled WPC compounds were environmentally assessed that were produced and studied in peer-reviewed paper I. Second, ECOLIFE® is an actually produced building product by Kappes Environment Technology (KET) with the window frame made of a co-extruded WPC profile wrapped in aluminium stripes. An overview of the main differences of the two product LCA studies is provided in Table 9.

Table 9. Product LCA studies

LCA step	Laboratory-scaled WPC compound	Product system ECOLIFE®
Goal and scope	Assessment of the environmental parameters of virgin vs. secondary materials crafted WPC compound Combination of environmental and technical parameters	Assessment of the environmental hotspots of WPC window product system Contribution of WPC to overall environmental parameters of ECOLIFE®
Functional unit	1 kg WPC compound	1 window measuring 1.23 x 1.48 m^2
LCI Data	Literature and laboratory data	Primary data from manufacturer
LCA databases	Thünen Institute´s ÖkoHolzBauDat GaBi professional database ecoinvent database	
WPC-manufacturing	Compounding Compression moulding	Extrusion-compounding Injection-moulding Co-extrusion
Site of manufacturing	Germany	China

Both studies were calculated in accordance with the EN 15804 (CEN 2013) and EN 16485 (CEN 2014c), which the latter specifies the LCA methodology for wood particles. The standards sets a convention for several issues where the ISO 14044 leaves space for adjustment (Wenker et al. 2015). Time and space related issues of production, use and EoL are addressed in the EN 15804 by dividing the life cycle of a product into modules (Figure 4). The manufacturing stage comprises raw material supply (module A1), transport to manufacturing site (module A2) and the manufacturing of the product (module A3), hence a cradle-to-gate analysis for which attributional, steady state data need to be collected – thus an ALCA. Besides from the manufacturing stage, the use stage (module B) and EoL stage (module C) can be reported separately based on scenario information. In addition, and necessary for transparent reporting on potential environmental impacts, the standard provides instructions on allocation and on separating environmental burdens from benefits and loads beyond the system boundary in module D.

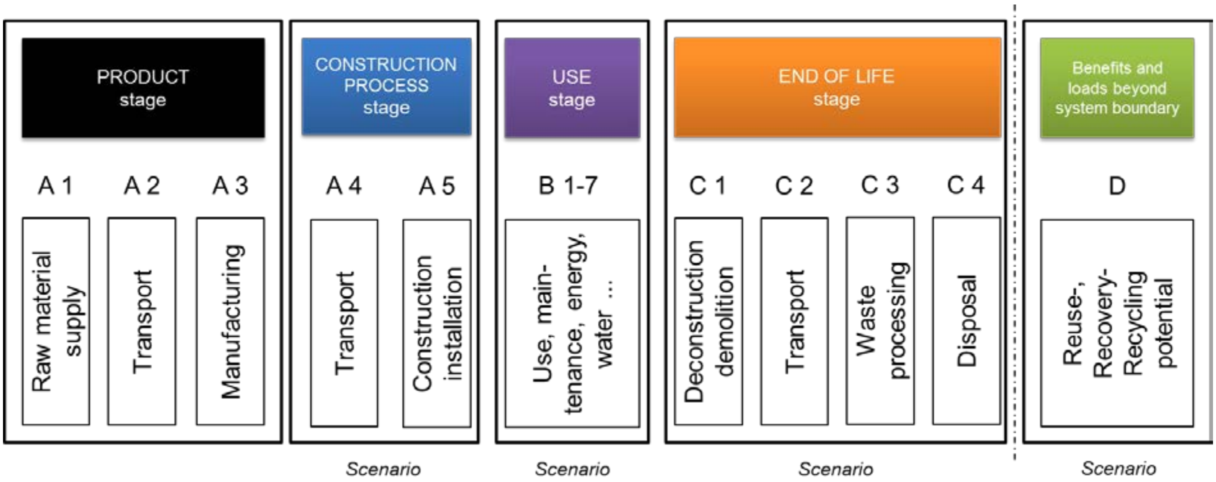


Figure 4. System boundary and modules according to EN 15804 (CEN 2013)

4.3.1.1 Laboratory-scaled WPC compound

The system boundary is visualized in Figure 5 for the WPC compounds made from virgin or secondary materials. The specific LCA information of the semi-finished WPC compounds is well described in Section 5.3 (Peer-reviewed paper III). In a sensitivity analysis, the environmental parameters were linked to the mechanical parameter tensile modulus of elasticity (MoE).

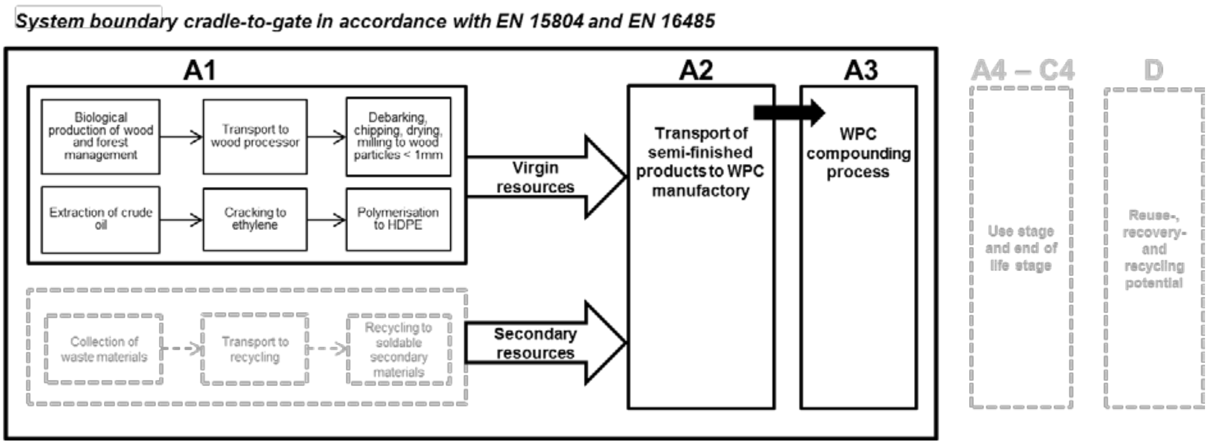


Figure 5. System boundary of product LCA. Dotted lines and boxes mean cut-off

4.3.1.2 Case study ECOLIFE®

KET is a German/Chinese manufacturer producing the WPC window system called ECOLIFE® in China. The system boundary is visualized in Figure 6. Specifically, the product category rule Part B was applied: "Requirements on the EPD for windows and doors" of the Institute Construction and Environment e. V. (IBU 2015c).

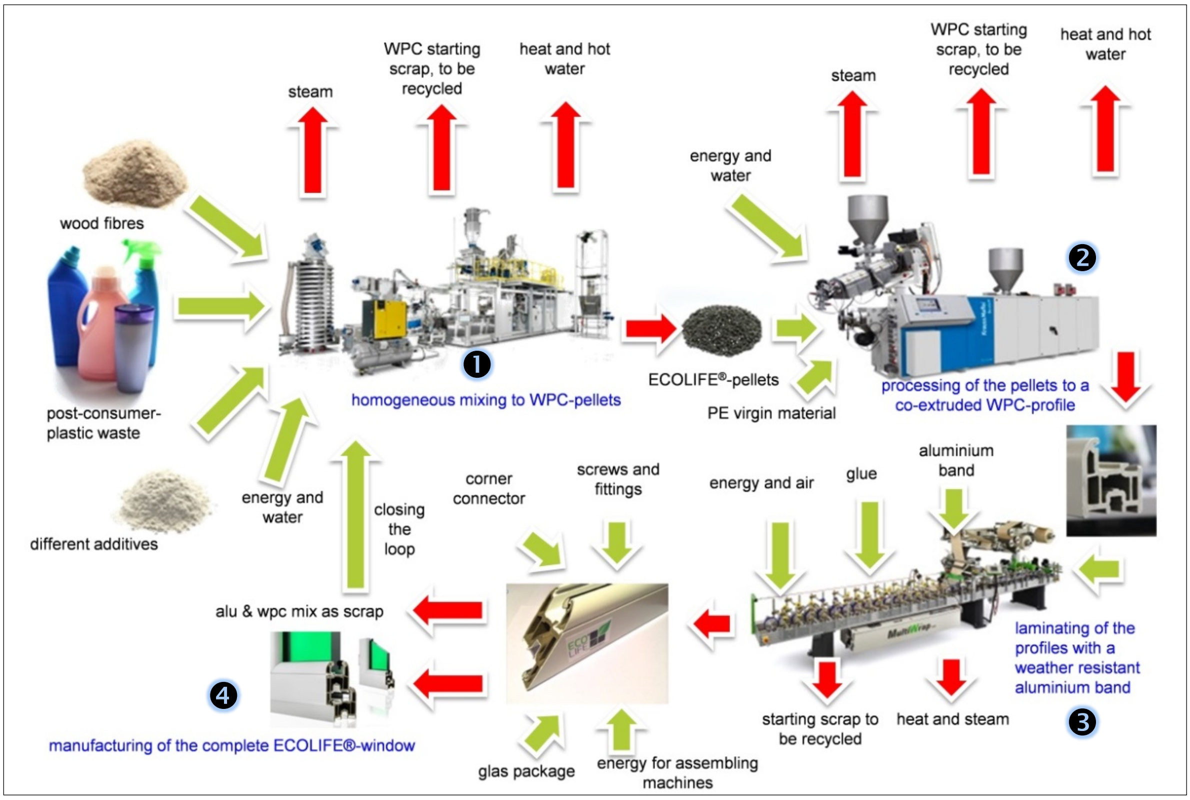


Figure 6. System boundary of KET ECOLIFE®

P1 – WPC Granulating. Wood fibres from the wood processing industry are homogeneously mixed with secondary HDPE from post-consumer packaging waste and different additives by using a co-rotating parallel twin screw extrusion line. WPC scrap can be added in a certain percentage.

P2 – Co-extrusion to WPC window profile. From the WPC pellets a co-extruded profile (virgin HDPE co-extrusion layer) is manufactured by using a counter-rotating parallel screw combined with a single-screw extruder for the co-ex-layer. The profile is fed through a die, a dry calibration and a water cooling stage, followed by a haul-off, a cutting saw and a stacking table.

P3 – Aluminium coating. The co-extruded profile is covered with a thin aluminium band by gluing this band to the inner and outer surface of the profile after a special surface treatment.

P 4 – Window fabrication. Finally the complete windows are assembled.

The LCA is a cradle-to-gate analysis and comprises the modules A1 – A3 in accordance with the EN 15804 (CEN 2013):

- **Raw Material Supply (A1)**

Table 10 presents the WPC mixture of the WPC profile (P 1–2). This module further comprises the provision of precursor and intermediate materials, such as 1.46 m^2 of insulation glass 2-panes, 1 piece of aluminium fittings, PUR glue, gasket and screws.

Table 10. Raw materials for production of co-extruded WPC profile

Inputs	**Share**
Virgin wood powder	40%
Secondary HDPE	28%
Virgin HDPE	20%
Additives	11%
Outputs	
Co-extruded WPC profile	100%

- **Transport (A2)**

KET imports the following raw materials to China, which were modelled in the LCA software with a container ship of 27,500 dead weight tonnes with heavy fuel, containing 1 w-% sulphur for 20 x 10^3 km transport distance: virgin wood powder (GER), secondary HDPE (GER), additives (EU), aluminium sheet (GER), PUR hot melt (GER), fittings (GER) and injection moulded PVC accessories (GER). The remaining resources are obtained regionally in China. A lorry was assumed with 17.3 load capacity at 85% utilization for 50–300 km. The diesel was considered as a China-specific average mix with 0.32 w-% from renewable bio-resources.

- **Manufacturing (A3)**

The company uses the average Chinese energy_{el} grid-mix for onsite energy_{el} demand. This grid-mix consists mainly of 77% hard coal, 17% hydropower, 2% nuclear, and 2% natural gas. Energy_{th} is used for on-site heating purposes. This module further comprised the usage of grease, compression air and diesel for machinery, process water, and water treatment system. Production of packaging materials were assigned to this module.

4.3.2 System LCA

Although the EN 15804 provides guidance on allocation of multi-outputs in the EoL stage, it is not applicable for the assessment of the preferable waste management option of WPC. Following the standard, the environmental loads of waste collection are categorized in C1, transport to recycling site in C2, recycling and energy recovery in C3, incineration and disposal in C4. At first sight, an allocation could be used in module C3 and split into recycling and energy recovery. However, this would not be possible to assess the best ecological EoL pathway, because according to EN 15804, a scenario should be chosen for the EoL of the *single* product under study. The post-consumer WPC can, therefore, be either 100% recycled or 100% incinerated, or as Stübs et al. (2012) partly recycled and partly incinerated the WPC decking, but the EoL treatment systems cannot be compared to each other.

The generated function (or benefits) of recycling WPC to secondary material (kg) differs to secondary fuel (MJ). The ISO 14044 states (ISO 2006) "*Systems shall be compared using the same functional unit and equivalent methodological considerations* [...]". This is done by equalizing the systems by, for instance, expanding the product system, which does not provide the additional functions (ISO 2006). The additional function is added to this system by adding the single supply chain of the missing function, the so called *complementary function* (Fleischer & Schmidt 1996). As a result, the same function respectively the same functional unit is provided by both systems by expanding the function.

The method is referred as the "basket-of-benefits" – Nutzenkorbmethode (Jungbluth & Firschknecht 2006; Klöpffer & Grahl 2012). A general visualization of this method is presented in Figure 7. The LCI for the EoL processes was based on literature data and is well described in peer-reviewed paper III (Section 5.3).

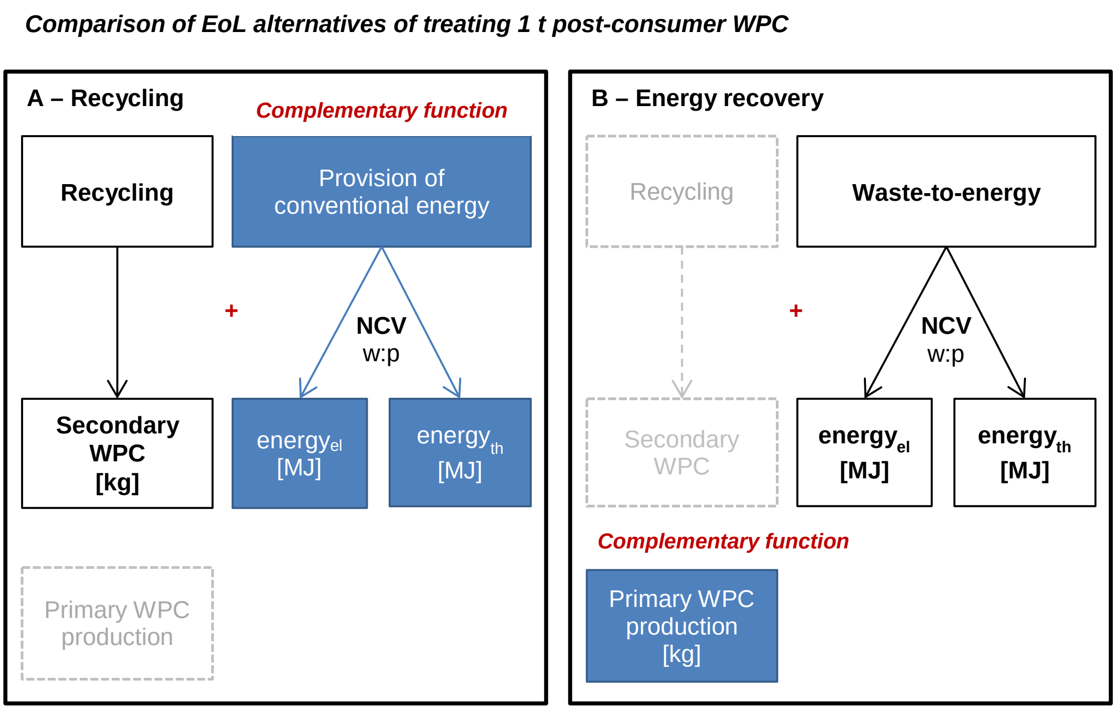


Figure 7. Comparing two recycling systems by ensuring functional equivalency – the basket-of-benefits approach; based on Klöpffer & Grahl (2012)

NCV … net calorific value
Black box and lines… primary function
Blue box… complementary function
Grey dashed boxes… outside system boundary

4.3.3 How to account for material inherent properties

Considering EN 15804, wood inherent biogenic carbon and primary energy used as raw material are treated as material inherent properties. According to EN 16485, the assumption of biogenic carbon neutrality of wood is valid for wood from countries that have decided to account for article 3.4 of the Kyoto Protocol, or for wood originating from forests, that operate under established certificates schemes for sustainable forest management. Under the described circumstances, the biogenic carbon content of wood, expressed as CO_2 in the global warming potential (GWP) parameter, is transferred to the product system as -1 in the product stage in module A1 and A3. The wood inherent carbon leaves the product system as +1 within modules A1 and A3 for wood burned for energy generation within the production, and within the EoL stage (module C3) for the actual product itself, because it is contained in the recycled wood which has reached the end of waste status. Hence, the biogenic carbon balance of a wood product is deemed to be neutral considering all modules from A to C. Summing up these aspects, biogenic carbon neutrality can be assumed for virgin wood particles derived from German forests as well as co-products from the sawmill industry. Furthermore, wood inherent carbon contained in recycled waste wood is considered the same way as it still is a material inherent property in secondary materials.

4.3.4 LCA software, databases and impact assessment method

All LCAs were carried out by using thinkstep GaBi ts LCA software version 7.2.0.8 with GaBi professional database v6.115 (thinkstep 2015) as well as ecoinvent v2.2 database (ecoinvent Centre 2010). Additionally, background data for wood-based materials was taken from Thünen Institute´s ÖkoHolzBauDat project (Rüter & Diederichs 2012). The environmental impact mid-point method CML-IA (Guinée 2002) was used, as demanded by the EN 15804 (CEN 2013): global warming potential (GWP), depletion potential of the stratospheric ozone layer (ODP), acidification potential of land and water (AP), eutrophication potential (EP), formation potential of tropospheric ozone photochemical oxidants (POCP), abiotic depletion potential for non-fossil resources (ADPE), abiotic depletion potential for fossil resources (ADPF).

A complete EPD as required by the Institute of Institute Construction and Environment e. V. (IBU 2015c) for example, contain environmental parameters of resource use (i.e., use of primary energy resources, use of net freshwater), as well as output flows and waste categories (i.e., hazardous waste disposed, components for re-use) of a product. Environmental results in this thesis focused only the environmental impacts as described above, as the expressed potential impacts on the environment resulting from the product stage and EoL stage of WPC.

5 Publications and additional results

5.1 Peer-reviewed paper I

Title	**Substitution potentials of recycled HDPE and wood particles from post-consumer packaging waste in wood-plastic composites**

Authors	Philipp F. Sommerhuber	(CD 70%, EX 100%, ED, 70%)
	Dr. Johannes Welling	(CD 10%, EX 0%, ED 10%)
	Prof. Dr. Andreas Krause	(CD 20%, EX 0%, ED 20%)
Journal	Waste Management	
DOI	DOI:10.1016/j.wasman.2015.09.011	

CD… Conceptual Design

EX… Conducting experiments

ED… Editing

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Substitution potentials of recycled HDPE and wood particles from post-consumer packaging waste in Wood–Plastic Composites

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ABSTRACT

The market share of Wood–Plastic Composites (WPC) is small but expected to grow sharply in Europe. This raises some concerns about suitable wood particles needed in the wood-based panels industry in Europe. Concerns are stimulated by the competition between the promotion of wooden products through the European Bioeconomy Strategy and wood as an energy carrier through the Renewable Energy Directive. Cascade use of resources and valorisation of waste are potential strategies to overcome resource scarcity. Under experimental design conditions, WPC made from post-consumer recycled wood and plastic (HDPE) were compared to WPC made from virgin resources. Wood content in the polymer matrix was raised in two steps from 0% to 30% and 60%. Mechanical and physical properties and colour differences were characterized. The feasibility of using cascaded resources for WPC is discussed. Results indicate the technical and economic feasibility of using recycled HDPE from packaging waste for WPC. Based on technical properties, 30% recycled wood content for WPC is feasible, but economic and political barriers of efficient cascading of biomass need to be overcome.

1. Introduction

The European Bioeconomy strategy towards a sustainable growth has led to a considerable promotion of using wood and wood-based products. Wood resources are facing a strong competition between material and energy utilization (European Commission, 2012; Pülzl et al., 2014). Wood prices have increased due to the high price for fossil energy, triggered by fossil fuel substitution, which has negatively affected the profitability of the wood processing industry. Resource efficiency is a predominant topic for the future of the wood processing industry in Europe. In addition to the Roadmap for a Resource Efficient Europe (European Commission, 2011), the term Circular Economy, which aims at reducing both input of virgin materials and output of wastes by closing economic and ecological loops of resource flows, is on the rise (Haas et al., 2015). The cascading use of biomass, which aims at recycling waste as a secondary resource prior to energy recovery, is highlighted as a solution to prevent resource scarcity and price volatility (Höglmeier et al., 2013; Keegan et al., 2013; Höglmeier et al., 2014). The potential for a closed loop recycling system appears to be considerable, as 50–70% of waste wood is recovered directly as energy instead of being recycled for material utilization in products (Meinlschmidt et al., 2013).

Wood–Plastic Composites (WPC) are finding more and more acceptance due to their low moisture absorption, low density, resistance to biological attack, good dimensional stability, and a combination of high specific stiffness and strength (Valente et al., 2011; Zimmermann and Zattera, 2013). WPC tend to be a good intermediate step in the cascade chain of biomass and are recyclable (Migneault et al., 2014; Teuber et al., 2015). According to Eder (2013), WPC production and use are expected to increase further in Europe. This will result in increasing competition for wood resources, which need to be affordable and which should be derived from sustainably maintained forests. In addition, plastics derived from fossil-based hydrocarbon sources for WPC imply significant environmental impacts.

In this study, waste is defined as a material with no function, therefore co-products from sawmill are not defined as waste as some studies have considered, i.e., Nourbakhsh et al. (2010), Boeglin et al. (1997). Several studies have been published investigating secondary resources for WPC. Studies focusing solely on waste wood for WPC are, i.e.: Gozdecki et al. (2015), Krause et al. (2013), Zimmermann and Zattera (2013), Chen et al. (2006), Balasuriya et al. (2003). Migneault et al. (2014) investigated a HDPE-WPC matrix with 20–40% proportions of various virgin wood

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species and lignocellulose residues, pointing at wood and wood cellulose content as the crucial diminishing factor of WPC properties. Polyolefins (PE, PP) are known be well sortable from other plastics (i.e., PET, PVC) in the recycling process to substitute virgin polyolefins in secondary products with good mechanical properties (Hu et al., 2013). A large number of studies have been published focusing on recycled HDPE for WPC, i.e., Adhikary et al. (2008), Cui et al. (2010), Cui et al. (2008), Kazemi-Najafi et al. (2006), Selke and Wichman (2004), Yam et al. (1990), stating WPC can be successfully manufactured using recycled HDPE. In addition, WPC from bioplastics have been studied, but the currently high prices of bio-plastics prevent its further use (Kim, 2014).

The following studies investigated mechanical properties of WPC made from both recycled wood and plastics. Kamdem et al. (2004) compared mechanical properties of WPC made from chrome copper arsenate (CCA)-treated wood particles blended with virgin and recycled HDPE. Chaharmahali et al. (2008) studied the possibility of producing wood–plastic panels from medium density fibreboard (MDF) and particleboard (PB) with 60–80% wood content as filler with recycled HDPE from milk bottles using a melt-blend/hot-press method. Shalbafan et al. (2013) demonstrated that flat pressed WPCs can be produced from residues of light-weight foam core particleboards which consisted of wood particles (with cured urea–formaldehyde resin) and expanded polystyrene (EPS). The wood flour content ranged from 75% to 88%. The panels were produced by crushing the used foam core particleboards and adding up to 2% coupling agent.

A common conclusion among these studies was the similarity of the mechanical properties of WPC made from recycled resources compared to WPC made from virgin resources. Incorporation of wood particles in the pure plastic matrix resulted in increasing density, flexural strength, flexural and tensile modulus of elasticity, water absorption, and decreasing impact strength and tensile strength.

However, no previous research has been conducted on WPC in the context of a growing and competing biomass demand and the political, technical, economic feasibility, and customer acceptance of using secondary resources for WPC from specific waste streams. The first part of this article investigates market information, availability, and price situation of secondary resources for WPC. The focus is on waste wood and polyolefin plastics from a specific waste category: post-consumer packaging. In the second part, laboratory manufactured WPC specimens made of post-consumer recycled resources are characterized by mechanical properties (flexural and tensile tests and impact strength), physical properties (water absorption and density) and colour properties. The geographical focus is Germany.

2. Market information, availability, and price situation of resources for WPC

2.1. Wood particles

WPC production has increased and is projected to increase further, in contrast to the overall decrease of wood-based panel consumption in Europe according to FAOstat (FAO, 2015). Market data on WPC is taken from Eder (2013) and Teuber et al. (2015). In Europe, the increase of WPC production from the years 2010 to 2012 was 15%. Over the years 2012 to 2015, the production was projected to grow by 26%. In Germany, about 53 k tonnes of wood particles and fibres were used by WPC producers in the year 2012. Wood particles used for WPC production constituted only 1% of wooden material consumed by the wood-based panels industry. WPC outdoor deckings comprise the biggest share of the WPC market (67%). Due to self-committed quality standards of German WPC decking producers, the wood has to be derived from FSC/PEFC certified virgin wood and A I post-consumer wood, which is classified as untreated natural wood according to the German Waste Wood Act (German Government, 2003). Further explanation on waste wood categories is presented in the Supplementary Information.

2.1.1. Post-consumer wood availability

Post-consumer wood constitutes 6.3 Mio t/a in Germany. Energy recovery of wood is the predominant end-of-life option (78%). The particleboard industry uses 20% post-consumer wood to substitute virgin particles. Disposal in landfills is almost negligible (Mantau et al., 2012) due to the ban of organic waste materials in accordance to the European Waste Framework Directive 2008/98/EC (European Commission, 2008).

About 1.3 Mio t/a (30%) of post-consumer wood are energetically recycled in small-scale combustion plants. This amount includes waste wood A I and harvested timber products like particleboards without wood preservatives and PVC, categorized as A II, according to the German Waste Wood Act (German Government, 2003). The contamination level of this material is low, which makes it suitable as a secondary resource in products. The largest share of post-consumer wood (5 Mio t/a = 70%) can be classified as A III, which is a mixture of untreated, treated and contaminated waste wood. A separation into fractions of this amount for material recycling (A I–A II) and energy recovery (A III–A IV) would presume an intensified and cost-intensive separation process considering the prohibition of diffusion of hazardous substances through recycling activities as outlined in the German Waste Management and Product Recycling Act (German Government, 2012) and the European Waste Directive 2008/98/EC (European Commission, 2008). Automated computerized sensor sorting based on detection processes like near infrared spectroscopy (NIR), ion mobility spectrometry (IMS) and X-ray-fluorescence analysis are promising techniques for the cascading use of wood (Meinlschmidt et al., 2013), but are not used up to now in large scale post-consumer wood flows.

2.1.2. Price situation of wood particles

The competition between material and energy utilization of wood has led to an increase of the market prices of waste wood particles (Fig. 1).

Prices of waste wood of category A I increased 3% on average per year from 2005 to 2015, and are tending to further increase

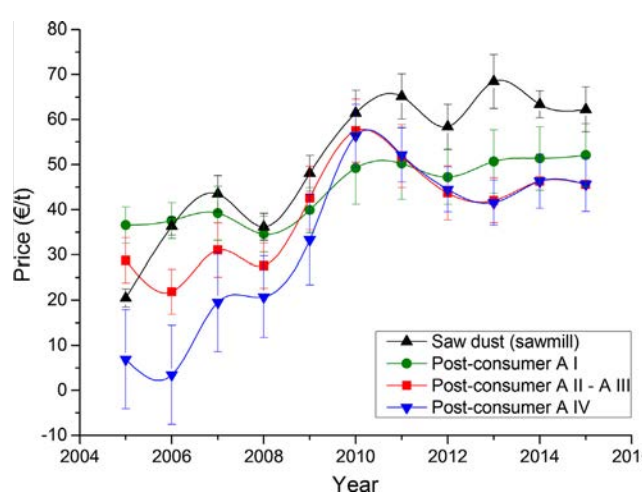

Fig. 1. Price development of wood particles from co-products and post-consumer wood in Germany in €/t. Particle size of post-consumer wood is <150 mm. Post-consumer wood moisture content is 33%. Prices are based on absolute dry mass. Sources: adapted from EUWID Recycling und Entsorgung (2005–2008) and EUWID Holz und Holzwerkstoffe (2009–2015).

slightly. Prices of waste wood from waste wood-panels (A II–A III) increased 4% on average per year in the same period. Contaminated waste wood (A IV) increased 19% in the same period. The increase in prices can be related to feed-in tariffs for thermal electricity produced from biomass based on the German Renewable Energy Act (German Government, 2014). Price changes of A II, A III and A IV have been very volatile, tending to decrease slightly. Market prices of lignocellulose virgin wood particles and fibres from round wood for WPC production with particle size <1 mm was about € 400/t in 2015 in Germany. Detailed information regarding the mean values of price development per year, standard deviation, and conversion factors are given in the Supplementary Information.

2.2. Thermoplastic polyolefins for WPC

The total crude oil use by 2012 was about 4.1 billion t of which 5% (205 Mio t) were used globally by the plastic industries (Franke et al., 2014). The yearly consumption of plastic converters in the EU-27 plus Norway and Switzerland is about 46 Mio t. Of that, 10% (4.6 Mio t) are of recycled origin. About 60% are consumed by the packaging industry (18 Mio t) and the building & construction industries (10 Mio t) (Villanueva and Eder, 2014). In 2011, Germany produced 10 Mio t of plastic granulates, of these 30% were converted into packaging materials.

2.2.1. Post-consumer plastics availability

About 1.5 Mio tonnes/year of packaging waste (HDPE, LDPE, PP, metals etc.) are disposed of separately and collected by *Duales System* licence partners. These activities are based on recovery rates in accordance with the German Packaging Materials Ordinance (German Government, 1998). According to Franke et al. (2014), the collection potential is 40–46% greater because households do not collect this amount separately due to consumer behaviour. Up to now, about 0.5 Mio t of PE and 0.2 Mio t of PP are recycled to secondary resources by specialized plastic recycling facilities each year.

Interestingly, plastic packaging constitutes the largest plastics market share, but plastic packaging materials are intended, in most cases, to remain only briefly in the product life cycle when compared to building products. Haggar and Kamel (2011) mentioned 40% of plastics are used for less than one month. Of these packaging materials, thermoplastic polyolefins (LDPE, LLDPE, HDPE and PP) comprise the biggest share (Plastics Europe, 2015). In WPC production in Europe, thermoplastic polyolefin are the predominant plastic resources. In Germany, about 30 k t of PE and 44 k t of PP were used for WPC deckings production in the year 2012 (Eder, 2013). Compared to the yearly amount of recycled plastic granulates, the thermoplastic polyolefins demand for WPC production could be completely satisfied by recycled granulates from post-consumer packaging materials.

2.2.2. Price situation of polyolefins granulates for WPC production

Recovered plastics prices are not determined by production costs, as they would be in an efficient market. Instead, recovered plastics prices are linked to the price of virgin plastics in the long run. Virgin plastic prices are related to the crude oil price. The price of recycled plastics (600–800 €/t for PE and PP) is 30–50% less than the price of virgin plastics (1200 €/t) in the year 2015 in Germany. Price range depends on the quality standards of the recycled plastic granulates (Villanueva and Eder, 2014).

2.3. Discussion on future development of economic and political challenges of using cascaded resources for WPC

In Germany, energy recovery from biomass is promoted by indirect subventions (feed-in tariffs) which have led to considerable increases of the prices of wooden co-products and waste wood during the last ten years (2005–2015). Financial subventions should be also discussed for material recycling of wooden particles, if the price of virgin wood particles were to increase steeply. These financial incentives may lead to economically feasible, dedicated waste wood recycling facilities with automated sorting techniques to sort efficiently commingled waste wood into specific categories for material recovery (A I–A II) and for energy recovery (A III–A IV). For category A II a great potential exists for cascading use in wood products, such as WPC, in the national waste wood flow.

Currently, fluctuations in prices of wooden particles are likely to be not as critical for the WPC-working industry as they are in the conventional wood-based panel industry with more than 90% wood particles proportion in the products, i.e., particleboard, MDF. The price of specialized (0.1 mm) virgin wood particles is about 400 €/t, 62 €/t (±5) for sawdust, 51 €/t (±7) for A I waste wood and 44 €/t (±6) for A II–III waste wood. It has to be stated, regarding particle size, impurities, and low moisture content (<7%), that specialized virgin wood particles are directly useable for compounding with plastics to WPC, when compared to co-products or waste wood. Drying effort from <7% to <1% moisture content is low compared to moisture content of waste wood (15–33%). Other efforts and the related costs of waste wood valorisation such as sorting, grinding, drying, sieving to suitable particle size for compounding to WPC are considerably higher compared to co-products.

In addition, hazardous impurities of waste wood in waste wood categories in which no hazardous impurities should be expected according to the German Waste Wood Directive (Riedel et al., 2014) lead to a potential diffusion of hazardous substances. Diffusion of hazardous substances through recycling activities is prohibited by the Waste Framework Directive 2008/98/EC.

In contrast, plastics and additives are more cost-intensive than wood particles in the WPC industry, which leads to strong dependency on hydrocarbon fossil price changes. Using recovered short-lived plastic packaging materials from fossil resources (i.e., PE, PP) is feasible for WPC as legislative framework exists that regulates compulsory recovery rates for plastic packaging waste (German Government, 1998). This framework helped to establish an efficient recycling market for post-consumer lightweight packaging materials. Recycled polyolefins granulates are available in sufficient quantity and quality to substitute virgin polyolefin in WPC.

However, there is still a great potential to recover more post-consumer plastic waste in Germany as the compulsory recovery rate is rather low (22.5%) for material recycling of plastic. A compulsory recovery rate for post-consumer wood packaging exists as well but is even lower (15%) compared to plastics, metals (50%), glass, paper, and cardboard (60%). Increasing this rate for post-consumer packaging wood and applying the rate on other post-consumer wood categories as well, i.e., furniture, would probably result in a more efficient cascading use of biomass. The trade-off between substituting recycled wood in WPC and biomass as an important renewable energy carrier needs to be considered and should be investigated by, i.e., Life Cycle Assessment.

3. Materials

3.1. Wood

Waste wood was provided by Buhck Umweltservices GmbH & Co. KG, a local recycling company in Hamburg, Germany, in October 2014. The waste wood was stored without enclosures and was chipped at the recycling site to a particle size of <200 mm. It was predominantly composed of post-consumer transport pallets and some post-consumer harvested timber products with a high soft wood content. The biggest proportion was identified as Norway spruce followed by a small amount of Scots pine particles. The hard wood content consisted mostly of European beech. Some impurities like metals (nails) and plastics (polystyrene granulates, packaging materials) were identified by visual inspection at delivery at the laboratory. The waste wood category was classified as A II according to the German Waste Wood Directive (German Government, 2003). Kiln dried virgin Norway spruce was used for comparison purposes.

3.2. Plastics

Recycled HDPE, called recythen® HDPE, was provided by INTERSEROH Dienstleistungs GmbH, Germany. It is a recycled material from a post-consumer recycling process. The raw material is based on HDPE and may contain small amounts of other PE types. It can be used for injection moulding and extrusion. The melt flow index (MFI) was 2 g/10 min according to DIN ISO 1133. Virgin HDPE was chosen based on the MFI properties of the recycled HDPE for comparison. The virgin HDPE, called SABIC® HDPE CC253, was provided by Saudi Basic Industries Corporation (SABIC). The MFI was 1.8 g/10 min according to DIN ISO 1133.

3.3. Additives

Maleic anhydride polyethylene (MAPE), called Compoline CO/UL EP, was used as the coupling agent for the compounding process and was provided by Auserpolimer S.R.L., Italy. The MFI was 2 g/10 min according to ISO 1133.

4. Methods

4.1. Grinding

The moisture content of waste wood was 33%, determined by the oven drying method (103 °C for 24 h) after delivery. Waste wood was manually sorted from nails before reducing the particle size from <200 mm to <100 mm by a counter blade cutter.

The moisture content of the virgin wood particles was 9%, determined by the oven-drying method (103 °C for 24 h). Virgin spruce boards did not need to be sorted from impurities for cutting. Waste and virgin wood chips were separately processed further to <1 mm mesh size in a *Retsch* laboratory grinder.

4.2. Compounding

Virgin and recycled HDPE and MAPE were obtained directly usable for compounding. For that purpose, wood particles had to be oven-dried (103 °C for 72 h) to be able to blend with HDPE and MAPE. A laboratory internal mixer named *HAAKE Reomix 3000 OS* with tangential co-rotating twin-screw extruder geometries at 50 rotations per minute was used for compounding processes for 15 min. Material was fed manually. The temperature for compounding was set to 170 °C. To enhance the compounding of wood particles and HDPE, coupling agents are used in WPC manufacturing. In this study, MAPE was used as the coupling agent. Wood contains many hydroxyl groups for the esterification reaction with MAPE, which enhances the mechanical strength of WPC (Migneault et al., 2014).

4.3. Compression moulding

The WPC mixtures were further processed in a *Retsch* grinding mill with a mesh size of 8 mm. The milled WPC-compounds and pure HDPE (virgin and recycled granulates) were pressed in a metal frame size of $250 \times 170 \times 4\ mm^3$ using a *Siempelkamp* computerized hydraulic hot press. The temperature of the press plates was set to 180 °C. First, the pressure was set to 20 bar for 370 s. The upper plate was lifted shortly for 10 s to release water vapour from wood particles of the compound. Then the pressure was increased to 60 bar for 117 s and to 100 bar for 22 s. Finally, the temperature of the plates was decreased to 90 °C for 169 s. The 100%-plastic-plates were produced slightly different to the compounds to decrease shrinkage. After final pressing at 100 bar, the temperature of the plates was decreased to 60 °C instead of 90 °C with a slower cool down below 120 °C.

The experimental design is presented in Table 1.

4.4. Characterization

Three sheets ($250 \times 170 \times 4\ mm^3$) were produced for each specimen group (A–F) by compression-moulding. Dumbbell-shaped specimens ($170 \times 10 \times 4\ mm^3$) were mill-cut from sheets with a rotary cutter for tensile testing. Rod-shaped specimens ($80 \times 10 \times 4\ mm^3$) were sawn using a circular saw from sheets for all other tests. Specimens were conditioned and tested in 20 °C/65% relative humidity.

Mechanical characterization is presented in Table 2.

A water absorption test was conducted according to DIN EN 15534-1. The water absorption of the compounds was calculated by differential weighing of the specimens for each time period (at Days 0, 1, 2, 4, 7, 14 and 28) after total immersion into demineralized water at a temperature of 20 °C (±2). Water absorption was calculated according to the formula:

$$\text{Water absorption } (\%) = \frac{W_t - W_0}{W_0} * 100 \quad (1)$$

where W_t is the specimen weight after a given immersion in water and W_0 is oven dry mass of the specimen weight after constant mass was reached (weight change <0.1% after 24 h).

Density of the specimens was calculated on the assumption that the water used for immersion has a density of 1 g/cm^3 and was calculated according to the formula:

$$\text{Density } (g/cm^3) = \frac{W_0}{V_0} \quad (2)$$

where W_0 is oven dry mass of the specimen weight and V_0 is the Volume of the specimen. Ten specimens were used for water absorption and density properties.

4.5. Colour

Differences in the colour of the specimens were compared using the CIElab colour method, which evaluates the dimension of lightness ($L*$) and dimension of colours ($a*$) and ($b*$). Values of $L*$ range from black (0) to white (100). Values of $a*$ range from red (+50) to green (−50). Values of $b*$ range from yellow (+50) to blue (−50). Specimens were scanned using an *Epson Expression 10000 Scanner* and evaluated in *Adobe Photoshop* in the $L * a * b*$ colour spectrum.

Table 1
Experimental design: *p* indicates virgin content, *r* indicates recycled content.

Wood content	0%		30%		60%	
Resources specimen groups	A	B	C	D	E	F
	pPE	rPE	pWPC	rWPC	pWPC	rWPC
Waste wood A I-II	–	–	–	29%	–	58%
ALBA recythen® HDPE	–	100%	–	68%	–	39%
Virgin spruce	–	–	29%	–	58%	–
SABIC® HDPE CC253	100%	–	68%	–	39%	–
MAPE	–	–	3%	3%	3%	3%

Table 2
Mechanical characterization methods.

Characterization	Standard	Testing machine	Method	Crosshead speed (mm/min)	Load cell (kN)	Energy of pendulum (J)	Sample size
Flexural tests	ISO 178	Zwick/Roell Universal testing machine	3-point bending	2	5	–	12
Tensile tests	EN ISO 527-3 (Type 1 B)		Makro-extensometer	10 (specimens A & B) and 1 (C-F)	10	–	12
Charpy impact test	ISO 179-1 (1eU[b]), un-notched	Zwick/Roell HIT5.5P		–	–	1	12

5. Results and discussion of experimental design

5.1. Physical properties

5.1.1. Density

Densities were almost equivalent in the specimen groups based on the wood content level (A & B, C & D, E & F). Increase in density of WPC was independent of whether virgin or recycled resources were used. WPC densities were in the range of 1.05(±9E−4)–1.17 (±2E−3) g/cm^3 and correlated well with literature (Klyosov, 2007; Migneault et al., 2014). The density of wood particles was not investigated in detail but the density was almost equal for both virgin and waste wood particles. Since the wood cell wall has a density of approx. 1.4 g/cm^3, no significant porosity is found in the material. The density increased linearly in both pWPC and rWPC, based on almost equal densities of pHDPE A 0.95(±1E−3) g/cm^3 and rHDPE B 0.96(±1E−3) g/cm^3. It can be further assumed that the contamination with inorganics, i.e. metals, sand, was very low in the laboratory-recycled waste wood.

5.1.2. Water absorption

As expected, water absorption was very low for pure plastic immersed in water (pPE – A 0.07% and rPE – B 0.04% after 28 days). At 30% wood content, water absorption increased to 1.27% (pWPC – C) and 1.38% (rWPC – D) after 28 days. At 60% wood content, recycled specimens (rWPC – F 6.12%) exhibited water absorption compared to virgin specimens (pWPC – E 3.93%).

The increase of water absorption of rWPC and pWPC depending on the wood content is well known. Since wood cell wall consists of hydroscopic substances like carbohydrates and lignin it exhibits a water uptake. The higher the wood proportion in the WPC matrix, the more likely water will be absorbed. Butylina et al. (2011) mentioned possible incomplete encapsulation of wood fibres and probable occurrence of wood fibre aggregates in a PP-matrix as additional influences on the water absorption properties of WPC.

The water absorption is higher in the case of rWPC. This may be linked to hydrophilic anionic surface-active agents (soaps) in addition to the previously mentioned decisive influences. Tensides are used for washing sorted plastics waste in the recycling process of HDPE granulates. These tensides reduced the surface tension of water, which negatively affected the sorption behaviour of wood particles.

WPC projected continuous water absorption beyond 28 days, regardless of whether virgin or recycled resources were used (Fig. 2).

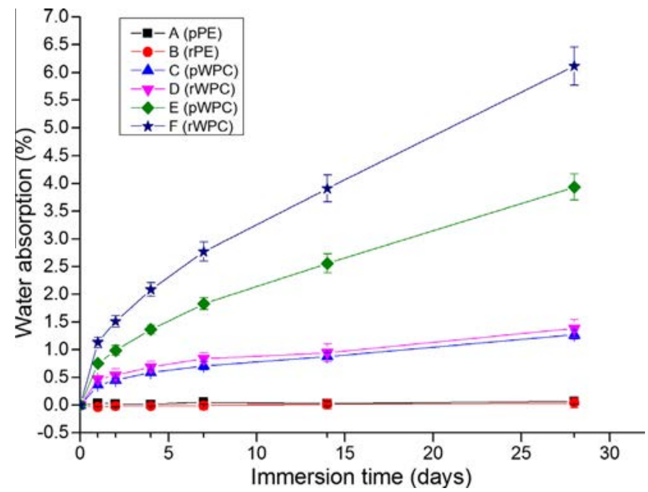

Fig. 2. Water absorption.

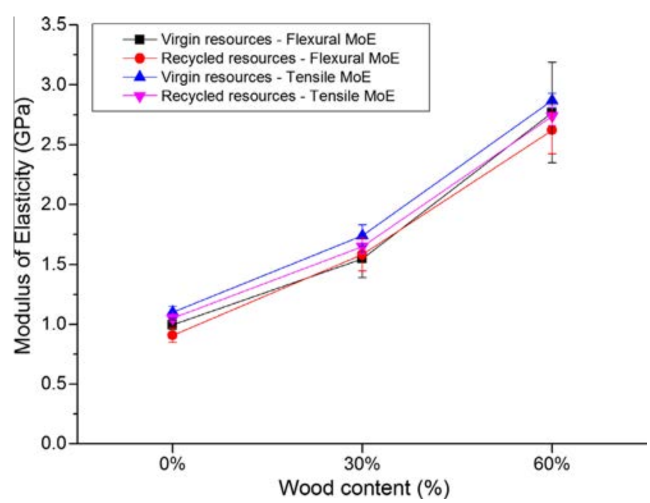

Fig. 3. Flexural and tensile modulus of elasticity.

5.2. Mechanical properties

5.2.1. Modulus of elasticity

Fig. 3 presents the modulus of elasticity derived from flexural (fMOE) and tensile (tMOE) tests. tMOE correlated well with fMOE in each specimen group. The 100% polymer matrix specimens constituted the lowest fMOE (A–0.99 GPa; B–0.91 GPa). By adding wood to the polymer matrix, stiffness increased linearly to 2.77 GPa (pWPC – E) and 2.67 GPa (rWPC – F). Virgin and recycled specimen exhibited very comparable results, but virgin materials were always a little higher. Since the wood modulus is higher than the plastic modulus, the composite modulus increased with increasing wood content. An increase of stiffness of the composites, by incorporating wood particles to the HDPE matrix was also observed by, i.e. Migneault et al. (2014), Adhikary et al. (2008), Razi et al. (1997).

5.2.2. Tensile properties

Virgin and recycled HDPE showed comparably high elongation at F_{max}. These values are much higher than measured on WPC specimens. With increasing wood content, the elongation was reduced to approx. 1% at 60% wood content. Virgin and recycled specimens were not significantly different. Tensile strength decreased by adding 30% wood content (pWPC – C 18.5 MPa and rWPC – D 15.7 MPa) with a slight increase by adding 60% wood content (pWPC – E 21.6 MPa and rWPC – F 16 MPa). Fig. 4 presents tensile strength on the left side and elongation at F_{max} on the right.

The drop in tensile strength by the incorporation of wood in the polymer matrix is a known disadvantage of WPC regardless whether virgin or recycled HDPE is used, which was also reported by Migneault et al. (2014). HDPE is a ductile material, as can be seen in the elongation at F_{max} of specimens A and B. Reinforcement was not expected since the wood used was in the form of particles and not fibres. The reduction in tensile strength was relatively low, compared to the reduced amount of plastic in the cross section showing that the wood particles were able to take some load from the plastic independent of the origin of the wood.

A proposition to achieve better tensile properties of WPC could be the incorporation of wood fibres to the polymer matrix, as investigated by Butylina et al. (2011) in a PP-matrix. Using smaller wood particles may lead to better tensile properties, as observed by Razi et al. (1997). However, Migneault et al. (2014) stressed that the correlations between aspect ratio of particles and the WPC were insignificant to the properties of tensile strength development. Using wood particles of <1 mm (mesh size) seemed appropriate for the experimental part, as WPC properties are mainly affected by the wood proportion, applied process, and the additives (Migneault et al., 2014; Krause et al., 2013; Kumari et al. 2007).

5.2.3. Flexural strength

Flexural strength is presented in Fig. 5. The flexural strength increases with increasing wood content. Virgin materials showed an increase from 25.4 MPa (pPE – A) to 33.6 MPa (pWPC – C) to 42.6 MPa (pWPC – E), while recycled material exhibited a lower increase. This could be due to higher tensile strength of the virgin composite. The wood exhibited a certain reinforcement effect in the flexural test. Compared to the reduced tensile strength it can be assumed that compression strength is increased to a high extent.

Flexural strength of 30% WPC made from recycled resources showed similar strength compared to pWPC, but showed no further increase for 60% wood recycled material. Lower properties of rWPC with 60% waste wood content were measured for tensile strength as well. It is expected that this behaviour is due to impurities in recycled plastics and recycled wood, which hindered a proper interface formation between the two materials. As non-HDPE plastic particles, i.e., small amounts of Polystyrene (PS)

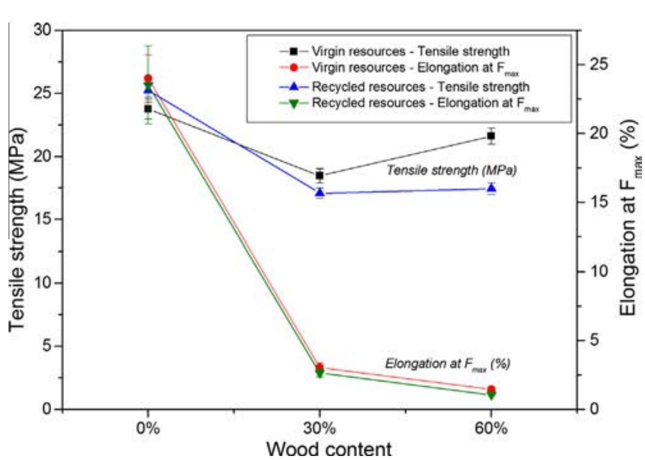

Fig. 4. Tensile strength and elongation at F_{max}.

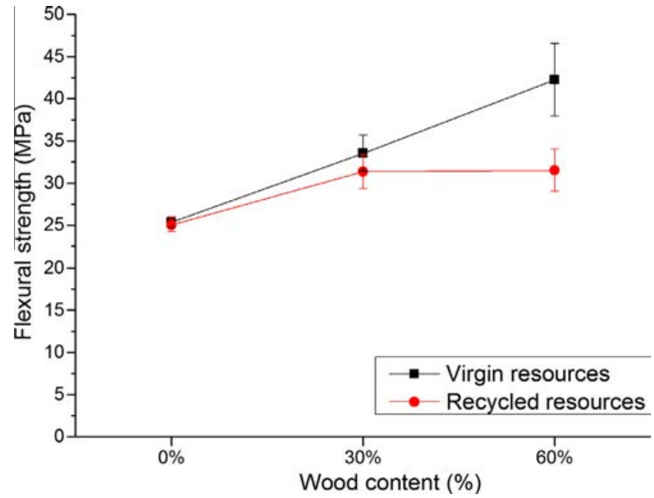

Fig. 5. Flexural strength.

particles were found during the visual inspection in the waste wood, the possibility of immiscibility may have occurred in the compounding process of rWPC – F, which probably resulted in poor mechanical properties. Immiscibility of plastics is a common shortcoming of plastic recycling and is discussed by Kazemi-Najafi (2013) with relation to recycled plastics in WPC. The impurities can lead to a less good envelopment of the single wood particles resulting in a weaker composite as well.

The poor interfacial interaction can be traced back to the laboratory compounding and compression moulding processes as rheological flow ability decreases for HDPE with increasing wood particle proportions. This would prove that a full interfacial adhesion was not developed in the 60% rWPC – F composite, compared to the 60% pWPC – E, as densities of E 1.17(±2E−3) g/cm^3 and F 1.16(±3E−3) g/cm^3 were slightly different. Scanning Electron Microscopy (SEM) pictures are presented in Fig. 6 and discussed in Section 5.3.

5.2.4. Charpy impact strength

All specimens were tested un-notched. The Charpy impact strength of the pure HDPE specimens (A and B) could not be tested with a 1 J pendulum because no break occurred in either specimen. A and B were further tested with a 6 J pendulum and withstood the applied energy as well. The impact strength of pWPC specimens made of virgin content was higher than rWPC specimens in each wood proportion group. Specimens made with 30% wood proportion resulted in 8.4(±1.5) kJ/m^2 (pWPC – C) and 7.6(±1.5) kJ/m^2 (rWPC – D). Specimens made with 60% wood proportion resulted in 8 ± 2.4 kJ/m^2 (pWPC – E) and 5.4 ± 1.07 kJ/m^2 (rWPC – F). The same observation was made that the composites containing 30% wood were almost equal, as the higher wood proportion led to reduced impact strength for recycled material. The reasons for this behaviour are the same as discussed for the other properties. Foremost, impact strength of WPC is affected by the wood proportion in the matrix. Wood particles decreased the impact strength of the polymer matrix, which is in accordance with previous studies (Migneault et al., 2014; Kiaeifar et al., 2011).

5.3. Microstructure analysis

Scanning Electron Microscopy (SEM) was conducted for evaluation of the wood and polymer matrix distribution in composite. Specimens were prepared by cutting a 3–5 × 2 mm^2 square from tested Charpy impact strength samples. The surface cross sections

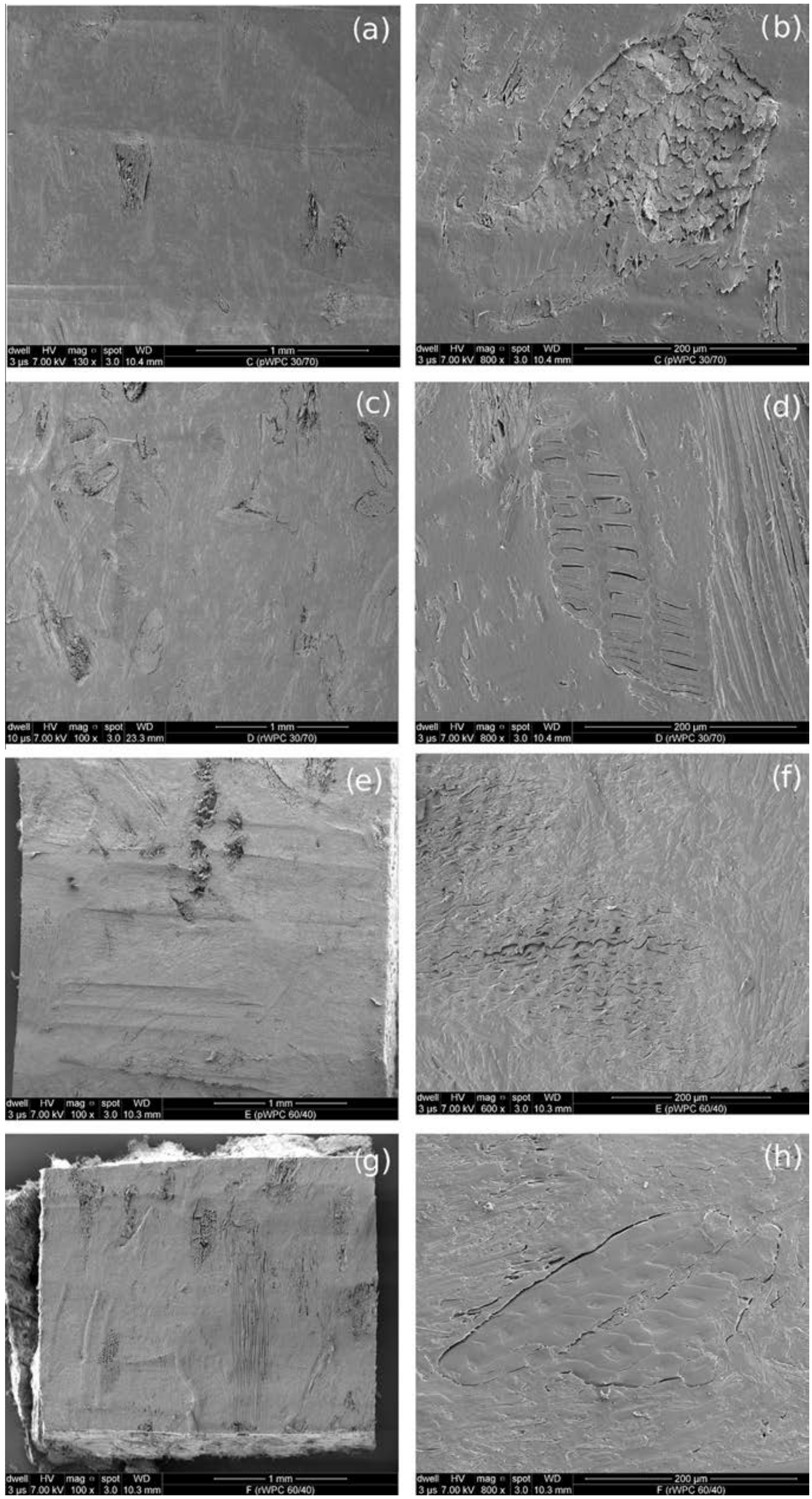

Fig. 6. Scanning Electron Microscopy (SEM), left column 1 mm, right column 200 μm.

were investigated and are presented in Fig. 6. The tested samples showed a close contact between wood cell wall and polymer matrix for all composites. It can be seen on all images that nearly all wood particles were filled with polymer or compressed so that no voids are visible. However, rWPC of 60% wood content showed wood particles with some gaps between wood and polymer, indicating a weak interface. Polymer impregnation into wood lumens differed from particle to particle, as can be seen when comparing

Table 3
Colour characterization.

Specimen	$L*$	$a*$	$b*$	
A pPE (0/100)	96	−1	1	
B rPE (0/100)	48	−2	1	
C pWPC (30/70)	45	4	14	
D rWPC (30/70)	39	−1	5	
E pWPC (60/40)	37	−2	13	
F rWPC (60/40)	32	1	8	

the pictures on the left (1 mm) with those on the right pictures (200 μm). Fig. 6g) presents a good example that the wood particles were heterogeneously oriented in the matrix, which is due to the low mould flow during compression moulding process.

5.4. Colour characterization

Results are presented in Table 3. Virgin HDPE granulates were delivered in white colour. Luminescence ($L* = 96$) of the specimens pPE (A) was close to maximum (100) with colour hues close to zero ($a* = -1$, $b* = 1$). Recycled HDPE granulates were delivered in grey colour. Luminescence ($L* = 48$) of the rPE (B) determined the greyish look of rWPC specimens (D and F) which was also related to the darker colour of recycled wood particles in comparison to virgin wood particles. Virgin WPC specimens (C and E) looked brownish with colour hues of $a*$ close to zero, and a balanced mix of red and green colours, with lower $b*$ values.

WPC deckings are available in a variety of colours ranging reaching from dark–grey to wood-brown and brick-red by adding colour pigments. Colour perception is individual and based on the application, market trends, and geographical market entities. The addition of wood to plastic reduces the possible colours to some extent since light colours are difficult to realise. The use of more dark/grey recycled resources reduces this possibility further. The WPC manufacturers need to offer a variety of different colours to the customer. This is probably a main difficulty for the use of recycled material in WPC.

Høibø and Nyrud (2010) concluded that surface homogeneity of wood products is the predominant visual aspect for customers. Osburg et al. (2015) investigated customer acceptance of WPC among environmentally conscious customers. Customers preferred eco-friendly materials (solid wood) over materials with greater environmental impact (plastics). The more eco-friendly the customer was, the stronger this preference became. WPC perception was in the middle between solid wood and plastic materials.

6. Conclusions

The European WPC market share is small but steadily increasing, in contrast to the wood-based panels market in Germany. Up to now, the demand for wooden particles in German WPC production constitutes 1% of the wood demand of the wood working industry. However, with the promotion of the European Bioeconomy strategy, more wood products are coming to market increasing the pressure on affordable resources and their availability. Using recycled biomass as a substitution material is already implemented in particleboards manufacted in Germany. In WPC manufacturing, smaller particle sizes are used (<1 mm). Inorganic impurities are very likely to damage WPC processing techniques, the moisture content of particles needs to be very low to be able to interface with plastics and additives. Treatment costs to achieve these qualities from waste wood exceed the economic benefits of the substitution potential of waste wood particles compared to co-products.

Successful substitutions of resources or materials need to consider the functional equivalence in addition to economic and political considerations. In the experimental study, waste wood (A II) and recycled HDPE from post-consumer packaging were compounded in a laboratory mixer with MAPE, compression moulded in a computerized hydraulic heated press and compared to WPC made from virgin Norway spruce particles and HDPE. This processing method is not a common, practically applied method, but is often used on the laboratory scale. Representativeness of results would benefit by applying extrusion or injection moulding technique. The overall results showed that virgin and recycled resources were very comparable in many properties such as water uptake and mechanical properties. At high wood content (60%), the recycled material exhibited less good properties than virgin resources. This could be due to the impurities in recycled HDPE and recycled wood. At lower wood content (30%), there were no significant differences between recycled and virgin materials, showing a high substitution potential from a product-design point of view. A difficulty could be the darker colour of the recycled material (both plastic and wood) reducing the possibility to manufacture products in any colour.

To conclude from the perspective of a potential WPC manufacturer:

- *Economic feasibility of using recycled wood (A I – A II) in WPC is questionable*, due to small price difference vs. co-products up to now.
- Using recycled HDPE granulates from post-consumer packaging is economically feasible for WPC.
- *Inorganic residues in waste wood are likely to damage extrusion and injection moulding WPC technology* and need to be sorted carefully from the waste wood.
- *rWPC made of 30% recycled waste wood and 70% recycled HDPE* can be substituted for pWPC made of virgin resources, in applications where stiffness is crucial.
- *Darker hues* of WPC made from recycled resources would not influence customers' preferences.

The following recommendations are proposed for a competing demand for biomass in terms of the European Bioeconomy strategy by upscaling laboratory results on the national perspective of Germany:

- Resource demand for polyolefin-WPC consumption can be completely satisfied by recycled polyolefin and recycled wood.
- *High level and variances of moisture content in waste wood* (15–33%) could be minimized by keeping waste wood enclosed indoors or at least in canopied storages at the recycling site.
- *Increase and expand compulsory rates* of both post-consumer waste wood and plastics.
- *Discuss financial incentives* on political agendas to increase feasibility of waste wood recycling

- *Environmental trade-off(s)* between substituting secondary resources in WPC for using them as an energy carrier for fossil fuel substitution has to be further investigated.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2015.09.011.

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5.2 Peer-reviewed paper II

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Wood–plastic composites as potential applications of recycled plastics of electronic waste and recycled particleboard

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ABSTRACT

Wood–plastic composites were injection-molded from recycled acrylonitrile–butadiene–styrene and polystyrene from post-consumer electronics in the interest of resource efficiency and ecological product design. The wood content was raised in two steps from 0% to 30% and 60%. Reinforcement performance of recycled particleboard was compared to virgin Norway spruce. Styrene maleic anhydride copolymer was used as the coupling agent in the composites with a 60% wood proportion to investigate the influence on interfacial adhesion. The composites were characterized by using physical and mechanical standard testing methods. Results showed increased stiffness (flexural and tensile modulus of elasticity), water uptake and density with the incorporation of wood particles to the plastic matrices. Interestingly, strength (flexural and tensile) increased as well. Wood particles from Norway spruce exhibited reinforcement in terms of strength and stiffness. The same results were achieved with particleboard particles in terms of stiffness, but the strength of the composites was negatively affected. The coupling agent affected the strength properties beneficially, which was not observed for the stiffness of the composites. The presence of cadmium, chromium, copper, arsenic and lead in the recycled resources was found by an elementary analysis. This can be linked to color pigments in recycled plastics and insufficient separation processes of recycled wood particles for particleboard production.

1. Introduction

Uncontrolled recycling activities of waste of electrical and electronic equipment (WEEE) cause severe impacts to human health and the environment. Open sky incineration, cyanide leaching and simple smelters to recover precious metals as well as landfilling the residues are common practices in underdeveloped, emerging, and some developed countries. The uncontrolled release of toxic substances including heavy metals, polycyclic aromatic hydrocarbons (PHAs), polybrominated diphenyl ethers (PBDEs) and many other hazardous molecules through crude recycling methods, cause severe impacts at the local, regional and global levels (Kiddee et al., 2013; Premalatha et al., 2014).

The European Union (EU-25) generates about 8.9 × 10^6 t of WEEE each year. Of that, 66% are domestically recovered within the EU and 18% (1.8 × 10^6 t) are exported outside the EU. A significant amount of WEEE is therefore recycled under the aforementioned undesirable conditions. In addition, valuable resources are lost through export – often illegal – accounting to a loss of 1.7 × 10^9 EUR/yr. within the EU (Huisman et al., 2015). Revenues from recycling WEEE-plastics constitute about 9% of the total revenues resulting from WEEE recycling (Cucchiella et al., 2015). Of these plastics, about 50% of the mass consist of acrylonitrile–butadiene–styrene (ABS) and polystyrene (PS) (Köhnlechner, 2014; Premalatha et al., 2014; Zoeteman et al., 2010).

It is estimated that 1.5 × 10^6 t of recycled ABS and PS from WEEE will be available as secondary resources in the year 2019 (Köhnlechner, 2014), as the compulsory collection rate of the European Directive 2012/19/EU on WEEE (European Commission, 2012a) shall increase to 65%. In addition, Salhofer et al. (in press) stated that in future exports will be only possible in compliance with the European Waste Shipment Regulation. However, recycling

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is only efficient and useful if a market exists for the recycled resources in accordance with the European Waste Framework Directive 2008/98/EC (European Commission, 2008).

Wood–plastic composites (WPC) are on the increase in Europe (Carus et al., 2015). On the one hand, this will result in an increasing demand for plastics derived from fossil-based hydrocarbon sources, which implies significant environmental impacts. On the other hand, a competing demand also exists for wood resources, which need to be affordable, and should be derived from sustainably managed forests (Mantau et al., 2010). In terms of resource efficiency (European Commission, 2011b) and circular economy (Haas et al., 2015), WPC tend to be a good intermediate step in the cascade chain of resources (Teuber et al., 2016).

In this study it is hypothesized that recycled ABS and PS from WEEE and recycled particleboard offer a great technical potential to apply the resource efficiency and circular approaches in innovative WPC products.

The suitability of substituting recycled crystalline thermoplastics like the polyolefins in WPC have been intensively investigated, i.e., Yam et al. (1990), Kazemi-Najafi et al. (2009), Khanjanzadeh et al. (2012), Kazemi-Najafi (2013) and Sommerhuber et al. (2015). However, research is limited concerning mechanical properties of WPC made from non-crystalline thermoplastics like ABS and PS regardless the origin of resources.

Kuo et al. (2009) showed that the tensile strength and flexural strength of virgin content ABS-WPC with 3% maleic anhydride polypropylene were lower than the pure ABS and specific tensile modulus was 40% lower. Chotirat et al. (2007) demonstrated that as the wood content was increased to a proportion of 33% in a virgin ABS matrix, the flexural and tensile modulus of elasticity (MOE) increased 50% and 63%, respectively, with decreasing flexural, tensile and impact strength. Yeh et al. (2009) compared mechanical properties of WPC made from virgin and recycled post-consumer ABS from unknown sources. The authors stressed that the recycled ABS contained various impurities, and had poor and variable mechanical properties when compared to virgin ABS. However, mechanical properties were mostly affected by the wood content. At a 50% proportion of wood flour, mechanical properties remained unchanged when the virgin ABS in the matrix is replaced by recycled ABS. Tensile modulus of the recycled ABS-WPC with 50% wood flour and 10% styrene maleic anhydride (SMA) was 12% higher than without SMA.

Pracella et al. (2010) investigated virgin PS with cellulose and oat content. PS-WPC was brittle in performance, MOE increased 32% by adding 40% cellulose and 2.5% PS-co-MA, while tensile strength and elongation at break decreased. Increasing the SMA to 5% lowered the increase in MOE to 12%. Wang et al. (2005) investigated the influence of the process factors on the physical properties of WPC panels made from three recycled-plastic packaging materials (PE, PP, and PS). The PS-WPC showed superior properties to PE and PP. The flexural strength was superior to pure PS. With an increasing of wood flour content from 30% to 50%, the specific modulus increased by 57%. Similar results were found in the study from Lisperguer et al. (2010). Lisperguer et al. (2007) studied the effect of wood acetylation on thermal behavior of PS-WPC and stated that acetylated wood flour produces WPC with better thermal stability than non-acetylated wood flour.

Chaharmahali et al. (2008) incorporated recycled particleboard in a HDPE matrix stating that mechanical properties are comparable to virgin wood particles HDPE-matrix. Likewise, Gozdecki et al. (2015) concluded the same behavior in a PP-matrix.

These short reviews support continuing research in WPC made from recycled resources. Therefore, mechanical performance of WPC made from recovered PS and ABS from post-consumer WEEE with recycled particleboard are investigated. In addition, scanning electron microscopy (SEM) evaluates the wood and polymer matrix distribution.

It has to be stated that diffusion of hazardous substances through recycling activities is prohibited by the European Waste Framework Directive 2008/98/EC (European Commission, 2008). Therefore, heavy metal content is investigated using elementary analysis to discuss potential applications.

2. Materials

2.1. Wood

The particleboard was provided by Pfleiderer AG, Germany. Both sides of the 18 mm thick particleboard were coated with a melamine overlay. The glue content (urea-formaldehyde (UF)) was determined based on nitrogen content with an elemental analyzer (vario EL cube, Elementar). UF content was calculated to be 8.5% therefore reflects the average particleboard produced in Germany, i.e., Diederichs (2014). Kiln-dried virgin Norway spruce was used for comparison.

2.2. Plastics

Recycled ABS and PS originating from WEEE, called WEplast, were provided by wersag GmbH & Co. KG, Germany. The specific monomer ratio of ABS was not known. Known properties according to the datasheets of the used recycled PS and ABS are presented in Table 1.

2.3. Additives

Styrene maleic anhydride copolymer (SMA), called SMA 3000 P, was produced by Cray Valley, France and provided by Gustav Grolman GmbH & Co. KG, Germany. SMA was used as a coupling agent to investigate its suitability in recycled ABS and PS-WPC matrices containing a 60% proportion of wood. The SMA had a molar ratio of styrene/maleic anhydride close to 3/1. The resin was provided in powder form.

The experimental design is presented in Table 2.

3. Methods

3.1. Composite preparation

Particleboard and Norway spruce were chipped with a counter blade cutter (BENZ TRONIC PLUS 3000) to a size of $20 \times 5 \times 5$ mm^3. The moisture content of the virgin wood particles was 9%, determined by the oven-drying method (103 °C/24 h). As both wood sources were stored indoors in same relative humidity over a period of >1 month, comparable moisture content for particleboard can be assumed. Particleboard and virgin wood chips were further processed separately in a laboratory grinder (Retsch SM 2000) to <1 mm mesh size. The morphology can best be described as *particles*. The specific morphology of particles was not determined further.

PS and ABS were delivered sorted as colored 'shredder material' with an average particle size of 6–8 mm. The plastic particles were separately extruded in a Leistritz ZSE27iMAXX-400 co-rotating, intermeshing, twin-screw extruder for homogenization purposes. The process temperature was set to 180 °C for PS and 200 °C for ABS. The screw speed for both materials was 80 rpm. The pressure was 53 bar (PS) and 60 bar (ABS). The extruded plastics were directly cut to lenses of 5 mm in diameter using a Hot Face Pelletizer (Leistritz), and air-cooled afterwards.

Table 1
Properties of recycled PS and ABS according to datasheets of WEplast. Mechanical behavior adapted from Chanda and Roy (2007).

Properties	Standard	Unit	WEplast PS	WEplast ABS
Mechanical behavior	–	–	Brittle	Tough, hard, rigid
MFI	DIN EN ISO 1133	g/10 min	7	25
MOE	DIN EN ISO 527-1	GPa	2.2	2.5
Tensile strength	DIN EN ISO 527-1	MPa	31	48
Charpy impact strength un-notched	ISO 179-1 eU	kJ/m^2	20	11
Moisture content	–	%	0.1	1
Specific content	–	%	98	98.5
Impurities	–	–	Wood (0–0.1%) Rubber (0–0.2%) Flame retardants (0–0.1%) PVC (0–0.1%) PS, PPO (0.5–1.5%)	

Table 2
Experimental design.

Sample code		Plastics		Wood		Additive SMA (%)
		PS (%)	ABS (%)	Norway spruce (%)	Particle-board (%)	
A	Ps	100	–	–	–	–
B	Abs	–	100	–	–	–
C	Ps/W30	70	–	30	–	–
D	Ps/W60	40	–	60	–	–
E	Ps/W60/S	39	–	58	–	3
F	Ps/Pb30	70	–	–	30	–
G	Ps/Pb60	40	–	–	60	–
H	Ps/Pb60/S	39	–	–	58	3
I	Abs/W30	–	70	30	–	–
J	Abs/W60	–	40	60	–	–
K	Abs/W60/S	–	39	58	–	3
L	Abs/Pb30	–	70	–	30	–
M	Abs/Pb60	–	40	–	60	–
N	Abs/Pb60/S	–	39	–	58	3

Wood particles were dried at 103 °C for 24 h to reach a moisture content of <1% to be able to interface with polymers and the coupling agent. WPC granulates were extruded in a Leistritz ZSE27iMAXX-40D co-rotating, intermeshing, twin-screw extruder. Table 3 presents processing parameters. The WPC mixture was cut directly into granulates using a Hot Face Pelletizer (Leistritz), and air-cooled afterwards.

After extrusion-compounding, extruded PS and ABS lenses and WPC granulates were injection-molded into mechanical test specimens using an ARBURG Allrounder 420C Golden Edition injection-molding machine. Although the nozzle temperature was in some cases >200 °C, lower mass temperature can be expected as degradation of wood was not observed.

Table 3
Extrusion compounding and injection molding parameters.

Sample		Mass temp. (°C)	Extrusion compounding			Injection molding	
			Mass flow (kg)	Rotation speed (1/min)	Pressure (bar)	Nozzle temp. (°C)	Dosing time (s)
A	Ps	134	3	80	53	215	13.36
B	Abs	171	6	80	60	215	12.88
C	Ps/W30	133	3	80	58	195	16.16
D	Ps/W60	147	4	80	86	206	8.63
E	Ps/W60/S	152	4	80	104	206	9.61
F	Ps/Pb30	144	4	80	51	195	14.83
G	Ps/Pb60	151	4	80	79	205	16.36
H	Ps/Pb60/S	152	5	100	97	205	17.43
I	Abs/W30	160	4	110	50	195	13.39
J	Abs/W60	170	4	110	80	205	18.44
K	Abs/W60/S	162	4	110	77	205	17.13
L	Abs/Pb30	158	3	110	50	195	14.06
M	Abs/Pb60	171	3	110	65	200	24.10
N	Abs/Pb60/S	173	3	110	70	200	25.94

3.2. Composite characterization

Dumbbell-shaped specimens (170 × 10 × 4 mm^3) were used for tensile testing. Rod-shaped specimens (80 × 10 × 4 mm^3) were used for all other tests. For each test, 10 specimens were conditioned and tested in 20 °C/65% relative humidity. Flexural and tensile tests were conducted using a Zwick Roell Universal testing machine. Flexural test was conducted according to DIN EN ISO 178 (DIN EN ISO, 2003). The properties were measured in a three-point bending test at a crosshead speed of 5 mm/min and a load frame with 5 kN load cell. Tensile test was conducted in accordance to DIN EN ISO 527-3 using type 1A specimens (DIN EN ISO, 1996). The properties were measured using a makroextensometer. The crosshead speed was 1 mm/min and a load frame of 5 kN load cell was used.

A water absorption test was conducted according to DIN EN 15534-1 (DIN EN, 2014). The water absorption of the compounds was calculated by differential weighing of the specimens for each time period (at days 0, 1, 2, 4, 7, 14 and 28) after total immersion into demineralized water at a temperature of 20 °C (±2). Water absorption was calculated according to the formula:

$$\text{Water absorption }(\%) = \frac{W_t - W_0}{W_0} * 100 \tag{1}$$

where W_t is the specimen weight after a given immersion in water and W_0 is oven dry mass of the specimen weight after constant mass was reached (weight change < 0.1% after 24 h).

Density of the specimens was calculated according to Archimedes' principle based on the assumption that the water used for immersion has a density of 1 g/cm^3 and was calculated according to the formula:

$$\text{Density }\left(\text{g}/\text{cm}^3\right) = \frac{W_0}{V_0} \tag{2}$$

where W_0 is oven dry mass of the specimen weight and V_0 is the Volume of the specimen. Ten specimens were used for water absorption and density properties.

Scanning electron microscopy (SEM) was conducted to investigate morphology characteristics of WPC specimens and to evaluate the wood and polymer matrix distribution. Specimens for SEM were prepared by cutting a square of 3–5 × 2 mm^2 from tested flexural strength samples. Vapor coating of the samples were done in a BioRAD SC 510 SEM Coating System. The surface microstructure was analyzed in a Quanta FEG Type 250.

According to published work of, i.e., Schlummer et al. (2007), heavy metal content in recycled WEEE was expected. The ICP-OES (Inductively Coupled Plasma Optical Emission Spectrometry) method was used to analyze the elements: silver (Ag), arsenic (As), bismuth (Bi), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb) and thallium (Tl) in recycled ABS, PS and Norway spruce and recycled particleboard. The samples were separately ground and dissolved in Aqua regia with multi-element standard solution IV Certipur to determine the previously mentioned elements. The analysis was conducted with an inductively coupled plasma atomic emission spectroscopy iCAP 6300 duo ICP/OES.

4. Results and discussion

4.1. Physical properties

4.1.1. Density

Densities of specimens are presented in Table 4. The density of recycled ABS was higher (B – 1.054 g/cm^3) than the density of recycled PS (A – 1.043 g/cm^3). ABS product range is heterogeneous due to various possibilities of molecule ratios because ABS consists of three different monomers: acrylonitrile, butadiene, and styrene. For instance, Kuo et al. (2009) reported 1.02 g/cm^3 of virgin ABS. Higher density of ABS up to 1.2 g/cm^3 is also possible, which results in great variability of post-consumer ABS.

Table 4
Density.

Sample		Density		Change in density to plastic matrix	
		MV (g/cm^3)	SD (g/cm^3)	Δ g/cm^3	Δ %
A	Ps	1.043	0.001	–	–
B	Abs	1.054	0.002	–	–
C	Ps/W30	1.136	0.002	0.204	+9
D	Ps/W60	1.247	0.001	0.202	+20
E	Ps/W60/S	1.245	0.002	0.105	+19
F	Ps/Pb30	1.143	0.001	0.214	+10
G	Ps/Pb60	1.242	0.003	0.210	+19
H	Ps/Pb60/S	1.241	0.002	0.110	+19
I	Abs/W30	1.148	0.002	0.007	+9
J	Abs/W60	1.257	0.003	0.106	+19
K	Abs/W60/S	1.253	0.003	0.105	+19
L	Abs/Pb30	1.149	0.002	0.013	+9
M	Abs/Pb60	1.251	0.005	0.115	+19
N	Abs/Pb60/S	1.237	0.009	0.101	+17

The density of wood particles was not investigated in detail. Wood particles were very likely to be further reduced and squeezed during the compounding process. This resulted in wood cell walls which have a density of approximately 1.4 g/cm^3 (Migneault et al., 2014). A detailed description will be presented in Section 4.3. The addition of wood content in the polymer matrix resulted in an almost linear increase in density up to 20% (1.245 g/cm^3) in PS-WPC (E) and 19% (1.251 g/cm^3) in ABS-WPC (M), regardless of whether Norway spruce or recycled particleboard were used. However, specimens made from particleboard exhibited slightly higher densities than specimens made from spruce. This can be linked to possible impurities (small pieces of silica or metals) as German particleboards are produced with <30% recycled wood content.

Density was not significantly affected by incorporation of 3% SMA in all PS-WPC and Norway spruce ABS-WPC.

4.1.2. Water absorption

Fig. 1 presents water absorption. Water absorption was very low for recycled PS (A – 0.2%) and ABS (B – 0.7%) after 28 days. After 28 days, the 30%-wood proportion WPC (C, F, I, L) exhibited about 4% water absorption. In the 60%-wood proportion WPC, Norway spruce (D) resulted in the highest water absorption in the PS-matrix (16%) and in the ABS-matrix (12%) after 28 days.

The increase of water absorption by incorporation of wood particles to the plastic matrix is well known, i.e., Gozdecki et al. (2015). A wood cell wall consists of hydroscopic substances, like carbohydrates and lignin, which leads to water uptake. The higher the wood proportion in the WPC matrix, the more water will be absorbed. Since particleboard contained 8.5% UF, the hydroscopic content is slightly reduced, which resulted in lower water absorption of particleboard-WPC. UF is only soluble in acid and alkaline solutions and boiling water.

The introduction of SMA to the 60%-WPC-matrix was beneficial in both PS and ABS specimens after 28 days, which could be linked to increased interfacial adhesion with wood particles. Except

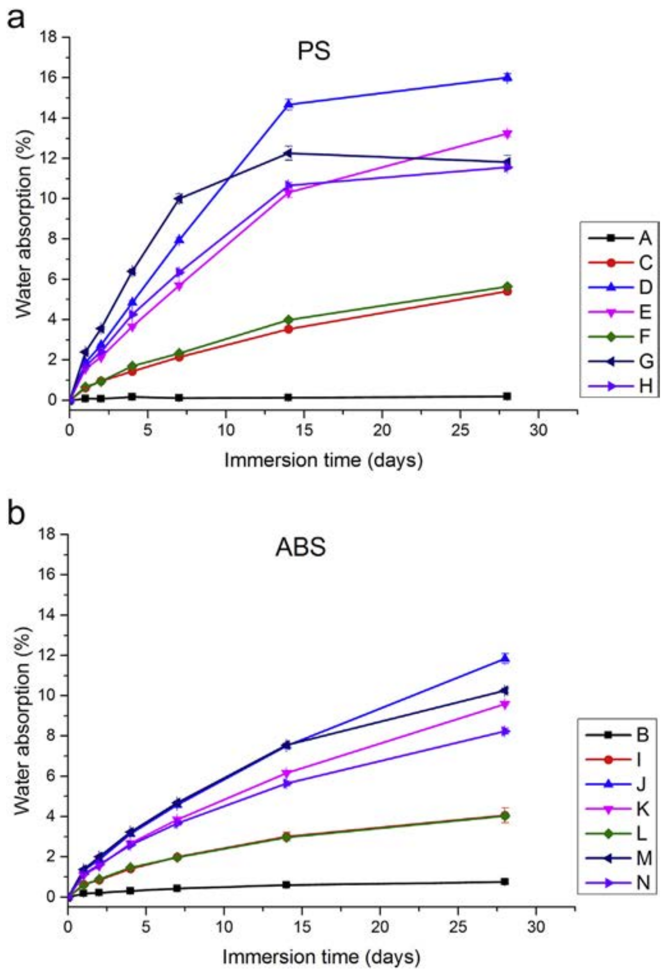

Fig. 1. Water absorption of a) PS-WPC and b) ABS-WPC.

specimens of group G (60% particleboard in PS-matrix) showed very similar water absorption after 28 days. All other WPC specimens projected continuous water absorption beyond 28 days, regardless of whether Norway spruce or recycled particleboards were used.

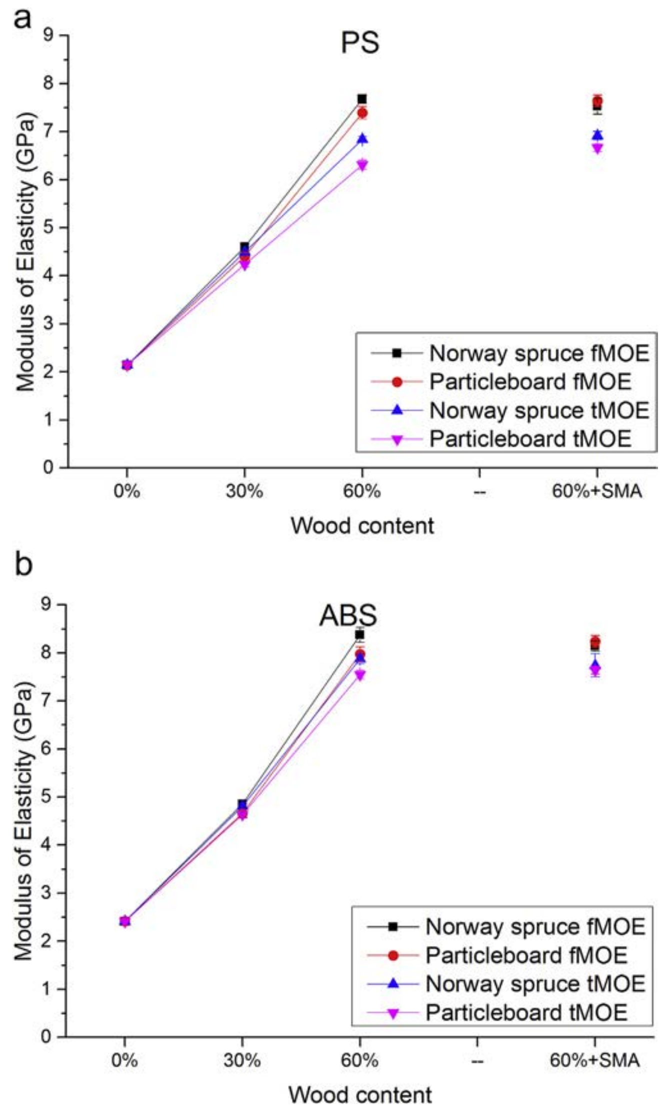

Fig. 2. Modulus of elasticity of a) PS-WPC and b) ABS-WPC.

4.2. Mechanical properties

4.2.1. Modulus of elasticity

Fig. 2 presents the modulus of elasticity derived from tensile (tMOE) and flexural (fMOE) tests. tMOE correlated well with fMOE in each specimen group. The 100% polymer matrix specimens constituted the lowest fMOE (A – 2.1 GPa; B – 2.4 GPa). The incorporation of wood in PS increased the stiffness linearly to 7.7 GPa (D) and 7.3 GPa (G). Since the wood modulus is higher than the plastic modulus, the modulus of WPC increased with increasing wood content. Hence, the incorporation of wood in ABS increased the stiffness linearly as well to 8.3 GPa (J) and 7.9 GPa (M).

Chotirat et al. (2007) achieved lower fMOE (3.1 GPa) and tMOE (1.5 GPa) by using injection-molded virgin ABS and 33.3% sawdust. Comparing fMOE and tMOE properties of PS and ABS-WPC to polyolefins-WPC, i.e., HDPE, PS and ABS-WPC resulted in superior results. Sommerhuber et al. (2015) reported a MOE of 2.7 GPa for flat-pressed WPC made from recycled HDPE and 60% recycled waste wood. The authors stressed that the results might be better if extrusion/injection-molding techniques were to be used. However, the results were comparable with, i.e., Migneault et al. (2014) and Adhikary et al. (2008).

4.2.2. Tensile properties

Fig. 3 presents tensile strength on the left side and elongation at F_{max} on the right side of the plots. Pure PS (A – 12.4% ± 6) showed superior elongation at F_{max} when compared to ABS (B – 4.2% ± 2). With increasing wood content, the elongation was reduced to approx. 0.5% at 60%-wood content regardless of whether Norway spruce or recycled particleboard were used. The introduction of wood significantly reduced the ductility of the polymers as wood has a low ductility which resulted in poor elongation at F_{max} properties.

The introduction of 3% SMA in 60%-wood WPC resulted in values between the results of elongation at F_{max} of 30% and 60%-wood WPC with an average value of 0.7%. In contrast, SMA significantly improved tensile strength properties in both PS-WPC (E – 29.4 MPa; H – 24.8 MPa) and ABS-WPC (K – 48.9 MPa; N – 43.5 MPa), which is in good accordance with the literature. Yeh et al. (2009) stated that the increase in strength with added SMA is due to the improved adhesion between wood and polymer. This

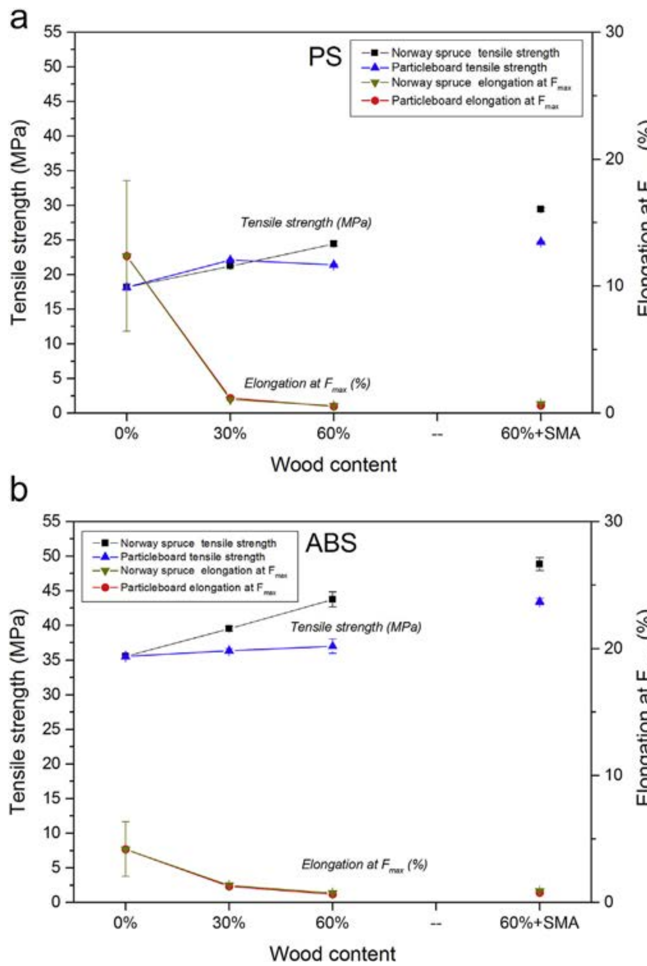

Fig. 3. Tensile strength and elongation at F_{max} of a) PS-WPC and b) ABS-WPC.

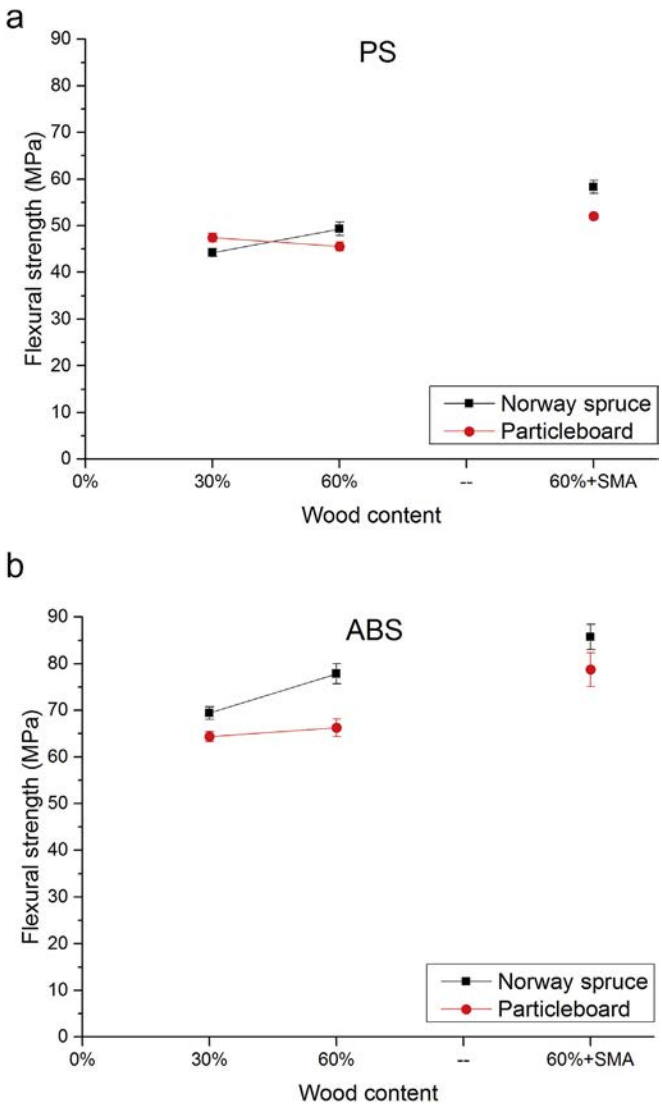

Fig. 4. Flexural strength of a) PS-WPC and b) ABS-WPC.

resulted from the reaction between maleic anhydride in SMA and the hydroxyl groups on the surface of cellulose.

In contrast to elongation at F_{max}, the origin of wood significantly influenced tensile strength. Pure ABS (B – 35.5 MPa) exhibited higher tensile strength than pure PS (A – 18.2 MPa). Incorporation of Norway spruce resulted in superior tensile strength compared to particleboard. The introduction of Norway spruce linearly increased tensile strength in PS to 24.4 MPa (D) and in ABS to 43.8 MPa (J). The introduction of particleboard increased tensile strength in PS to 21.4 MPa (G) and in ABS to 37 MPa (M). Interestingly, a drop in tensile strength through the incorporation of wood was reported by Chotirat et al. (2007), but in our study Norway spruce and particleboard exhibited reinforcement properties. Both wood resources were able to take more tensile load in the matrix than the pure plastics.

4.2.3. Flexural strength

The neat plastic specimens showed ductile performance at 3.5% standard elongation criterion, as no specimens did break. This resulted in flexural stress of 39.8 MPa ± 0.2 (A – Ps) and 65.5 MPa ± 0.3 (B – Abs) at 3.5% elongation.

Fig. 4 presents flexural strength properties of WPC specimens. The increase of wood content in the plastic-matrices increased the flexural strength in both PS-WPC and ABS-WPC. The introduction of 30% wood resulted in the lowest flexural strength at F_{max} in both PS-WPC (C – 44.2 MPa; F – 47.5 MPa) and ABS-WPC (I – 69.4 MPa; L – 64.4 MPa). Flexural strength performance by incorporation of particleboard correlated well with tensile strength as particleboard-PS exhibited higher flexural strength at 30% wood content which decreased by increasing particleboard content to 60% (45.6 MPa). Using Norway spruce resulted in linearly increasing flexural strength to 58.3 MPa (E – Ps/W60) and 85.7 MPa (K – Abs/W60).

The introduction of SMA benefited the flexural strength for both Norway spruce and particleboard-WPC. Therefore, SMA significantly improved tensile and flexural strength properties of recycled PS and ABS-WPC regardless of the wood origin, which can be linked to improvement of interfacial adhesion between wood particles and plastic matrix and is in good agreement with the literature, based on virgin ABS-WPC (Yeh et al., 2009). Therefore, it was likely that the interfacial bonding in the particleboard PS-WPC without SMA was not sufficient as was the case in the particleboard ABS-WPC. SEM pictures are presented in Figs. 5 and 6 and discussed in Section 4.3.

Further, weaker strength results of particleboard-WPC could be linked to the presence of UF (8.5%) which hindered the interfacial adhesion between UF-afflicted particles and plastic matrix. In contrast, Gozdecki et al. (2015) reported better flexural and tensile strength in a comparison study of particleboard-WPC and virgin

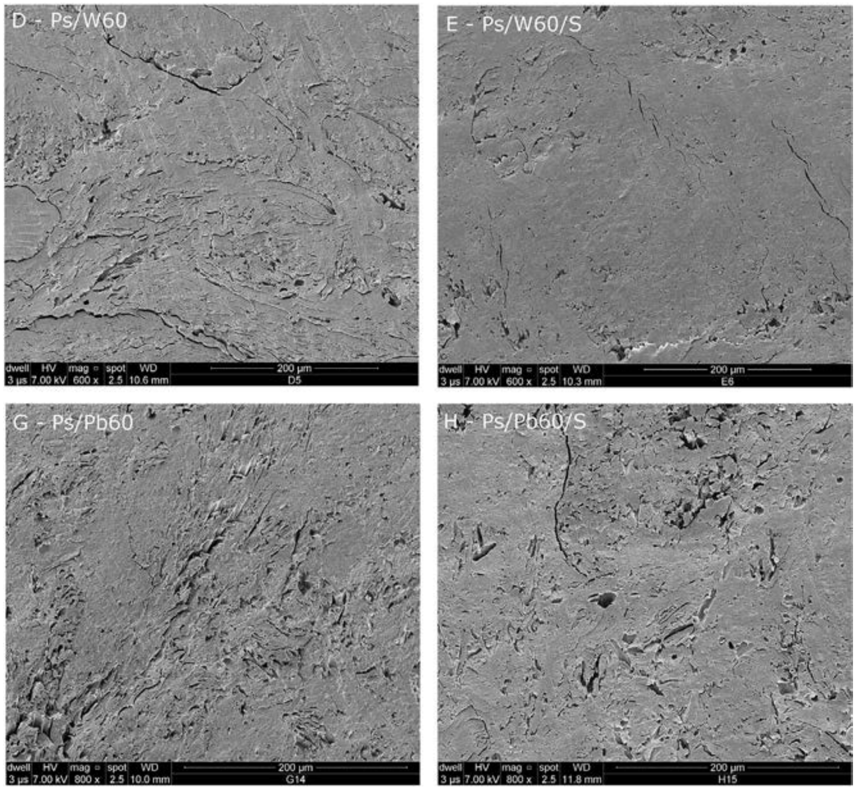

Fig. 5. Scanning electron microscopy (SEM) of PS-WPC.

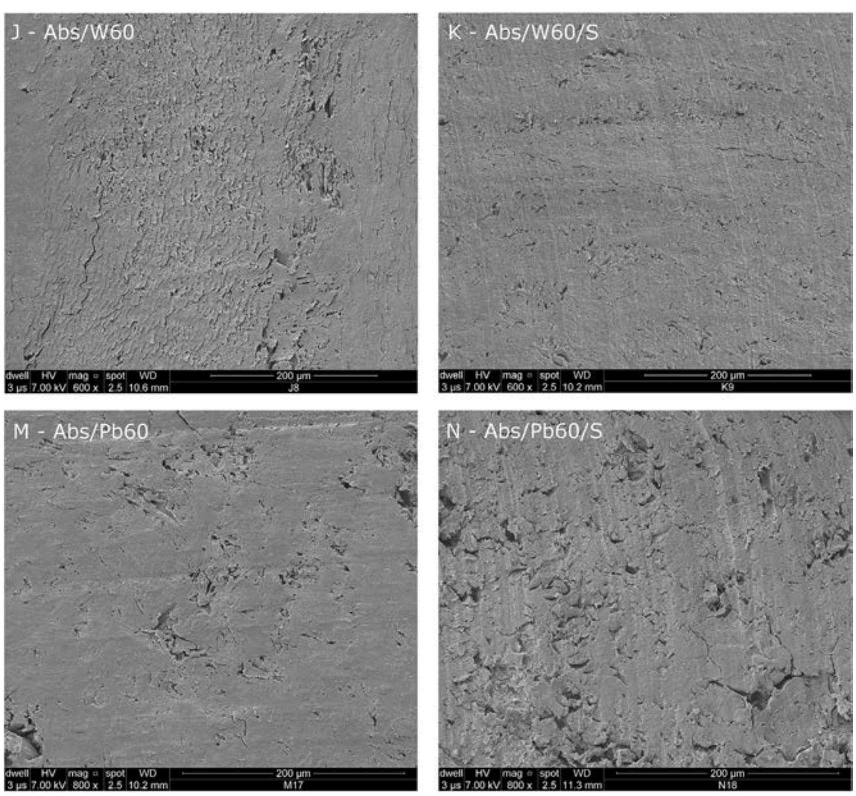

Fig. 6. Scanning electron microscopy (SEM) of ABS-WPC.

wood particles-WPC in a PP matrix. In their study, different geometries of wood (flour, particles) were used where virgin wood particles exhibited much higher length-to-thickness properties than recycled particles from particleboard. The authors linked the presence of UF in particleboard particles to the almost equal mechanical properties of WPC made from virgin wood particles.

4.3. Microstructure analysis

Figs. 5 and 6 present surface cross sections of the Norway spruce reinforced composites on the left side and particleboard reinforced composites on the right side with a 60% wood proportion. The samples exhibited a close contact between polymer matrix and wood particles without SMA in (samples D, G, J, M) with a slightly better interfacial bonding of wood particles and plastic matrix with SMA (E, H, K, N). Microstructural voids were visible in all specimens, whereas the structure of wood lumens seemed to be better maintained in Norway spruce composites (D, E, J, K) than in particleboard composites (G, H, M, N).

Particleboard reinforced PS-WPCs showed weaker physical and mechanical performance compared to all other WPC specimens. ABS is non-polar and polar in structure and therefore relatively hydrophilic in structure, which tends to wet the wood surface (Chotirat et al., 2007; Yeh et al., 2009). This resulted in good interfacial bonding and was likely to lead to better physical and mechanical performance compared to PS composites, regardless of the origin of the wood content. According to the datasheets of WEplast, ABS showed a higher MFI of 25 compared to PS of 7, which influenced rheological behavior of the WPC matrix in extrusion and injection molding processes.

4.4. Elementary analysis

It was assumed that recycled ABS and PS from WEEE contained residues of hazardous substances from, i.e., flame-retardants and heavy metals among other substances, according to the works of, i.e., Schlummer et al. (2007), Oguchi et al. (2013) and Dimitrakakis et al. (2009). Results in Table 5 reflect the pure PS, ABS, Norway spruce and particleboard without compounding to WPC.

High concentrations of Cd were found in the recycled ABS, which can be linked to Cd-containing color pigments (Fink et al., 2000; Schlummer et al., 2007). The higher contents of Cr (+3.35 ppm), Cu (+2.3 ppm) and Pb (+9.8 ppm) in recycled particleboard particles compared to Norway spruce were likely to be derived from recycled content in particleboard. As was only found in the recycled resources. Water-borne preservatives based on copper, including in part chrome or arsenic, was commonly used as fungicide for outdoor applications of wood products. Today, the use in Europe is strongly regulated (European Commission, 2012b). Some preservatives, such as copper chrome arsenic (CCA), are not applied anymore in Germany and Europe. Pb is likely to be derived from lead tetra oxide rustproof primer pigments for, i.e., nails in solid wood products, but its use has been restricted since 2005 in Germany.

Table 5
Elementary analysis. LOD – limit of detection. Values in ppm.

Element	rPS	rABS	Norway spruce	Particleboard
Ag	<LOD	<LOD	0.03	<LOD
As	0.20	0.30	<LOD	0.21
Bi	<LOD	<LOD	<LOD	<LOD
Cd	10.65	58.99	0.30	0.26
Co	6.88	10.84	0.07	0.26
Cr	4.53	7.05	0.21	3.57
Cu	11.69	19.07	0.93	3.24
Ni	5.74	10.92	0.24	0.95
Pb	8.45	4.13	0.75	10.54
Ti	<LOD	<LOD	<LOD	<LOD

In Germany, recycling of waste wood as secondary material or fuel is regulated by the German Waste Wood Directive (German Government, 2003). Waste wood is categorized into four categories (A I, A II, A III and A IV) as can be seen in Table 6. Only waste wood categories A I and A II are permitted to be recycled in particleboards. Biocide-treated wood is classified as A IV and therefore not suitable as a secondary material in wood products. However, Riedel et al. (2014) reported increased Cr and Pb content in waste wood categories, in which no hazardous impurities should be expected (A I and A II). Diffusion of hazardous substances through recycling activities is prohibited by the German Waste Management and Product Recycling Act (German Government, 2012) and the European Waste Framework Directive 2008/98/EC (European Commission, 2008).

Therefore, it can be assumed that the separation processes of recycled wood particles for particleboard production were not sufficient to separate the waste wood into a non-contaminated fraction for material recovery (A I and A II) and a contaminated energy recycling fraction (A III and A IV). The multiple accounting of EWC categories (European Waste Catalogue) to different waste categories of A I, A II and A III seems to be a possible explanation for the diffusion of contamination in waste wood categories.

4.4.1. Application considerations

The level of elements is very likely to change over time as the composition of products and the regulation of substances will change over time. Schlummer et al. (2007) compared the results of an elementary analysis of recycled plastics from WEEE to previously released studies, which had been published more than eight years prior to their publication. The authors reported lower levels of Ca, Pb, Cr, and Ni in housing shredder material by factors 2–7 and lower levels of Sb, As, Cr, Ni in residues of shredder material by factors 4–30. This can be linked to changing standards and optimization of separation technologies. However, the authors concluded that the main drawback of recycling WEEE polymer fractions is the presence of PBDD/F (polybrominated dibenzodioxins and dibenzofurans), PBDE (diphenyl ethers) and Cd at levels close to or above legislative thresholds. PBDE can be linked to flame-retardants, which were not specifically investigated in this study, but can be expected according to the datasheets of WEplast PS and ABS (0–1% proportion of flame-retardants in recycled PS and ABS).

In respect to potential applications of WPC from these secondary resources, threshold values of hazardous substances need to be considered according to DIN EN 71-3 (DIN EN, 2002). The standard is used for products made of particleboard, which are used in domestic applications and in close contact to humans. The standard can be analogously used for WPC products in comparable applications, i.e., outdoor deckings. In the case of closed-loop recycling – recycled plastics are used in the same product category – threshold values of the RoHS Directive (European Commission, 2011a) need to be considered.

In both cases, threshold values were not exceeded. However, the recycled samples were not contamination-free, which could be minimized by improvements in sorting techniques and an accurate allocation of waste wood into waste wood categories.

In terms of steady secondary resource availability (*quantity*), recycled ABS and PS from WEEE as well as post-consumer particleboards can be considered as sufficient to substitute primary materials in WPC production. However, the great heterogeneity of monomer ratios in ABS needs to be carefully discussed (*quality*), which impacts the thermal degradability of wood. At temperatures

Table 6
Methods for waste wood recycling.

Waste wood category	EWC	Examples	Permissible recovery option according to German Government (2003)		Possible contaminations
			Secondary material	Secondary fuel	
A I	03 01 05 15 01 03 17 02 01 20 01 38	Natural wood from shavings and packaging, cable drums after 1989	✓	✓	Metals, lacquer, primer, plastics
A II	03 01 05 17 02 01 20 01 38	Furniture, veneer residues without PVC, derived timber products without harmful contaminations, particleboards	✓	✓	Metals, pigments, concrete, sand, PVC veneered particleboards
A III	17 02 01 20 01 38 20 03 07	Furniture containing halogenated organic compounds	(✓)	✓	Metals, PVC etc.
A IV	15 11 0* 17 02 04* 17 06 03* 19 12 06*	Waste wood with treated wood preservatives (i.e., railway sleepers, telephone masts)	–	✓	

between 100 °C and 200 °C, heating of wood produces emissions of water vapor, carbon dioxide and traces of organic compounds during processing. Above 200 °C, significant thermal degradation occurs, with major mass loss beyond 250 °C (Borrega and Kärenlampi, 2008). For ABS, the processing temperature of extrusion and injection molding machinery may vary between 180 and 260 °C, depending on the specific molecule ratio of the polymer. The processing temperature for PS may vary between 160 and 230 °C. In our study, it was possible to keep the extrusion-compounding temperature below the degradation point of wood to produce WPC.

Results of elementary analysis were limited in terms of sampling and sample size, as only a very small amount (50 kg) of the annually recovered WEEE was used in our study at a specific time. Therefore, continuous sampling should be applied over a long period, if recovered plastics from WEEE and particleboard are considered for WPC applications. However, the first results showed satisfying results using plastics from properly collected WEEE and its recycled resources.

5. Conclusions

WPC specimens made of recycled WEEE-plastics (PS and ABS) were successfully produced with virgin Norway spruce and recycled particleboard. Results of *physical* properties showed:

- increased density (9–19%) and water absorption after 28 days (4–12% in PS; 4–16% in ABS) of WPC by adding 30% and 60% wood content respectively to the recycled plastic matrices
- at 60% wood proportion, incorporation of recycled particleboard exhibited slightly lower (−1%) water absorption after 28 days than Norway spruce
- physical properties of 60% wood-content WPC benefited from coupling agent (SMA)

Results of *mechanical* properties of WPC exhibited:

- linear increase of tensile and flexural MOE by increasing wood content
- tensile and flexural strength benefited by wood proportion resulting in high strength at high wood content
- ABS with particleboard achieved satisfying results without SMA
- particleboard reinforced PS-WPC benefited from SMA due to better interfacial bonding

Elementary analysis was conducted to investigate the content of heavy metals in the recycled resources:

- recycled ABS exhibited high Cd content (59 ppm) which can be linked to color pigments in WEEE-plastics
- particleboard showed higher contents of Cr (+3.35 ppm), Cu (+2.3 ppm) and Pb (9.8 ppm) compared to Norway spruce which is very likely to be derived from previous recycling processes of particleboards production,
- The heavy metal element As was only found in the recycled resources

Finally the question has to be asked if it is technically feasible to produce WPC from secondary resources is it feasible in terms of environmental considerations as well? Material complexity of products tends to be getting higher, which results in the need for more sophisticated waste management systems, policies and treatment technologies. Using secondary resources as input material from open-loop recycling to substitute primary material may on the one hand be environmentally preferable, but producing materials where only closed-loop recycling is feasible with given state-of-the-art technologies or diffusion of hazardous substances is likely to occur, does not support the precautionary principle of sustainability. Therefore, future research is needed to investigate environmental trade-offs of using secondary resources for WPC including the end-of-life of the composites.

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5.3 Peer-reviewed paper III

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Full length article

Life cycle assessment of wood-plastic composites: Analysing alternative materials and identifying an environmental sound end-of-life option

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ABSTRACT

In the drive towards a sustainable Bioeconomy, a growing interest in the development of composite materials made of plastics compounded with wood particles, known as wood-plastic composites (WPC), can be observed. Wood is seen as one of the cornerstones for sustainable economic growth, while the use of thermoplastics from hydrocarbon fossil resources and additives for WPC potentially cause severe environmental impacts along the entire life cycle. In this study, the life cycle stages of raw material supply and end-of-life pathways of WPC were assessed environmentally from different perspectives with life cycle assessment (LCA). The utilization of alternative raw materials reflected the WPC producer's point of view. Harmonized product LCA standards were applied and combined with physical parameters of actually produced composites to give credit to substitution potentials in terms of resource quality. The downstream pathways of post-consumer WPC products reflected the recycler's perspective. A system LCA approach was needed where systems with equal functions were generated to secure a comparison of end-of-life (EoL) treatment systems. Results showed that WPC produced from secondary materials is the ecologically and technically superior alternative. Recycling of the composites would be the ecologically preferable pathway, but the recycled WPC content in novel WPC is a sensitive issue when comparing both EoL treatment systems. Yet, incineration of the composites is the predominant EoL pathway due to current recycling directives and lack of markets for secondary WPC material.

1. Introduction

The European Bioeconomy Strategy highlights the use of wood as one of the cornerstones for a sustainable economic growth (EC, 2012). The manifold alternatives for using wood led to a competition between the various utilization pathways as a renewable energy carrier, as a renewable precursor for the chemical industry and in the biofuel industry in context of promoting Sustainable Development (UNCED, 1992; UNFCCC, 1998; UN, 1998). This raised concerns of wood availability, especially due to direct and indirect incentives for wood used as an energy carrier (Geldermann et al., 2016; Höglmeier et al., 2014). Mantau et al. (2010) concluded that there will be not enough wood from sustainably managed forests for the competitive markets of material use and fuelwood in 2030.

A potential solution to overcome resource scarcity is seen in the cascading use of biomass and in using by-products more efficiently in the wood-based products sector. Sirkin and Houten (1994) stated that "*resource cascading has been utilized as a method for achieving resource conservation in contexts where resources have been regarded as precious or vital*". The cascading principle is to some extent manifested in national acts of the European member states with the execution of the European Waste Framework Directive 2008/98/EC (EC, 2008). This directive prioritizes the end-of-life (EoL) alternatives of waste by demanding the so-called waste management hierarchy: (1) prevention, (2) preparing for re-use, (3) recycling (without any kind of incineration), (4) other recovery (i.e., energy recovery) and (5) disposal. In addition to cascading, various strategies and concepts, such as the Circular Economy on European level (EC, 2011; EP, 2015; Haas et al., 2015) and ProgRess on German level (BMUB, 2015), have been developed to strengthen recycling and circulating materials aiming at an efficient utilization of resources.

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1.1. Wood-plastic composites

In pushing towards a Bioeconomy, a growing demand can be observed in composite materials made of plastics compounded with a significant share of wood particles, forming the so-called wood-plastic composites (WPC) (Partanen and Carus, 2016). The modifiable affinity of the molecular structure of the polymer matrix of thermoplastics in combination with various kinds of fillers, additives and reinforcement materials results in a functional improvement of the neat polymer matrix (Klyosov, 2007). WPC is defined "*as a material or product made thereof being the result of the combination of one or several cellulose-based material(s) with one or several thermoplastics, intended to be or being processed through plastic processing techniques*" (CEN, 2014a).

According to Carus et al. (2015), about 260 kt of WPC were produced in the year 2012 in Europe. The European WPC production represents approximately 9% of the globally produced WPC. The outdoor deckings market is the biggest distribution channel for products made of WPC. In Germany, about 80% of the distributed WPC deckings are in accordance with the "Quality and test specifications for production control of deckings" of Qualitätsgemeinschaft Holzwerkstoffe e.V. (Qualitätsgemeinschaft Holzwerkstoffe e.V., 2016). According to this voluntary commitment, WPC should be derived from FSC-certified wood from sustainably managed forests or waste wood categorized as untreated, natural wood – class A I (German Government, 2003) – with more than 50% wood content. The plastics shall be derived from primary or post-industrial production (Qualitätsgemeinschaft Holzwerkstoffe e.V., 2016).

1.2. Wood-plastic composites and the environment

WPC products replace either solid wood products (i.e., outdoor deckings and wall claddings in the building and construction sector) or neat plastic products (i.e., decorative claddings in the automotive sector). Geldermann et al. (2016) stated that WPC cannot compete with solid wood with regard to the low environmental impact of wood, but is an environmentally friendly alternative to neat plastics. Teuber et al. (2016) stated that WPC tend to be a possible step in a cascading use of biomass before energy recovery of the biomass.

The environmental assessment of products and services is conducted based on quantitative data by the life cycle assessment (LCA) methodology, which is standardized in ISO 14040 (ISO, 2006a) and ISO 14044 (ISO, 2006b). The technical feasibility of producing WPC from secondary or cascaded resources has been intensively analysed since the early 1990s, for instance by Yam et al. (1990), Youngquist et al. (1992), Adhikary et al. (2008), Ashori (2008), Sommerhuber et al. (2015) and Sommerhuber et al. (2016). To the best of the authors' knowledge, an ecological analysis based on quantitative data is missing in all of these studies. Studies exist reporting on WPC produced with virgin raw materials on a laboratory scale for which an LCA was conducted ex ante (Hesser, 2015; Mahalle et al., 2014; Qiang et al., 2012, 2014; Xu et al., 2008), with the focus on different alternative materials (Väntsi and Kärki, 2015) or focusing on integrating durability issues of WPC into LCA (Miller et al., 2015).

The following studies used the LCA methodology to compare WPC deckings to solid wood deckings. Bolin and Smith (2011) and Bergman et al. (2013) conducted an LCA of solid WPC outdoor deckings with PE and HDPE as the plastics matrix with wood particles from co-products. Both authors considered also recycled plastics and concluded that WPC from recycled plastics is environmentally better than WPC from virgin plastics, but inferior to solid wood deckings in the North-American context. However, a consideration of technical functions and the application context is missing. Stübs et al. (2012) conducted an LCA of solid and hollow WPC outdoor deckings, comparing the results to solid wood deckings from tropical Bilinga (*Nauclea diderrichii*) and Scots pine (*Pinus sylvestris*). The authors concluded that WPC had better environmental performance due to a prospective higher service life and less maintenance than the solid wood deckings. In addition, Environmental Product Declarations (EPD) were published for WPC products in a German context, stating a reference service life of 30 years for deckings and 40 years for claddings according to manufacturers' specifications (IBU, 2015a,b). A combination of the functional unit and service life should be seen critically, not only because WPC is facing durability challenges under outdoor conditions, which depends on macro and micro-climate conditions as well as on the application context (Catto et al., 2016; Ibach et al., 2013). Methods for potential durability improvements can be linked to, for instance, changing processing conditions or adding additives (Stark and Gardner, 2008). The possibility of using additives in WPC is one reason why these composites are often promoted to be comparable or even better than solid wood deckings. However, the additives in synthesized products and plastics are problematic for human health and the environment along the entire life cycle (Thompson et al., 2009).

Considering the EoL of WPC, the separation of wood particles and thermoplastic polymers from the composite is technically challenging. During the compounding process of WPC, the thermoplastic matrix is heated to the crystalline melting point (T_m). Then, wood particles are added to the melted thermoplastic and mechanically irreversibly bonded to the plastic-matrix. The WPC matrix is cooled until the thermoplastic molecules solidify, which is known as the glass transition temperature (T_g) (Klyosov, 2007). Recycling of post-consumer thermoplastic to secondary materials is possible in comparison to thermosets or elastomers, but recycling of the constituents of WPC to secondary materials for cascading and improvement of resource efficiency is currently economically unfeasible (Meinlschmidt et al., 2014; Sommerhuber et al., 2015).

Among the LCA studies of WPC, Thamae and Baillie (2008) investigated the replacement of glass fibre reinforced polypropylene (PP) car door panel with a wood fibre reinforced PP panel. The EoL was modelled using the avoided burden approach, which allocates the potential credits of energy generation of waste incineration to the manufacturing of a product. Allocation of EoL processes is of major importance for the LCA results and is discussed critically within the LCA community (Bergman et al., 2014a; Heijungs and Guinée, 2007; Nicholson et al., 2009; Sandin et al., 2014). In the context of applying the waste management hierarchy (EC, 2008) on post-consumer WPC, an ecological comparison is needed. The study of Väntsi and Kärki (2015) is an example of assessing the environmental profile of alternative materials for WPC (recycled mineral wool and polypropylene) and its end-of-life treatment options (incineration with energy recovery versus landfill).

1.3. Aim and objectives of the study

The upstream processes (resource alternatives) as well as the downstream processes of WPC need to be thoroughly ecologically analysed. With the underlying assumption that it is technically feasible to produce WPC completely from secondary materials (recycled wood and plastics), we ask the first research question (*RQ 1*):

(1) What is the ecological difference of WPC made from virgin vs. secondary materials?

The focus is on the ex-ante LCA analysis of WPC compounds, as manufactured and analysed prior in Sommerhuber et al. (2015) and described in Section Product LCA. The thermoplastic matrix is HDPE compounded essentially with softwood particles (Table 1).

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Table 1
Mass-related resource ratios and mechanical properties of WPC specimens from Sommerhuber et al. (2015).

Materials	$Y_{30\%wood}$		$Z_{60\%wood}$	
	Y_V	Y_R	Z_V	Z_R
Waste wood (category A II)	–	29%	–	58%
Recycled HDPE	–	68%	–	39%
Norway spruce	29%	–	58%	–
Virgin HDPE	68%	–	39%	–
MAPE	3%	3%	3%	3%
Physical properties				
Density [g/cm^3]	1.05	1.05	1.17	1.16
Stiffness				
Flexural MoE [GPa]	1.55	1.58	2.77	2.62
Tensile MoE [GPa]	1.74	1.65	2.87	2.74
Strength				
Flexural strength [MPa]	33.6	31.4	42.3	31.6
Tensile strength [MPa]	18.5	15.7	21.6	16.0

MoE-Modulus of Elasticity.
V-virgin material.
R-secondary material.

In context of a growing WPC market, where the European recycling system will be facing a growing material complexity caused by post-consumer WPC in the near future, we ask the second research question (*RQ 2*):

(2) What is the environmentally preferable EoL option of the composites: utilization as secondary fuel or as secondary material?

In addition, a qualitative discussion highlights the challenges and problems generated by producing and disposing of WPC with current technologies and political frameworks.

2. Method

The research questions consider the environmental impacts of the upstream process of WPC from cradle-to-gate (*RQ 1*) and the downstream pathways of the EoL of WPC (*RQ 2*). They can be further defined as the *processor's perspective* (*RQ 1*), where the decision is to use virgin or secondary materials for the production of WPC. From this perspective, the function of the compared systems is the provision of an amount of material with the same technical quality. More detail on the declared unit is presented in Section Declared unit.

On the other hand, *RQ 2* aims at the *recycler's perspective*. It assesses the potential environmental impacts of different waste treatment technologies of post-consumer WPC products to address the cascading approach and the waste management hierarchy as stated above. In this case, the functions of the systems to be compared are not identical. Therefore, comparable systems have to be formed, which is done by applying system expansion in this article.

Fig. 1 shows the difference between the two research questions, pointing at different perspectives. The approaches can be named "product LCA" for the processorís perspective and "system LCA" for the recyclerís perspective.

2.1. Product LCA

The LCA method as regulated by ISO 14040 (ISO, 2006a) and ISO 14044 (ISO, 2006b) is further specified for building products in EN 15804 (CEN, 2013), and beyond that, especially for wooden building products in EN 16485 (CEN, 2014b). EN 15804 (CEN, 2013) and EN 16485 (CEN, 2014b) focus on the LCA of a specific product with its respective life cycle phases. The single processes to provide the assessed product are further summarized to so-called modules. Benefits and loads from recycling processes are assigned to a module beyond the system boundary to separate recycling potentials from the original life cycle of the product.

2.1.1. System boundary

The product LCA considers a cradle-to-gate approach of a WPC compound as a semi-finished product. Fig. 2 illustrates the system boundary. The WPC compounds were manufactured on laboratory scale based on the work of Sommerhuber et al. (2015). The compounds were produced from high-density polyethylene (HDPE) granulates, wood particles and maleic anhydride polyethylene (MAPE) as the coupling agent. The materials used were derived either from virgin or secondary materials. The manufacturing process is described in the Supplementary information.

2.1.1.1. Raw material supply (A1). In this article, the focus is on WPC compounds derived from either 100% virgin wood and HDPE or 100% secondary wood and HDPE.

2.1.1.1.1. Virgin materials. The upstream processes of the production of virgin wood particles from Norway spruce (*Picea abies*) started with the biological production of soft wood logs. The logs had 55% moisture content (mc) and were further processed to wood particles of approximately 10%-mc and <1 mm particle size. The wood particles were the only timber product. It was assumed that no co-products were produced and therefore no allocation was needed. The LCI was analogously based on literature (Rüter and Diederichs, 2012; Schweinle, 1996; Stübs et al., 2012; Wegener et al., 2004). More details on LCI are provided in the Supplementary information.

The upstream processes of the production of virgin HDPE started with the extraction of crude oil. Polyethylene is polymerized from ethylene, which is extracted by cracking naphtha or natural gas in a steam-cracker. The coupling agent MAPE is produced using 95% LLDPE and 5% maleic anhydride. Plastics data were taken from aggregated LCI datasets from LCA databases (ecoinvent Centre,

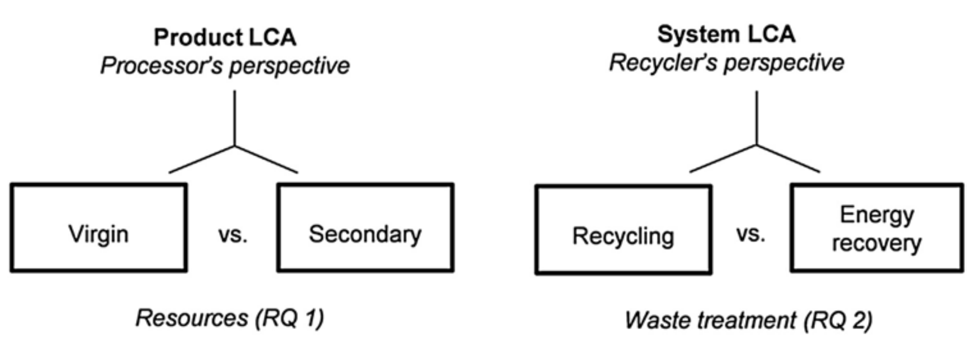

Fig. 1. Product LCA and system LCA in context of research questions.

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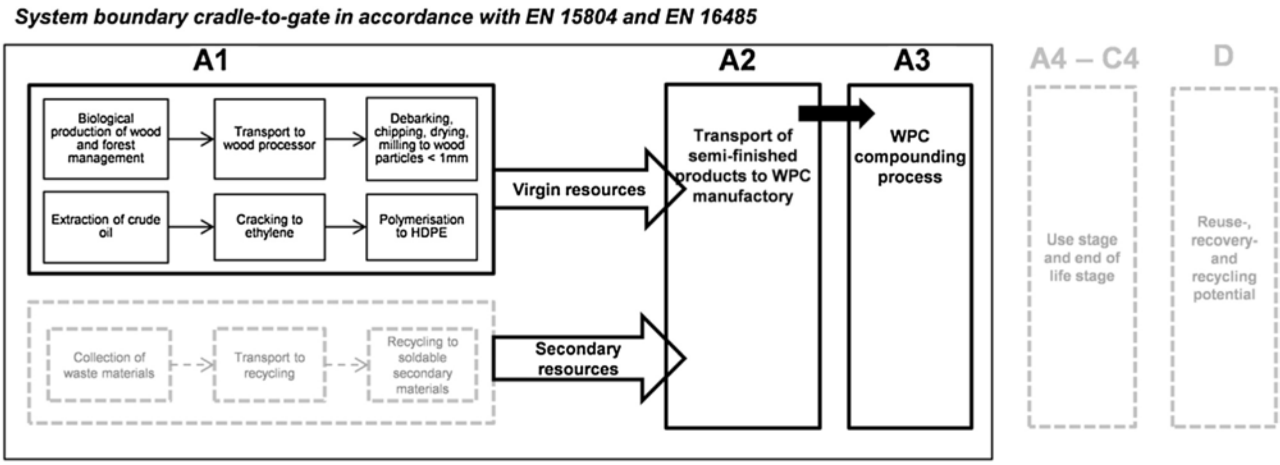

Fig. 2. System boundary of product LCA. Dotted lines and boxes mean cut-off.

2010; thinkstep, 2015). More details are provided in the Supplementary information.

2.1.1.1.2. Secondary materials from post-consumer packaging waste. Using secondary materials from post-consumer waste means, according to EN 15804 (CEN, 2013), that these materials enter the product system after reaching the end-of-waste status in the system in which they were previously used. Derived from this definition, secondary material is not to be called waste because it had definitely reached the end-of-waste status.

EN 16485 (CEN, 2014b) clearly defines the end-of-waste status for wood and wood-based construction products. The end-of-waste status is reached "*after sorting and chipping/crushing of any post-consumer wood excluding pressure impregnated wood, considering that a positive market value can be expected at this processing stage, a specific purpose is defined (use as fuel, use in particle board production), the chipped wood fulfils the technical specifications of solid fuels/chips for the production of particle boards and no hazardous substances exceeding legal limits can be expected*" (CEN, 2014b).

In the case of WPC production, this means that the recycled wood enters the product system in the shape of post-consumer wood chips including its material inherent properties of A1: wood inherent biogenic carbon and embodied energy. The recycled HDPE enters the product system in the shape of granulate and because of this, only the embodied energy from fossil carbon is accounted for. The LCI of secondary wood particles is provided in the Supplementary information.

2.1.1.2. Transport (A2). The raw materials were transported with a conventional truck, EURO 3 emission standard, a capacity load of 17.3 t and 85% utilization rate for an average distance of 100 km. Studies of previous LCA concluded that longer distances, i.e., 600 km (Stübs et al., 2012), had no significant influence on the life cycle impact assessment (LCIA) results of WPC.

2.1.1.3. Manufacturing (A3). This module comprised the compounding of raw materials to WPC compounds as a semi-finished product. According to the EN 15804 (CEN, 2013), emissions to air, soil and water occurring by manufacturing and auxiliary materials are categorized to this module. The provision of electricity ($energy_{el}$) and heat ($energy_{th}$) used for manufacturing was also categorized to this module.

The efforts of secondary wood particle processing were analogously used from virgin wood particles (crushing, milling, drying; emissions to air) with the exception of moisture content (33%-mc). A sorting loss of 20% was assumed. The compounding process needed 4 MJ/kg $energy_{el}$ (Stübs et al., 2012).

2.1.2. Declared unit

The functional unit provides the physical reference to which the inputs and outputs of the LCA are related. It further reflects the purpose of the product and refers to the goal and scope settings (ISO, 2006a). If the specific purpose of the product is not clear, i.e., in the building context, the functional unit is changed to the *declared unit* according to EN 15804 (CEN, 2013). Hence, in this paper the functional unit is considered as the declared unit because the products under study are modelled until they reach a semi-finished product status: 1 kg WPC compound with given mechanical properties as presented in Table 1.

2.1.3. Allocation

In the case of virgin wood particles, forest management was only dedicated to timber production (Rüter and Diederichs, 2012). Timber was debarked at the wood processor. Debarking residues were cut-off. Following EN 15804 (CEN, 2013), environmental burdens of waste treatment of residues originated in a process step – which are not defined as co-products – have to be accounted to the product system under study until the end-of-waste criteria can be applied. The potential secondary material or the energy generated from the recycling step is excluded from the system boundary. The potential credits and burdens related to the use of these secondary materials or fuels have to be accounted for in the separate, scenario-based module D. This module was not considered in the system boundary of this study. EN 16485 (CEN, 2014b) provides guidance for the allocation procedures of reuse, recycling and energy recovery of wood-based products depending on whether the waste flow reaches the end-of-waste status or not. In the WPC case utilizing secondary material, the waste flow definitely reached the end-of-waste status in the previous product system. This is why secondary material was considered only with its material inherent properties.

Allocation along the upstream processes of plastics, $energy_{el}$ as well as other resources and materials was provided in aggregated datasets of the GaBi professional database (thinkstep, 2015) and the ecoinvent database (ecoinvent Centre, 2010). Further information is presented in the Supplementary information.

2.1.4. Elementary analysis

Heavy metal contents in materials were analysed because they were expected (Dimitrakakis et al., 2009; Oguchi et al., 2013; Schlummer et al., 2007; Sommerhuber et al., 2016). The ICP-OES (Inductively Coupled Plasma Optical Emission Spectrometry) method (Moore, 1989) was used to analyse the elements silver (Ag), arsenic (As), bismuth (Bi), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb) and thallium (Tl) in recycled

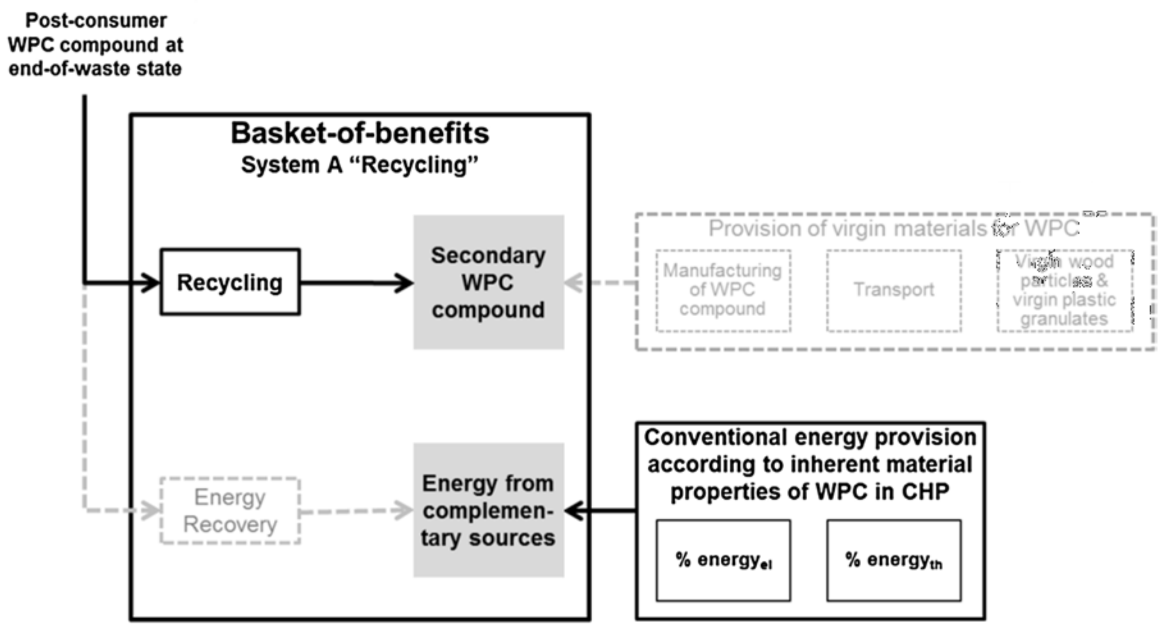

Fig. 3. System boundary of end-of-life system A—recycling. Dotted lines and boxes mean cut-off.

HDPE and waste wood, which were used in the WPC compound and presented in Table 1. The recycled post-consumer HDPE from packaging materials is called recythen® HDPE, provided by INTERSEROH Dienstleistungs GmbH, Germany. The waste wood was provided by Buhck Umweltservices GmbH & Co. KG, Germany. It can be classified as waste wood of category A II according to the German Waste Wood Directive (German Government, 2003), containing predominantly post-consumer transport pallets and some post-consumer harvested timber products with a high softwood content.

The samples were separately ground and dissolved in Aqua regia with multi-element standard solution, called Certipur® ICP multi-element standard solution IV, provided by Merck KGaA, Germany, to determine the previously mentioned elements. The analysis was conducted with an inductively coupled plasma atomic emission spectroscopy, called Thermo Scientific iCAP 6300 duo ICP/OES.

2.2. System LCA

Although EN 15804 (CEN, 2013) and EN 16485 (CEN, 2014b) give guidance on allocation at the end-of-life of a single product, which was considered in the EPDs for WPC deckings and WPC claddings at IBU (2015a, 2015b), they are not applicable to compare the environmental impacts of different waste treatment options. Therefore, a system LCA is needed where systems with equal functions are generated to secure that the systems can be compared.

When comparing two systems using the LCA method it is crucial that both systems provide the same function (Fleischer and Schmidt, 1996), and can be described by the same functional unit. If this is not initially ensured, one has to equalize the systems by, for instance, expanding the product system, which does not provide the additional function. The additional function is added to this system by adding the single supply chain of the missing function, the so called *complementary function* (Fleischer and Schmidt, 1996). As a result, the same function, and respectively the same functional unit, is provided by both systems.

The system expansion to ensure equality of functions is widely described and applied in literature, i.e., Fleischer and Schmidt (1996), Jungbluth and Frischknecht (2006), Heijungs and Guinée (2007), Höglmeier et al. (2014), Wäger and Hischier (2015). This approach is also referred as *basket-of-benefits* (Jungbluth and Frischknecht, 2006; Klöpffer and Grahl, 2012). Especially for the assessment of waste treatment at the EoL where material or energy recovery leads to completely different products and functions, the basket-of-benefits approach is a feasible method to perform an LCA to compare different EoL-options.

2.2.1. System boundary

In this study, we assumed that a market has been established for secondary WPC granulates. Therefore, a demand exists for post-consumer WPC products collected in context of recycling as a valuable secondary material for novel WPC. A more detailed discussion concerning the current political framework and recycling of WPC is given in Section 3.2.

Fig. 3 illustrates the *recycling* pathway system A. Recycling of WPC to secondary material is based on literature data from HDPE recycling (Vidal et al., 2009), because recycling of WPC is more related to plastics recycling than wood recycling. The complementary energy$_{el}$ and energy$_{th}$ need to be provided in system A in the context of functionality and comparability to the *energy recovery* pathway (system B). Production of virgin WPC compound is cut-off because it is substituted by secondary WPC compound.

In system B (Fig. 4), energy is recovered with respect to the net calorific value (NCV) of WPC in the context of waste-to-energy conversion. Because post-consumer WPC is incinerated and no longer available for recycling, virgin WPC needs to be produced and added to system B, to provide the same function as and ensure the comparability to system A.

The collection of waste and the transport to the recycling sites were cut-off because these efforts are balanced equally for both systems. Landfill of untreated organic products has been restricted (EC, 2008) and was therefore not considered in this study.

2.2.2. Functional unit

The objective of an LCA for waste treatment is to assess the potential environmental impacts of different waste management strategies. Therefore, the functional unit is to manage 1 t of post-consumer WPC for both systems A and B.

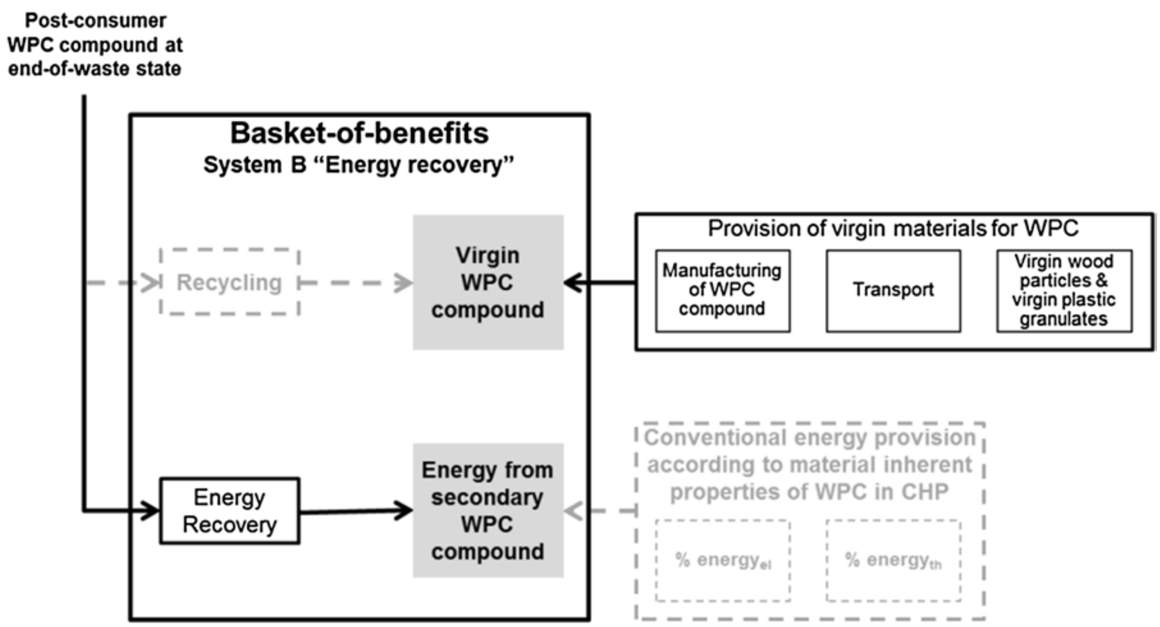

Fig. 4. System boundary of end-of-life system B—energy recovery. Dotted lines and boxes mean cut-off.

Table 2
Scenarios for system LCA.

Scenario	System A—Recycling		System B—Energy recovery	
	energy$_{el}$ (73%)	energy$_{th}$ (27%)	wood$_{Virgin}$ (%)	plastic$_{Virgin}$ (%)
Basic	Grid mix	Natural gas	49	51
Renewable Energy	Hydropower	Biomass	49	51
Increased wood content	Grid mix	Natural gas	68	32

2.2.3. Scenarios

The underlying scenarios are listed in Table 2. MAPE was used as a coupling agent with 3% content in WPC. It was treated as a plastic which raised the overall plastics content to 51% and decreased the wood content to 49% in the "50:50" (wood:plastic) WPC matrix in the basic scenario (100%–3% = 97% thereof calculating wood and plastics share).

Speckels (2001) and Höglmeier et al. (2014) stated that the source of energy crucially influences the equivalent system *recycling*, which cannot use waste wood as secondary fuel but uses conventional energy instead. Therefore, a "renewable energy" scenario was considered which uses hydropower and biomass for energy provision.

In the context of energy recovery from secondary wood as the favourable EoL pathway (Speckels, 2001), the wood content was increased to 70% to analyse the significance of wood content in the post-consumer WPC products.

2.3. How to account for material inherent properties?

Considering EN 15804 (CEN, 2013), and especially EN 16485 (CEN, 2014b), wood inherent biogenic carbon is treated as a material inherent property. According to EN 16485 (CEN, 2014b), the assumption of the biogenic carbon neutrality of wood is valid for wood from countries that have decided to account for article 3.4 of the Kyoto Protocol or for wood originating from forests, which are operating under established certificates schemes for sustainable forest management. Under the described circumstances, the biogenic carbon content of wood, expressed as CO_2 in the global warming potential (GWP) parameter, is transferred to the product system as −1 kg CO_2 in the product stage (modules A1 and A3). The wood inherent carbon leaves the product system as +1 kg CO_2 within modules A1 and A3 for wood burned in case of energy generation within the production, and within the EoL stage (module C3) for the actual product itself because it is contained in the recycled wood which has reached the end-of-waste status. Hence, the biogenic carbon balance of a wood product is deemed to be neutral considering all modules from A to C. Summing up these aspects, biogenic carbon neutrality can be assumed for virgin wood particles derived from German forests. Wood inherent carbon contained in recycled waste wood is considered the same way because it is still a material inherent property in secondary materials. The biogenic CO_2 is separately reported and discussed in the LCIA interpretation phase, because the EoL module of the product LCA (module C) was not part of the analysis and the wood inherent biogenic carbon would reduce the overall GWP emissions from cradle-to-gate. Further considerations of carbon emissions as outlined in, i.e., Bergman et al. (2014b), was not considered in this article.

2.4. Software, databases and impact assessment method

The LCA was carried out considering ISO 14040/44 (ISO, 2006a, 2006b), EN 15804 (CEN, 2013) and EN 16485 (CEN, 2014b) using thinkstep GaBi LCA software version 7.2.0.8 with GaBi professional database v6.115 (thinkstep, 2015) as well as the ecoinvent v2.2 database (ecoinvent Centre, 2010). Additionally, background data for wood-based materials was taken from Thünen Institute's ÖkoHolzBauDat project (Rüter and Diederichs, 2012). For all processes, LCI data with German background was prioritised. More information is provided in the Supplementary information.

The CML-IA mid-point impact method (Guinée, 2002) was used for the environmental impact assessment. The environmental

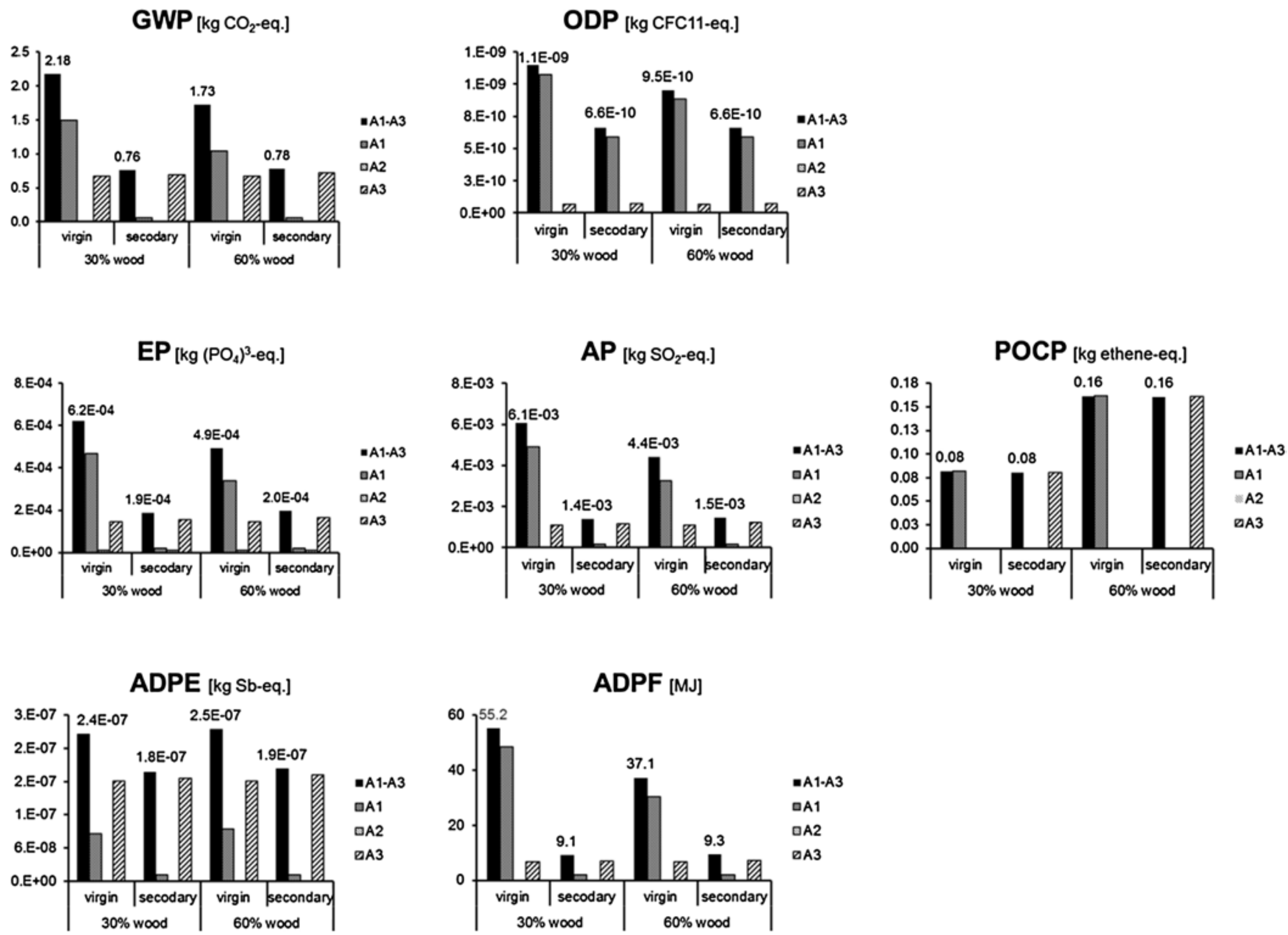

Fig. 5. Environmental parameters of 1 kg WPC compound. Wood inherent biogenic carbon is excluded in GWP.

parameters to be assessed were chosen as required in EN 15804 (CEN, 2013): global warming potential (GWP), depletion potential of the stratospheric ozone layer (ODP), acidification potential of land and water (AP), eutrophication potential (EP), formation potential of tropospheric ozone photochemical oxidants (POCP), abiotic depletion potential for non-fossil resources (ADPE), abiotic depletion potential for fossil resources (ADPF).

3. Results and discussion

3.1. Life cycle impact analysis

3.1.1. Product LCIA

Fig. 5 illustrates the cradle-to-gate (A1–A3) LCIA results of 1 kg semi-finished WPC compound. The potential environmental impacts for WPC produced from secondary materials are lower than WPC produced from virgin materials for each environmental parameter but with the exception of POCP and ADPE. Details are described below and hotspot analysis is presented in Table 3 showing the hotspots per WPC alternative (Y_V, Y_R, Z_V, Z_R).

3.1.1.1. Virgin WPC. If the virgin wood content is increased from 30% to 60% in the virgin WPC matrix (Y_V, Z_V) the potential environmental parameters GWP, ODP, EP, AP and APDF exhibit a better performance (GWP −21%; ODP −17%; EP −21%; AP −27%; ADPF −33%). This can predominantly be related to the lower efforts for the raw material supply of less virgin HDPE amount in module A1. The parameter GWP does not contain wood inherent biogenic carbon at this point of analysis, as it would cause misleading messages to account for wood inherent carbon in cradle-to-gate analyses. It would be accounted for as a negative value in module A1 but accounted for as a positive value in module C3. Consequently, wood inherent biogenic carbon is not displayed in the GWP results for the cradle-to-gate system boundaries. It will be discussed later in Section 3.1.1.4.

In the 60% wood content WPC, the upstream processes of virgin wood particles slightly raises the weight of environmental parameters (GWP +8%, +11% EP, +8% AP, +10% ADPE, +5% ADPF). The production of the coupling agent MAPE in A1 is of major importance for the ODP.

3.1.1.2. Secondary WPC. In contrast to the decrease of environmental impacts due a higher ratio of wood particles in the virgin WPC matrix, a higher amount of secondary wood particles raises the potential environmental impacts of WPC from secondary materials slightly. This can be related to the applied system boundary from the LCA standards EN 15804 (CEN, 2013) and EN 16485 (CEN, 2014b). According to the standards, secondary materials enter the system only with inherent properties in A1. The secondary plastics can be used directly without pre-treatment for the production of WPC in module A3. In contrast, secondary wooden material needs further separation and particle processing from wood chips to wooden particles of the size of approximately <1 mm in module A3.

Table 3
Product LCIA hotspot analysis.

	$Y_{30\%wood}$		$Z_{60\%wood}$	
	Y_V	Y_R	Z_V	Z_R
GWP	61% HDPE, 31% $energy_{el}$ compounding, 5% wood particles	88% $energy_{el}$ compounding, 7% MAPE	44% HDPE, 39% $energy_{el}$ compounding, 13% wood particles	85% $energy_{el}$ compounding, 7% MAPE
ODP	51% MAPE, 42% HDPE, 6% $energy_{el}$ compounding	90% MAPE, 10% $energy_{el}$ compounding	62% MAPE, 29% HDPE, 7% $energy_{el}$ compounding	90% MAPE, 10% $energy_{el}$ compounding
EP	64% HDPE, 23% $energy_{el}$ compounding, 7% wood particles	77% $energy_{el}$ compounding, 12% MAPE	47% HDPE, 29% $energy_{el}$ compounding 18% wood particles	73% % $energy_{el}$ compounding, 10% MAPE, 10% wood particles
AP	74% HDPE, 18% $energy_{el}$ compounding, 4% wood particles	80% $energy_{el}$ compounding, 13% MAPE	58% HDPE, 25% $energy_{el}$ compounding, 12% wood particles	76% $energy_{el}$ compounding, 12% MAPE, 9% wood particles
POCP	98% are emissions to air from drying process of wood particles (see Supplementary information)			
ADPE	68% $energy_{el}$ compounding, 17% HDPE, 11% wood particles	91% $energy_{el}$ compounding, 6% MAPE	65% $energy_{el}$ compounding, 21% wood particles, 10% HDPE	88% $energy_{el}$ compounding, 5% wood particles
ADPF	81% HDPE, 12% $energy_{el}$ compounding, 4% MAPE	75% $energy_{el}$ compounding, 21% MAPE	70% HDPE, 18% $energy_{el}$ compounding, 6% wood particles	73% $energy_{el}$ compounding, 21% MAPE

V-virgin material.
R-secondary material.

Therefore, only the MAPE content is responsible for emissions in A1. Overall, the potential environmental impacts can be predominantly related to the $energy_{el}$ for the compounding process in A3. The hotspot in energy for the preparation of wood particles can be traced to the provision of $energy_{th}$ for the drying of the particles, which is only significant for the WPC with a 60% wood content. The affected environmental parameters are EP and AP.

Transport processes (A2) are environmentally insignificant for WPC alternatives, which is in accordance with literature (Stübs et al., 2012). The higher environmental impacts of POCP (+97%) in comparison to WPC from virgin materials can be related to the emissions to air occurring in wood drying processes. It has to be stated that foreground WPC processing differs completely from wood processing for wood-based panels and detailed information of emissions to air is lacking. The compounding temperature of the WPC alternatives under study was 170 °C. At temperatures between 100 °C and 200 °C, heating of wood produces emissions of water vapour, and volatile organic compound (VOC) emissions (Borrega and Kärenlampi, 2008). It can be expected, that VOC emissions to air occur from HDPE and MAPE at theses temperatures as well, which is likely to be higher from secondary HDPE. Further research is needed to provide quantitative data to implement emissions to air from WPC processing in the LCI.

3.1.1.3. Hazardous content. The elementary analysis showed contents of heavy metals in the secondary materials, which affects the toxicity of the composites to air, soil and water (Table 4). In respect to potential applications of WPC from these secondary materials, threshold values of hazardous substances need to be considered in accordance with DIN EN 71-3 (DIN EN, 2002), for example. The standard is used for products made of particleboard, which are used in domestic applications and in close contact to humans. The standard can be analogously used for WPC products in comparable applications, i.e., outdoor deckings (Sommerhuber et al., 2016). These threshold values were not exceeded in the analysed materials.

An LCIA interpretation is missing, as it would go far beyond a midpoint LCIA and thus would need expanding the methodology. Pizzol et al. (2011a,b) compared different methodologies for LCIA regarding the environmental impacts of metals on human health, on aquatic and terrestrial ecosystems. The authors concluded that the comparison showed poor or no agreement between the methods. The differences are due mainly to the technique used to calculate the characterization factors. The method USEtox is recommended as a method in the context of human toxicity. The model provides recommended and interim characterization factors for human health and freshwater ecotoxicity impacts (Hauschild et al., 2008; Laurent et al., 2011; Rosenbaum et al., 2008; thinkstep, 2016). Future research on WPC processing may consider using the USEtox model (current version 2.0) developed by the United Nations Environment Program (UNEP) and the Society for Environment Toxicology and Chemistry (SETAC) Life Cycle Initiative.

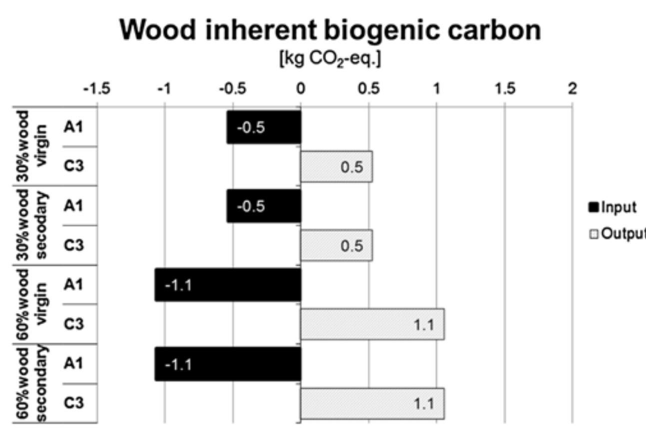

Fig. 6. Wood inherent biogenic carbon of 1 kg WPC compound.

3.1.1.4. Wood inherent biogenic carbon. Fig. 6 shows the carbon balance of wood assuming a carbon neutral balance according to EN 16485 (CEN, 2014b) as described earlier. Carbon is part of the product system (A1) and stored in the product until it is released when reaching the end-of-waste status at the EoL stage (C3). In this study, no additional wooden energy is used at the manufacturing stage (A3), as is the case in harvested wood products (Rüter and Diederichs, 2012; Wenker et al., 2016), for instance.

3.1.1.5. Linking potential environmental parameters with physical parameters of WPC. The applicability of WPC products depends on their specific physical and mechanical performances. In applications where stiffness is crucial, the modulus of elasticity (MoE) is an important parameter. In applications where strength is crucial, strength properties are of superordinate relevance. Based on physical evidence, Sommerhuber et al. (2015) concluded that the WPC

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Table 4
Heavy metal analysis of materials used in WPC compounds [unit = ppm].

	Ag	As	Bi	Cd	Co	Cr	Cu	Ni	Pb	Tl
rHDPE	<LOD	<LOD	<LOD	8.82	5.22	9.28	7.32	0.47	6.52	<LOD
rWood	<LOD	<LOD	<LOD	<LOD	0.46	1.94	2.89	0.76	3.93	<LOD

LOD-limit of detection.
r-secondary material.

Table 5
Increase of environmental parameters (Δ %) by linking physical parameters with product LCA.

	$Y_{30\%wood}$		$Z_{60\%wood}$	
	Y_V	Y_R	Z_V	Z_R
Change in mass based on tensile MoE (Δ)	39%	43%	-[a]	1%
Change in environmental parameter (Δ)				
GWP	27%	5%	–	5%
ODP	37%	39%	–	1%
EP	30%	10%	–	1%
AP	32%	9%	–	5%
POCP	39%	43%	–	1%
ADPE	13%	4%	–	1%
ADPF	34%	11%	–	1%

V-virgin material.
R-secondary material.
[a] Reference.

made of 30% secondary wood and 70% secondary plastics (Y_R) can substitute products made of virgin materials (Y_V).

The results of physical characteristics as shown in Table 1 were linked to the product LCA to provide important information for substitution potentials of secondary materials for WPC. For this purpose, the tensile MoE was considered because it is linearly dependent on the cross-section of a cuboid material. If comparing WPCs of different tensile MoE, the WPC with lower tensile MoE can be adjusted by increasing the mass-input, which respectively increases the volume of the WPC cuboid (Fig. 7). It has to be stated that other physical parameters would change by increasing the mass input to increase the cross-section and respectively the tensile MoE. However, this analysis was simplified to demonstrate how LCA data should be linked with physical characteristics for thorough discussion of substitution potentials of secondary materials.

The tensile MoE of Z_V (60% wood) was the reference because it exhibited the best tensile MoE (2.8 GPa) among the WPC alternatives (Table 1). Therefore, the tensile MoE of the other composites (Y_V, Y_R, Z_R) were adapted by linearly increasing the mass of the WPC. Energy$_{el}$ for compounding was kept constant at 4 MJ/kg.

Table 5 shows the %-change (Δ) of the environmental parameters (A1–A3) when the mass of the WPC compound is increased to achieve equal tensile MoE of the best alternative (Z_v). The potential environmental impacts of virgin WPC with 30% wood content (Y_v) are significantly increased in every parameter due to the increased amount of virgin HDPE. Its counterpart made of secondary materials (Y_R) shows significantly higher values in the OPD and POCP. The increase in ODP can be explained by the increased content of MAPE, whereas the POCP was influenced by the emissions to air originating from the drying processes of wood particles.

As a result, the WPC compound with 60% secondary wood (Z_R) has a high substitution potential in the context of linking technical and ecological parameters. Without this linkage, Z_R showed slightly higher potential environmental impacts than Y_R.

3.1.2. System LCIA

Fig. 8 presents the LCIA of the EoL of WPC by applying the basket-of-benefits approach. In the basic scenario, System A (recycling) generates 1 t secondary WPC. In the equivalent System B (energy recovery), the same amount is produced from virgin materials. Because post-consumer WPC is recycled to secondary material and not recovered to energy in System A, the system needs to be expanded with conventional energy, which would have been otherwise provided by incineration of the post-consumer WPC. Therefore, System A requires 3590 MJ energy$_{el}$ from German grid mix and 8300 MJ energy$_{th}$ from natural gas based on the NCV of wood (18 MJ/kg) and plastics (36 MJ/kg) and the conversion to energy$_{el}$ and energy$_{th}$ in the waste-to-energy plant. The same energy amount is generated through combined heat and power waste incineration plants for wood and plastic treated as residual waste in the equivalent System B. In the renewable energy scenario, the energy$_{el}$ was generated from German hydropower and the energy$_{th}$ from biomass in System A, which did not affect System B. In the scenario with higher wood content (70% wood content), the amount of required energy affected both systems. In System A, the energy$_{el}$ from German grid mix decreased to 3090 MJ and accordingly energy$_{th}$ from natural gas to 7160 MJ. Consequently, the same energy amount is produced by the waste incineration plants in system B.

Recycling of WPC to yield secondary WPC material is the favourable EoL pathway for all potential environmental parameters with the underlying LCI, assumptions and analysed materials. This result is in contrast to the basket-of-benefits LCIA results of waste wood (Speckels, 2001) and engineered wood products (Höglmeier et al., 2014), but Wäger and Hischier (2015) found similar results of the EoL pathways for WEEE plastics.

In addition, pointing at the WPC in this study the equivalent energy for recycling was not as crucial as it had been concluded in previous studies for wood products (Höglmeier et al., 2014; Speckels, 2001), as can be seen in the scenario "renewable energy". Recycling was also beneficial in the scenario with higher wood content (70%) in the WPC-matrix. In all three scenarios, the recycling activities for provision of secondary WPC in System A were a minor hotspot in context of GWP (11%—without wood inherent biogenic carbon), ADPF (13%), AP (18%) and EP (20%), in the basic scenario for instance. Nevertheless, it became significant for ODP (78%), POCP

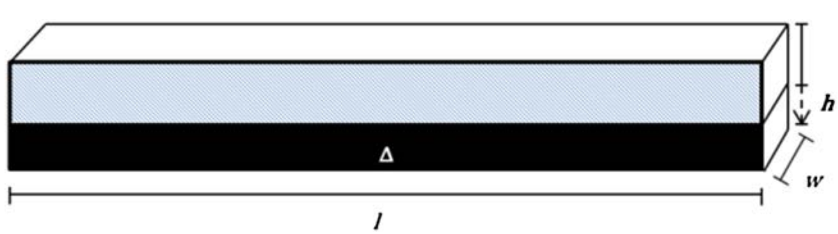

Fig. 7. Increasing the cross-section of a WPC cuboid by increasing mass-input and respectively the height and volume.

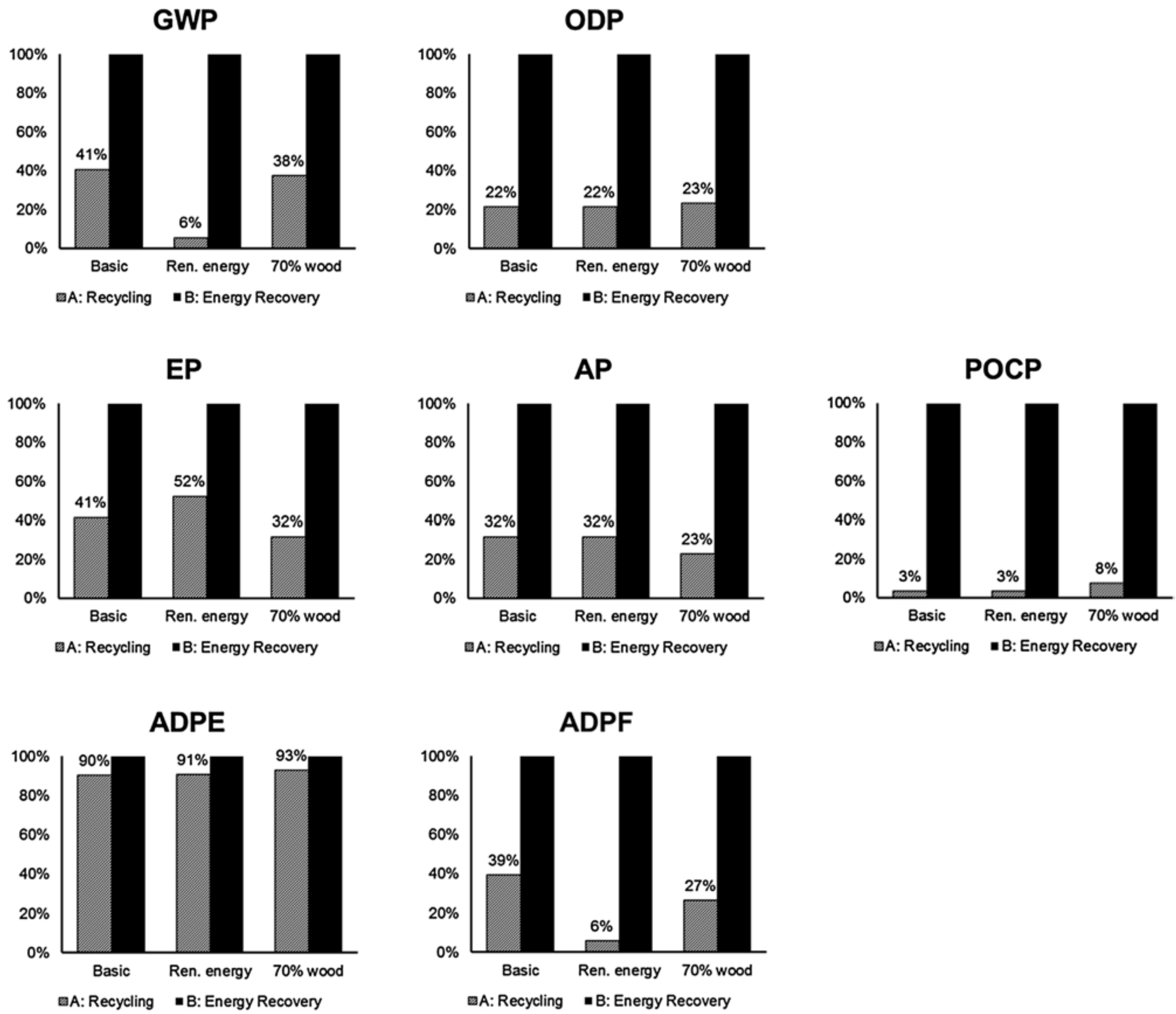

Fig. 8. LCIA of EoL pathways of WPC: recycling vs. energy recovery.

(97%) and ADPE (44%). The hotspots can be traced back to the energy needed for the recycling steps of post-consumer WPC to secondary material and for the provision of virgin WPC in the secondary WPC granulate formula. Because only 3% virgin WPC was used based on data analogously used from plastics recycling (Vidal et al., 2009), the virgin WPC and thereof the virgin plastics content was identified as an ecologically sensitive issue in the system LCA.

Therefore, an additional scenario was considered by increasing virgin WPC content in a secondary WPC formula based on the basic scenario assumptions. If the share of virgin WPC in the secondary compound WPC formula is increased to 50%, the environmental profile of recycling (system A) becomes close the profile of energy recovery (system B) in the environmental parameters GWP (-4%), ODP (-0.1%), POCP (-0.1%), ADPE (-42%), APDF (-2%). As a result, if virgin WPC material is needed in the secondary WPC formula, the use of secondary plastic should be considered.

3.2. Qualitative environmental considerations for the end-of-life of WPC

In previous sections, results showed that recycling of WPC is the environmentally preferable pathway of handling post-consumer WPC. In practice, post-consumer WPC needs to be sorted out from bulky waste if recycling to secondary materials is desired. According to Meinlschmidt et al. (2014), near infrared (NIR) spectroscopy is suitable to sort different WPC based on their molecular composition in the context of plastics into homogenous fractions (PE-WPC, PP-WPC etc.). Li et al. (2015) showed that a combination of Fourier transform infrared (FTIR) spectroscopy and partial least squares regression (PLSR) is promising in the context of determining the share of wood and plastics in post-consumer WPC. Automated sorting techniques are currently not economically feasible for post-consumer wood (Meinlschmidt et al., 2013), but an economic evaluation for post-consumer WPC is lacking and should be considered for future research.

The use of secondary WPC in novel WPC production needs to be technically feasible. Research results addressing recycling WPC to produce novel WPC is limited. Schirp and Hellmann (2013) demonstrated that flexural strength tests (flexural MoE, flexural strength, elongation at F_{max}) and water absorption were positively affected when producing novel WPC from 20 wt-% secondary WPC and 66 wt-% fresh WPC by adding a significant amount of stabilizers. As we examined in a previous section, a high amount of virgin WPC in the secondary WPC matrix leads to higher potential environmental impacts than the energy recovery alternative. As a result, using 2/3 of fresh WPC and 1/3 of secondary WPC in novel WPC would not be an ecologically feasible option.

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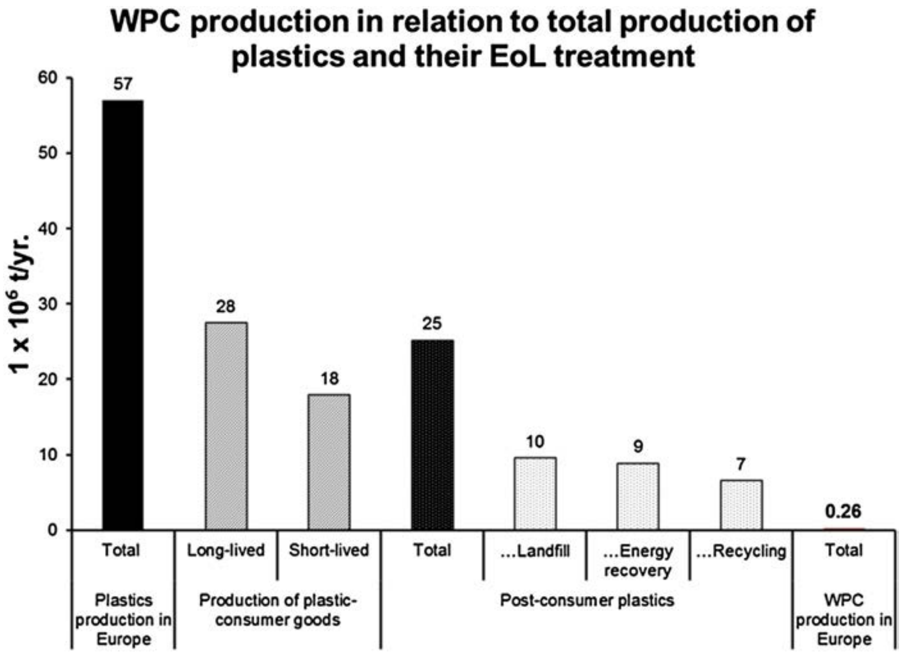

Fig. 9. Data on production and post-consumer of plastics and WPC adapted from Plastics Europe (2015) and Carus et al. (2015).

3.2.1. *Environmental issues of currently applied end-of-life pathways*

In this section, the question is raised, what is done in every day businesses concerning the EoL of WPC and how does this affect the environment?

In the EU, the Waste Framework Directive 2008/98/EC (EC, 2008) is mandatory for the Member states. It aims at protecting "*the environment and human health by preventing or reducing the adverse impacts of the generation and management of waste and by reducing overall impacts of resource use and improving the efficiency of such use* [. . .]" by following the waste management hierarchy. Crucial for any recycling activities is an established market, characterized by demand and supply of the secondary material or fuel. This market exists for post-consumer plastics and wood, but up until now not for post-consumer WPC. Secondary wood and plastic can be recycled in an open-loop system to products different from the predecessor products. For instance, wooden pallets are recycled to particle-boards and plastic bottles to textile fibres. Both the mentioned wood and plastic products can be categorized as packaging materials for which national recycling systems have been established, such as the "Duales System" in Germany or the Austrian Recycling Agency, which are based on the *Extend Producer Responsibility* (EPR) and national acts, such as the German Packaging Materials Ordinance (German Government, 1998).

A balanced market for demand and supply of secondary WPC is needed for an efficient recycling system, but due to the small number of stakeholders and the low production of WPC in Europe (260 kt) this is currently not available today. Fig. 9 demonstrates the amount of hydrocarbon plastic production in the EU-27 including Norway and Switzerland. According to Plastics Europe (2015), these countries contribute to 20% of the global plastics production each year. About 80% is used within the EU-27 for the production of various plastics assortments (i.e., PE, PP, HDPE, PVC, ABS) for a wide range of product categories, such as packaging materials. Only a small amount of these plastic assortments is recycled due to an insufficiently established market, which still has great potential to expand. It may be also questioned, whether the numbers presented by Plastics Europe (2015) represent the reality with respect to the share of recycling vs. energy recovery. A significant amount of plastic residues is discarded during recycling processes due to insufficient quality. This amount is incinerated or just landfilled, which is still the number one EoL pathway for plastics in Europe.

Therefore, the biggest share of post-consumer plastics is energetically recovered in incineration plants or simply landfilled without any recovery. This implies severe environmental problems, such as the diffusion of (micro)plastics in the environment (Essel et al., 2015; Rochman et al., 2015) resulting from littering, insufficient collection or landfill systems due to non-implemented recycling acts. Additionally, hazardous chemical reactions and substances like halogenated organic compounds (PCB, PCP) from PVC materials or polychlorinated dibenzo-dioxins (PCDD) and furans (PCDF) may be generated by incineration and insufficiently filter systems.

According to market data (Carus et al., 2015), PVC is of major importance for the European WPC-production. In 2015, Environmental Product Declarations (EPD) were published for the average WPC outdoor decking. The data was derived from 80% of the German WPC decking companies (IBU, 2015b). In addition, an EPD was released for the average WPC outdoor cladding (IBU, 2015a) representing the production of 100% of the German WPC cladding companies. Table 6 shows the supplied data on materials input, showing that the average WPC product contains PVC (or PE and PP) and 8% additives. Especially the PVC content and the additives in plastic products are likely to make these materials toxic to the environment and human health at the EoL (Thompson et al., 2009).

3.2.2. *Outlook*

WPC should be treated as post-consumer plastic material in the context of its molecular structure and recycling affinity. In addi-

Table 6
Input materials according to published EPD for WPC outdoor deckings and claddings (IBU, 2015a,b).

Material	Description	Ø (%)
Wood fibres	Co-products from sawmill	63
Plastic matrix	PE, PP, PVC	29
Additives	Coupling agent, lubricant, pigments, filler, dispersing agent	8

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tion, an adaption of recycling techniques is required if complex composite materials are to shape the future of the Bioeconomy.

According to Woidasky and Hirth (2012) innovation in technology is still regarded as the route to move towards a sustainable society. Molecular sorting (Forberger and Becker, 2016; Woidasky and Hirth, 2012; Woidasky et al., 2016), and fluorescence sorting (Langhals et al., 2014, 2015) in addition to NIR (Meinlschmidt et al., 2014) are potential technological options based on molecular recycling in comparison to the bulk sorting applied today. Nevertheless, market and economic constraints will remain crucial.

4. Conclusions

The aim of the study was to analyse the potential environmental impacts of using alternative materials for WPC production and finding the desirable EoL pathway for the composites. This was done by using the LCA method. In the context of assessing the ecological difference of WPC made from virgin vs. secondary materials (*RQ 1*), the product LCA methodology was chosen based on the LCA standards EN 15804 (CEN, 2013) and EN 16485 (CEN, 2014b).

Results of product LCA:

- In WPC made of virgin materials, the more wood used, the lower the potential environmental impacts.
- In WPC made of a high amount of secondary wood, processing of secondary wood particles contributes to the overall environmental impacts because secondary plastic granulates are directly useable in context of an established market of high quality secondary plastic granulates.
- Nevertheless, linking physical parameters with ecological parameters, the more secondary wood is used the lower the potential environmental impacts, which results in best ecological and technical alternative.

Because environmental problems arise for the EoL of WPC, the following question was proposed as *RQ 2*: What is the environmentally preferable option at the end-of-life stage of the composites: utilization as a secondary fuel or as a secondary material? To answer this question, the basket-of-benefits approach for system expansion was chosen to provide systems with the same functions.

Results of system LCA:

- Recycling of post-consumer WPC is the favourable EoL pathway.
- Recycling of post-consumer WPC is sensitive to the WPC formula, if a significant amount of virgin WPC is added to the secondary WPC.
- WPC is more related to plastics than wood recycling and needs to be treated as post-consumer plastics.
- A system wide take-back organization needs to be established to recycle post-consumer WPC.

The challenge will be to handle and recycle the waste appropriately at a high quality without diffusion of hazardous substances to generate markets for secondary WPC in terms of the EPR concept. Sirkin and Houten (1994), among first authors to research the cascading of resources, discussed resource quality loss and quality gain through recycling efforts, which consume energy, labour and additional resources. Upcycling, in their study proposed as "re-linking" – which is to provide the resources "*upwardly to a higher utility level in the same cascade, or to a new substance cycle*" – is unfeasible for products containing inter-linked material structures created by chemical bonding or the like. Therefore, WPC would definitely be the last step in a cascade chain of primary and secondary materials. Up to now, the wood, which would have been used as a material for solid wood or engineered wood product first, is enriched with plastics and additives in the WPC product. At the EoL, WPC have to be treated as bulky or hazardous household waste to be incinerated, which leads to environmental problems and a loss of valuable materials.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.resconrec.2016.10.012.

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5.4 Case study: LCA of ECOLIFE® window

Table 11 presents the environmental parameters of ECOLIFE® from cradle-to-gate. The environmental hotspots can be clearly linked to the supply chains of raw materials (module A1). Transports (A2) are of minor environmental importance for this product system, but certainly not negligible. Manufacturing (A3) is of minor importance as well when compared to module A1. The environmental loads in manufacturing are derived from provision of $energy_{el}$, auxiliary materials and packaging materials.

Table 11. Potential environmental impacts of the product stage (cradle-to-gate) of ECOLIFE® window system; wood inherent biogenic carbon is excluded in GWP.

Environmental parameter	Unit	Product stage		
		A1	*A2*	*A3*
Global warming potential (GWP)	kg CO_2-eq.	120.6	1.5	9.0
Ozone depletion potential (ODP)	kg CFC11-eq.	2.3E-07	5.7E-13	1.9E-12
Acidification potential (AP)	kg SO_2-eq.	0.7	0.0	0.1
Eutrophication potential (EP)	kg $(PO_4)^3$-eq.	8.5E-02	3.2E-03	4.8E-03
Formation potential of tropospheric ozone photochemical oxidants (POCP)	kg ethene-eq.	2.2	0.0	0.0
Abiotic depletion potential for non-fossil resources (ADPE)	kg Sb-eq.	4.7E-03	5.4E-08	5.8E-07
Abiotic depletion potential for fossil resources (ADPF)	MJ	1742	20	186

The environmental hotspots of ECOLIFE® is presented in Figure 8 and Figure 9 by presenting cumulated cradle-to-gate results (A1–A3) for each process step (P1–P4) to discuss the environmental share of WPC manufacturing in a product system. The WPC profile is produced in steps P1–3. It is questionable if the aluminium stripe should be included in the eco-profile of the WPC profile. However, it is necessary for strength and rigidity improvements for the WPC profile as well as to satisfy decorative effects. Therefore, it is included for this LCIA interpretation.

The supply chain of aluminium, $energy_{el}$ and virgin HDPE are the major contributors for the GWP, what is also visible for AP, EP and ADPF. The supply of virgin wood particles becomes an environmental hotspot for AP and EP, which is linked to the long shipping distance (A2) between Germany to China. The air emissions produced by drying the wood particles are the main contributor to POCP. Reliable data is missing for emissions to air of foreground WPC processes. The ODP is clearly dominated by foaming agents in the PUR hot melt. The ADPE is influenced by the estimated static lifetime of the non-fossil (mineral) resources needed for window assembly in P4.

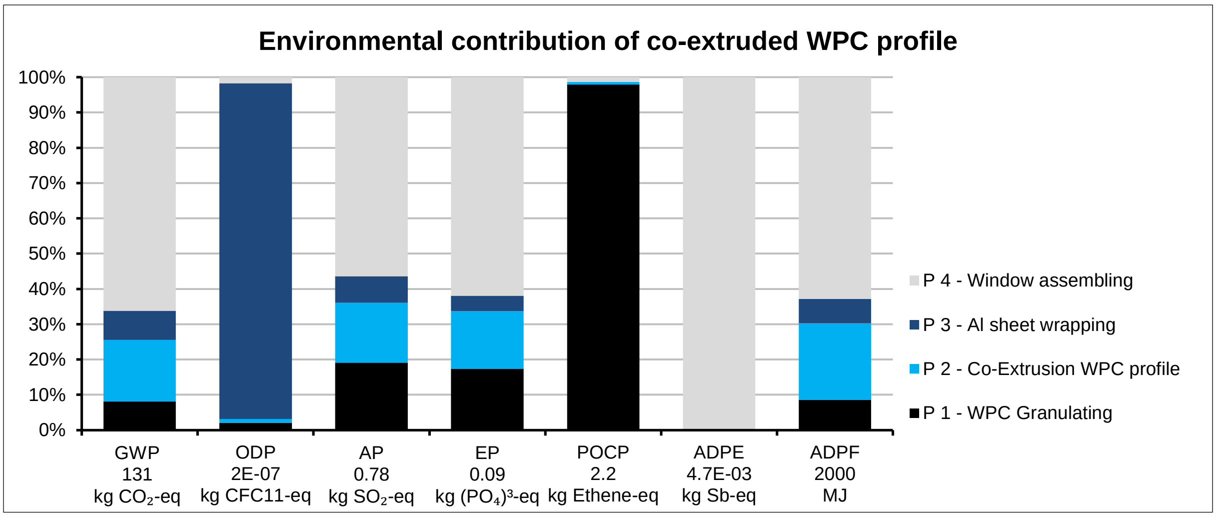

Figure 8. LCIA contribution of the co-extruded WPC profile. Each process (P) is the cumulated LCIA from cradle-to-gate (A1 – A3); wood inherent biogenic carbon is excluded in GWP

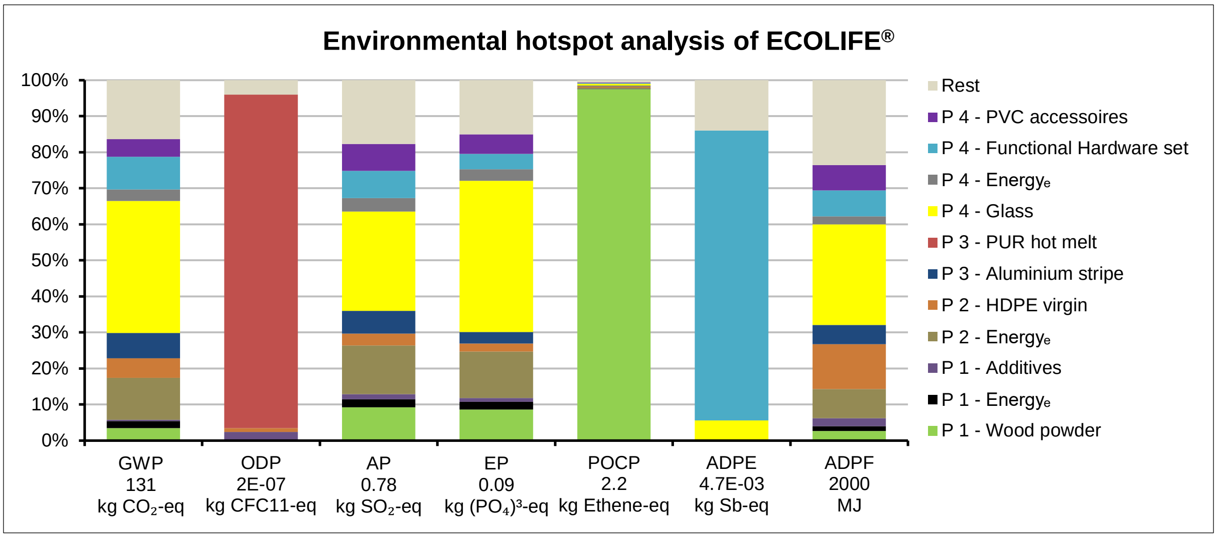

Figure 9. LCIA hotspot analysis; wood inherent biogenic carbon is excluded in GWP

Figure 10 illustrates the carbon balance of wood assuming a carbon neutral balance according to EN 16485. The wood inherent biogenic carbon is derived from wood particles and wood transport pallets (A1) and stored in the product until it is released when reaching the end of waste status at the EoL stage (C3).

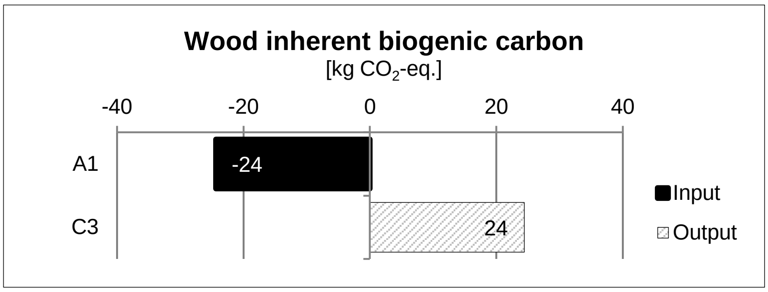

Figure 10. Wood inherent biogenic carbon of ECOLIFE®

6 Discussion and synthesis

6.1 Identification of potential secondary materials for WPC

Addressing the overarching research question:

(1) *Which secondary materials from which waste streams can be identified to substitute primary (virgin) materials in WPC production?*

Statistical data from Destatis and eurostat varied considerably from data found in reports, such as market data on German (waste) wood flows (Mantau 2012b, 2015; Mantau et al. 2012; Seintsch & Weimar 2012, 2013) and secondary plastics (Consultic 2014; Plastics Europe 2015; Villanueva & Eder 2014).

After studying and comparing statistical data and market data, the data from the above-mentioned reports were found to be more accurate. On the plastics side, they reflect the specific type of polymers from specific product categories: packaging, electronical equipment etc. In the statistical databases, the specific material within the category is unknown. Waste is classified in the European Waste Catalogue (EWC) (EC 2014). For example, plastics from lightweight packaging (assortments of PE, PP and others) are categorized as EWC 15 01 02. The cross amount of this category is first allocated to recycling in the statistical database, but after sorting a substantial amount is rejected due to impurities and residuals. These rejects are further incinerated and recounted in the databases (EUWID 2015; Thomé-Kozmiensky 2013). Concerning waste wood, categories are used to label the waste wood based on the level of contaminations (A I, A II, A III and A IV). Some waste wood assortments are categorized from different contamination levels in the same EWC in statistical databases.

6.1.1 Secondary wood particles

About 72 x 10^6 m^3/yr of coniferous and non-coniferous roundwood are consumed in Germany, as visualized in Figure 11. About 36% of roundwood are used as fuelwood only without being used as saw logs or industrial roundwood first in terms of using its carbon storage potential. Fibre- and particleboards consumes 10.5 x 10^6 m^3/yr of industrial roundwood. Post-consumer wood constitutes 10.9 x 10^6 m^3. Energy recovery of waste wood is the predominant EoL option (78%). The German particleboard industry uses 20% post-consumer wood to substitute virgin particles. Disposal in landfills is almost negligible (Mantau et al. 2012) due to the ban on organic waste materials in accordance to the European Waste Framework Directive 2008/98/EC (EC 2008).

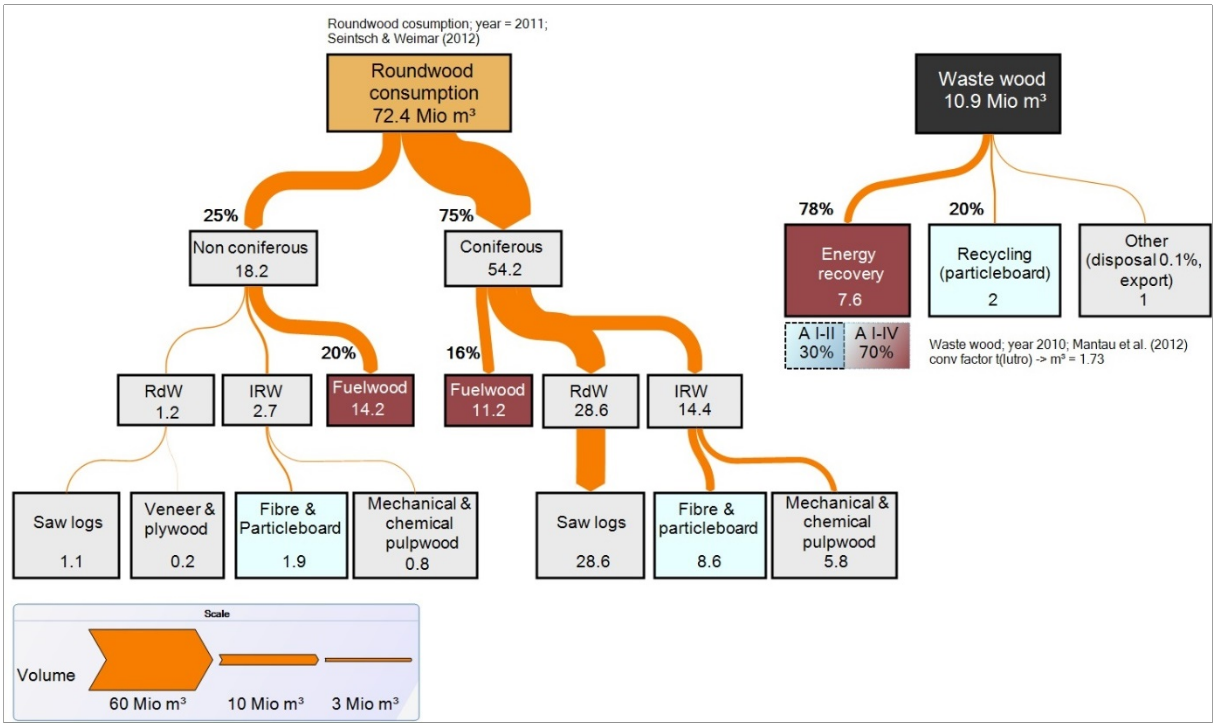

Figure 11. Roundwood and waste wood consumption in Germany; imports are excluded

Table 12 shows the amount of wood particles consumed by German and European producing WPC companies. It was not possible to allocate the wood particles to the coniferous and non-coniferous wood flow because the share is unknown.

Table 12. Wood particles consumed for WPC production; Conversion factor t to m³ = 4.9, based on 0.2 g/cm³ bulk density (measured) of Norway spruce particles (Benthien et al. 2016). WPC data for calculation based on Carus et al. (2015a); *forecasted, base year was 2012 and 20%/yr production growth was assumed

	2010		2012		2015*		2017*		2019*	
(in 1x 10^3)	t	m^3	t	m^3	t	m^3	t	m^3	t	m^3
Europe	100	491	118	580	149	729	179	875	214	1,049
Germany	45	221	53*	261	67	328	80	394	96	473

Although the amount of wood particles consumed by the WPC sector is small (2% of total fibre and particleboard producing industry), resource efficiency in the German wood sector is likely to become more important in terms of the political push to use more wood resources in products and as energy carrier (BMEL 2014; EC 2012b). Waste wood of category A II (mixed A II) and post-consumer particleboard (A II) can be identified on a quantitative relevance basis to substitute virgin or co-product particles in WPC. The largest share of post-consumer wood (5×10^6 t/yr = 70%) can be classified as A III, which is a mixture of untreated, treated, and contaminated waste wood. A separation into fractions of this amount for recycling (A I – A II) and energy recovery (A III – A IV) would presume an intensified and cost-intensive separation process which strongly influences the chosen management option for recovered wood (Meinlschmidt et al. 2013; Meinlschmidt et al. 2014; Merl et al. 2007). If a separation becomes feasible and wood becomes more viable in terms of resource cascading (Sirkin & Houten 1994), about 30% (1.3 x 10^6 t/yr.) of currently energetically used post-consumer wood could be additionally mobilized as secondary material according to data from BAV (2012) based on Mantau et al. (2012). This amount is currently burned in small/medium combustion plants regulated by 1st BImschV (German Government 2010).

A competing demand for secondary wood exists as the particleboard industry is already using secondary wood particles (20%). Combined heat and power plants or energy intense industries using secondary wood fuel to substitute fossil energy resources are the biggest competitors to the use of waste wood as a secondary material instead as a secondary fuel. This is the result of specific policy frameworks (EC 2009; German Government 2014). As visible in Figure 12, also secondary wood prices are rising, but remain well below the price for specialized virgin wood particles from softwood (250–400 €/t) (Carus et al. 2015b) and wood co-products. Financial incentives based on the EEG for utilizing secondary wood fuel were highlighted to push the price for wood assortments according to Härtl & Knoke (2014) and Höglmeier et al. (2014). However, the incentives are not a major influence on the price development of fuelwood, according to Bioökonomierat (2016), and also not likely to influence the price of waste wood in the near future. It is more related to the price for crude oil. The increase in oil prices resulted in a fall from the peak in June 2008 (135 USD/bbl) to an unexpected 28 USD/bbl in January 2016. At first glance, the price for waste wood seems to follow the same dependency. This is likely because in times where oil is cheap, there is no need for a renewable energy alternative in context of ecological economics principles (Neumayer 2013; Perman et al. 2003). This should result in decreasing prices of waste wood on the one hand to a sufficient supply of waste wood utilizable as secondary material. This tendency is already visible (Figure 12). However, this would only be one argument for decreasing waste wood prices.

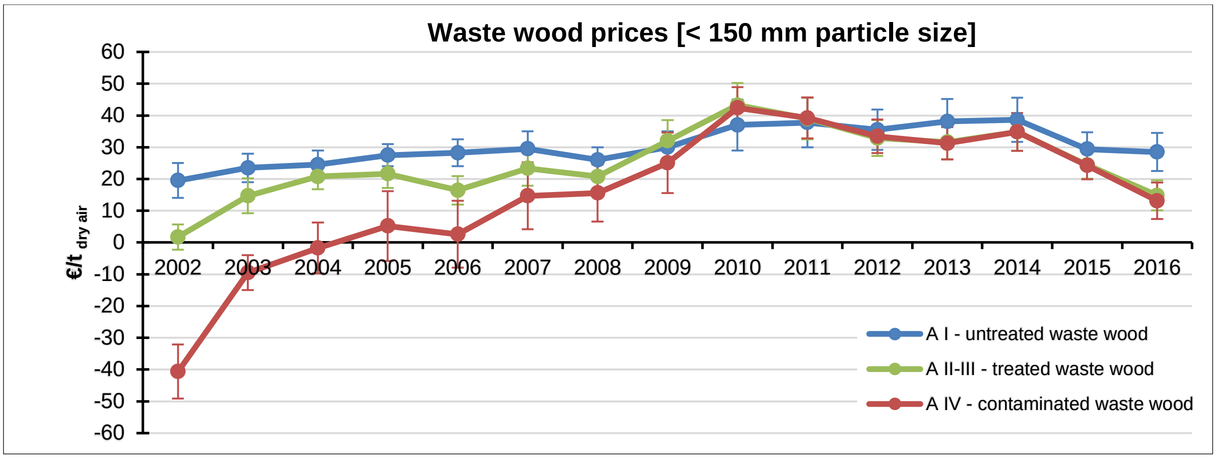

Figure 12. Nominal waste wood prices. Data based on EUWID Recycling und Entsorgung and EUWID Holz- und Holzwerkstoffe (multiple issues of years 2005–2015) and BAV (2012)

6.1.2 Secondary thermoplastic polymers

Plastic waste and its recycling play an essential part in the circular economy. The EU-27 generates about 25 x 10^6 t/yr of plastic waste – tendency growing – which is more than half of the yearly produced amount of virgin plastics. In Germany, plastic packaging constitutes the largest plastics market share, but plastic packaging materials are intended, in most cases, to have a short life cycle. About 1.53 x 10^6 t/yr of lightweight plastic packaging waste is collected separately in context of the German Packaging Materials Ordinance (German Government 1998). Of these packaging materials, thermoplastic polyolefins (LDPE, LLDPE, HDPE and PP) comprise the biggest share (Plastics Europe 2015). In Germany, about 30 x 10^3 t of PE and 44 x 10^3 t of PP (both virgin or post-industrial) were used for WPC decking production in the year 2012 (Carus et al. 2015a). About 500 x 10^3 t/yr of PE and 200 x 10^3 t/yr of PP from packaging waste are recycled to secondary plastics. So, based on this quantitative evaluation, secondary polyolefins would be readily available for substituting virgin materials in WPC. Nevertheless, export flows of secondary plastics should be studied in detail for future research.

In addition to post-consumer polyolefins from packaging, secondary plastics from controlled WEEE recycling are a promising alternative for utilization in WPC. An amount of 12 x 10^6 t of WEEE is expected by the year 2020. Of that, about 1.5 x 10^6 t of ABS and PS are contained in WEEE and should be available on the recycling market due to legislative requirements (EC 2012a; Köhnlechner 2014; Salhofer et al. 2015; Vidal-Legaz et al. 2016) in context of economic and environmental concerns related to illegal WEEE trade (Huisman et al. 2015; Premalatha et al. 2014). The use of virgin PS and ABS for WPC is currently not practiced in Germany, although some manufacturers in the USA are using PS (Klyosov 2007). ABS is much higher in price than polyolefin thermoplastics (EUWID 2016d).

WPC would be a value added product made of these secondary polymers, as they are readily available and plastic recyclers are looking for new markets. German recycling companies are already struggling due to the competition from virgin plastic converters (EUWID 2016a) as the profitability of using secondary plastics depends on the prices of crude oil and its virgin plastics. From the viewpoint of a manufacturer, using virgin plastics is cheaper in times of USD 30,-/bbl than adding secondary materials to the material formulation of products. Utilization of secondary granulates may only be economically feasible at crude oil price of ≥ 51 USD/bbl (45 €) (EUWID 2016b). The current crude oil price is still well below 51 USD/bbl (May 2016). Therefore, economic incentives for secondary materials are especially needed in times, when virgin materials are cheaper than the substitutable secondary materials as proposed by the "Roadmap to a Resource Efficient Europe" (EC 2011b).

However, looking at Figure 13, price for secondary HDPE (930 €/t ± 55) were two thirds of the price of virgin HDPE (1375 €/t ± 250) at a crude oil price of 37 €/bbl (max.) in March 2016 (Anonymous 2016; EUWID 2016c, 2016d). Prices of secondary ABS and PS are also about two thirds of the price of virgin plastics. With the underlying assumptions of recycling costs from post-consumer waste to secondary plastics (Villanueva & Eder 2014), using wood co-products would be the economically most feasible alternative for WPC. The price for additives are approximately in the same range as the standard thermoplastics prices, reducing even more the overall costs of the wood particles in WPC. Therefore, fluctuations in the price of wood particles are not likely to be not as critical for the WPC-processing industry as they are for engineered wood products with about 90% wood content, such as particleboards. Additional Sankey diagrams for secondary ABS and PS from WEEE and PP from packaging waste are provided in the supervised Master thesis of Wang (2016).

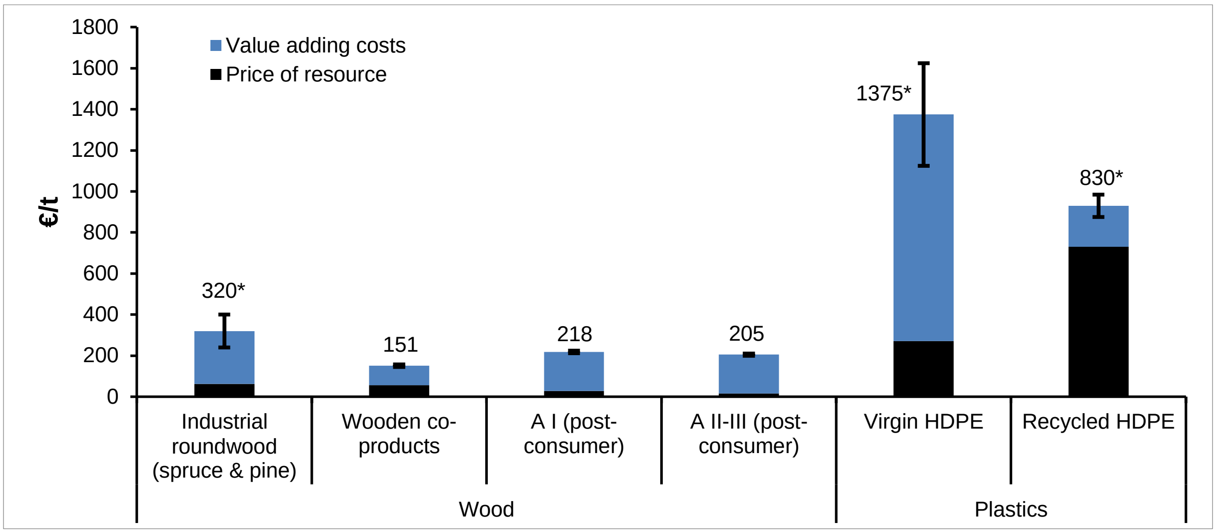

Figure 13. Price of neat materials and their value adding costs to be utilizable in WPC. Price information reflects market situation March–April 2016. Data from EUWID Recycling und Rohstoffe, Gerling (2016), Villanueva & Eder (2014), Kunststoff Information (2016), Carus et al. (2015b). *margin included; Wood co-products and waste wood are processed at the WPC manufacturer which costs are based on assumptions

6.2 Qualitative aspects of using secondary materials in WPC

Addressing the overarching research questions:

(2) *What are the differences in physical and mechanical properties of WPC produced from secondary materials in comparison to their virgin counterparts?*

(3) *What obstacles need to be considered in terms of applicability?*

6.2.1 Physical properties

Table 13 provides a comprehensive overview of the physical properties of laboratory-scaled WPCs. The incorporation of wood particles into the plastic matrices resulted in a higher density of WPC in comparison to the neat materials. Although the density of Norway spruce (*Picea abies*) is approximately 0.43 g/cm^3, in WPC, the wood lumens are filled with the thermoplastic polymers and squeezed during the compounding process. The density of the wood cell wall is approximately 1.4 g/cm^3 (Kollmann 1951), which increases the plastic matrix to a density of the WPC to > 1 g/cm^3. The density of wood particles was not investigated in detail. It is very likely that the density of the different wood sources were equal. This is because the density of the HDPE sources was approximately 0.95 g/cm^3 and secondary wood materials from mixed waste wood A II and particleboard contained a high share of softwood, mostly spruce and pine.

Table 13. Physical properties of WPC with 60% wood content including coupling agent. WW… post-consumer waste wood (mixed A II), PB… post-industrial particleboard (A II). Standard deviation in greyish subscript

	Density (g/cm^3)				Water absorption after 28 days (%)			
		WPC incorporated with…				*WPC incorporated with…*		
	Neat plastic	Norway spruce	WW	PB	Neat plastic	Norway spruce	WW	PB
pHDPE	$0.95_{0.001}$	–	$1.16_{0.003}$	–	$0.1_{0.1}$	–	$6.1_{0.3}$	–
rHDPE	$0.96_{0.001}$	$1.17_{0.002}$	–	–	$0.8_{0.1}$	$4.0_{0.2}$	–	–
rPP	$0.92_{0.001}$	$1.14_{0.003}$	–	$1.14_{0.003}$	$0.1_{0.3}$	$7.9_{0.4}$	–	$8.3_{0.2}$
rABS	$1.05_{0.001}$	$1.25_{0.003}$	–	$1.24_{0.008}$	$0.7_{0.1}$	$9.6_{0.2}$	–	$8.2_{0.1}$
rPS	$1.04_{0.002}$	$1.25_{0.001}$	–	$1.24_{0.003}$	$0.2_{0.1}$	$13.2_{0.1}$	–	$11.6_{0.1}$

According to EN 15534-4 (CEN 2014b) water absorption of WPC decking is required to be ≤ 7% (mean) in terms of durability. This was only achieved with secondary HDPE-WPC with waste wood A II from post-consumer packaging materials and their virgin counterparts but not with all other tested specimens.

6.2.2 Mechanical properties

The applied manufacturing process is a crucial parameter for the mechanical properties of WPC (Krause et al. 2013; Kumari et al. 2007; Migneault et al. 2008; Migneault et al. 2014). Two different process techniques were applied. The WPC specimens of mixed waste wood (A II) were produced in a laboratory-internal mixer, and compression moulded in a computerized hydraulic hot press. Specimens containing post-industrial particleboard were extrusion-compounded and injection moulded. Both processes resulted in different rheological flow abilities for the WPC compound and its resulting mechanical properties. This is visible in higher standard deviations of compression moulded WPCs (Table 14). The wood content distracts the flow ability of plastics molecules during the melting process (Cui et al. 2010), regardless whether virgin or secondary wood particles were used. Flow is also affected by the MFI of the plastics, the content of wood particles, and the additives (Krause et al. 2013; Kumari et al. 2007; Migneault et al. 2014; Teuber et al.). In addition, the molecular structure of the monomers, which determine the type and density of the neat thermoplastic polymers, were also a crucial influence on the mechanical WPC properties.

Table 14. Mechanical properties. p…primary, r…secondary, W…Norway spruce, WW… post-consumer waste wood (mixed A II), PB… post-industrial particleboard (A II), M… MAPP, S… SMA

WPC formulation	Flexural tests: Strength F_{max} (MPa)		Flexural tests: fMoE (GPa)		Tensile tests: tMoE (GPa)		Tensile tests: Elongation at break (%)		Tensile tests: Strength F_{max} (MPa)		Charpy impact test: Impact strength (kJ/m²)	
	MV	SD	MV	SD	MV	SD	MV	SD	MV	SD	MV	SD
pHDPE	25.4	0.62	0.99	0.04	1.10	0.05	26.2	1.87	23.8	0.79	-	-
rHDPE	25.1	0.82	0.91	0.06	1.05	0.03	23.5	2.84	23.1	0.49	-	-
pHDPE/W30	33.6	2.12	1.55	0.16	1.74	0.09	3.33	0.34	18.5	0.57	8.35	1.52
rHDPE/WW30	31.4	2.00	1.58	0.14	1.65	0.06	2.65	0.29	15.7	0.37	7.61	1.48
pHDPE/W60	42.3	4.30	2.77	0.42	2.87	0.06	1.60	0.12	21.6	0.63	7.97	2.36
rHDPE/WW60	31.6	2.50	2.62	0.20	2.74	0.08	1.06	0.05	16.0	0.42	5.42	1.07
rPP	-	-	1.10	0.02	1.32	0.01	24.5	6.72	22.4	0.08	-	-
rPP/W30	37.3	0.33	2.50	0.02	2.81	0.03	3.30	0.47	20.5	0.09	-	-
rPP/PB30	37.2	0.47	2.34	0.03	2.46	0.03	3.46	0.24	18.8	0.30	-	-
rPP/W60	32.5	0.40	4.77	0.04	4.28	0.03	0.86	0.09	16.0	0.15	-	-
rPP/PB60	28.3	0.91	3.78	0.08	3.14	0.07	1.00	0.02	13.0	0.16	-	-
rPP/W60/M	63.2	1.56	5.58	0.12	6.61	0.08	0.97	0.05	34.4	0.41	-	-
rPP/PB60/M	55.2	1.05	5.00	0.08	4.63	0.09	1.28	0.02	28.1	0.22	-	-
ABS	-	-	2.42	0.08	2.42	0.02	4.20	2.15	35.6	0.21	-	-
rABS/W30	69.4	1.37	4.86	0.08	4.81	0.05	1.35	0.07	39.6	0.38	-	-
rABS/PB30	64.4	1.11	4.66	0.08	4.64	0.07	1.27	0.07	36.4	0.32	-	-
rABS/W60	77.8	2.15	8.37	0.16	7.88	0.10	0.75	0.04	43.8	1.06	-	-
rABS/PB60	66.3	1.92	7.98	0.15	7.55	0.09	0.65	0.06	37.0	1.04	-	-
rABS/W60/S	85.7	2.69	8.14	0.11	7.74	0.24	0.89	0.04	48.9	0.92	-	-
rABS/PB60/S	78.8	3.62	8.24	0.12	7.64	0.08	0.77	0.04	43.5	0.48	-	-
PS	-	-	2.14	0.02	2.15	0.02	12.4	5.92	18.2	0.46	-	-
rPS/W30	44.2	0.79	4.60	0.06	4.50	0.04	1.08	0.03	21.2	0.11	-	-
rPS/PB30	47.5	0.95	4.41	0.08	4.24	0.06	1.20	0.07	22.1	0.07	-	-
rPS/W60	49.3	1.44	7.68	0.09	6.84	0.06	0.58	0.04	24.4	0.33	-	-
rPS/PB60	45.6	1.00	7.39	0.13	6.31	0.09	0.54	0.04	21.4	0.06	-	-
rPS/W60/S	58.3	1.38	7.53	0.16	6.92	0.08	0.73	0.03	29.4	0.23	-	-
rPS/PB60/S	52.0	0.70	7.64	0.12	6.67	0.08	0.61	0.04	24.7	0.04	-	-

6.2.3 Challenges of using secondary materials for WPC manufacturing

6.2.3.1 Waste wood sorting

Waste wood (mixed A II) was collected from a local waste wood supplier in Hamburg in October 2014. The waste wood was stored outside and crushed to a particle size of > 200 mm with a hammer mill at the recycling site. The waste wood contained mostly transport pallets from softwood, with a minor content of hard wood, residuals of metals (nails), glass, engineered wood products and plastic particles. The moisture content was measured to be 33% at the laboratory. The waste wood had to be dried for storage purposes and sorted from abrasive impurities. For the WPC made of waste wood, the computerized hydraulic hot press was used instead of extruder and injection moulding machinery to ensure that no undetected abrasive residuals would damage the WPC processing technology and tools. In summary, considerable effort was needed to prepare the crushed waste wood particles from > 200 mm grading size to their final shape of approximately < 1 mm particle size.

As discussed earlier, this effort expressed in value adding costs and the likeliness of undetected abrasive residues in the sorting process, brings to question the practicality of using secondary wood materials for WPC. However, from physical and mechanical points of view, the secondary wood materials are viable for WPC and achieve comparable results in stiffness as virgin Norway spruce particles or comparable co-products thereof. This is likely because the softwood content in the waste materials (A II) was high and it can be assumed it was high in particleboard as well.

6.2.3.2 Immiscibility of secondary plastic materials

From the perspective of WPC producers, a constant quality of secondary materials is crucial for a safe resource supply in the long term. Therefore, a sample comparison of two secondary PP "batches" was done by physical and mechanical characterization. Both secondary PP batches were obtained from INTERSEROH Dienstleistungs GmbH, Germany, in June 2014 and June 2015. As results show in Table 15, the properties were almost equal for both batches.

Table 15. Differences in physical and mechanical properties of secondary PP from two batches

	Unit	Secondary PP received in year 2014	2015	Δ '14 to '15
Density	g/cm^3	0.92	0.92	0%
Water absorption after 28 days	%	0.00	0.00	0%
Flexural MOE	GPa	1.10	1.07	▼ -3%
Tensile MOE	GPa	1.32	1.56	▲ 18%
Tensile strength	MPa	22.42	22.14	▼ -1%
Tensile elongation at break	%	24.50	33.76	▲ 38%

Weiss (2016) mentioned that a constant quality of secondary plastics is only achievable by compounding various batches of specific secondary plastics to achieve homogeneity, reduce the immiscibility and resulting disadvantages thereof. The immiscibility could also be reduced or eliminated by using additives according to Mantia et al. (1992), Goodship (2007), Waldman &

Paoli (1998), Kazemi-Najafi (2013). However, this would result in additional costs and worsen additionally the environmental profile of the composites (see Section 6.3.1).

6.2.3.3 Hazardous inorganic content

Threshold values of hazardous substances need to be considered in accordance with the DIN EN 71-3: Safety of toys (DIN EN 2002) with respect to the substitution of the identified secondary materials in WPC decking. The standard is used for products made of particleboards, which are used in domestic applications and in close contact to humans. In the case of closed-loop recycling – secondary WEEE plastics are used in the same product category – threshold values of the Restriction of Hazardous Substances(RoHS) Directive (EC 2011a) need to be considered. The ABS and PS were derived from category 3 (IT and telecommunications equipment) and mainly from category 5 (lighting equipment). A comprehensive overview of the observed elements with respect to the threshold values in context of the application potentials is provided in Table 16.

Table 16. Heavy metal analysis. Values in ppm. LOD… limit of detection. WW… post-consumer waste wood (mixed A II), PB… post-industrial particleboard (A II)

	Analysed materials							**Thresholds**	
Element	*rHDPE*	*rPP*	*rPS*	*rABS*	*Norway spruce*	*WW*	*PB*	*RoHS*	*DIN EN 71-3*
As	< LOD	1.17	0.20	0.30	< LOD	< LOD	0.21	-	25
Cd	8.82	2.09	10.65	58.99	0.30	< LOD	0.26	100	75
Co	5.22	5.53	6.88	10.84	0.07	0.46	0.26	-	-
Cr	9.28	5.26	4.53	7.05	0.21	1.94	3.57	1000	60
Cu	7.32	18.72	11.69	19.07	0.93	2.89	3.24	-	-
Ni	0.47	0.77	5.74	10.92	0.24	0.76	0.95	-	-
Pb	6.52	0.82	8.45	4.13	0.75	3.93	10.54	1000	90

Currently, it is prohibited to use Cd to give colour to PE, PP, and PS, as a pigment for lacquers and paints, and as a stabilizer for packaging and cable/wire insulation according to the REACH Regulation (EC 2006). Cd was found in all secondary plastics, with a high concentration in secondary ABS for which Cd is not prohibited (EC 2006). Copper chrome arsenic (CCA), are no longer allowed to be used in wood preservatives in Germany and Europe. Pb was historically used as pigment for white coatings of doors and window frames which have been further recycled to particleboards (BAV, 2012). The use of these pigments has been restricted since 2005 in Germany, so diffusion of Pb occurred. Diffusion of hazardous substances is regulated by the KrWG (German Government 2012). It's time related aspects is a significant shortcoming of using secondary materials as many substances have not been declared as hazardous or where not known to be so in the past. Riedel et al. (2014) reported as well on the presence of hazardous materials in waste wood categories, where no hazardous materials should be expected.

The analysis in this thesis was conducted as a screening analysis and had not the intention to be complete. The results reflect only a small range of inorganic hazardous substances. For example, the content of mercury is an important parameter for categorization of waste wood. Organic hazardous content would be also of much interest, such as PVC residuals and polychlorinated biphenyls. Further, an evaluation on human or animal health was not considered. The USEtox model could be considered in the context of impacts on human health and freshwater ecotoxicity (Hauschild et al. 2008; Laurent et al. 2011; Rosenbaum et al. 2008).

6.3 Ecological aspects of WPC

Addressing the overarching research questions:

(4) *What is the difference of secondary vs. virgin materials in WPC based on LCA?*

(5) *What is the ecological preferable EoL pathway of the composites*

6.3.1 Product stage

The supply chain of virgin HDPE can be identified as the main contributor to the environmental parameters when comparing 1 kg WPC compound as a semi-finished product from alternative materials. The higher the wood content (60%) was used in the virgin WPC matrix the better the environmental profile. The WPC compound produced from secondary materials containing 60% of mixed waste wood (A II) was the best alternative in context of linking environmental parameters and tensile MoE performances. However, using the semi-finished compound further for outdoor decking, the WPC becomes less durable with a higher wood content (Miller et al. 2015). This can be linked to the loss in mechanical properties (strength and stiffness) based on water uptake and biological degradation in terms of fungal decay (Krause & Gellerich 2014). Therefore, replacement of a complete WPC decking board would be needed what increases the total environmental impacts along the lifecycle. Using secondary HDPE is therefore an ecological important alternative to minimize the environmental parameters of WPC decking along its life cycle. Up to now, only post-industrial plastics are allowed for German produced WPC decking to receive the quality label of Qualitätsgemeinschaft Holzwerkstoffe e.V (Qualitätsgemeinschaft Holzwerkstoffe e.V. 2016).

If WPC is derived from secondary materials, the $energy_{el}$ for compounding wood particles and plastics mainly influenced the environmental parameters of the semi-finished WPC compounds. The coupling agents MAPP and MAPE were crucial for achieving satisfying mechanical properties in the polyolefins-WPC but were also one of the main contributors to the environmental parameters of the semi-finished compounds considering the little content (3 w-%) and the potential environmental impacts thereof. The literature review on LCA of WPC in Section 2.3 and the LCIA of ECOLIFE® exhibited the use of additional additives, which may also increase the potential environmental impacts resulting from WPC manufacturing (product stage). Results of ABS- and PS-WPC showed that the physical and mechanical properties were not significantly benefited from SMA as the coupling agent. This would lower the environmental parameters of WPC from secondary WEEE plastics in comparison to the polyolefins-WPC at much better physical and mechanical parameters as discussed in Section 6.2. Nevertheless, secondary ABS showed a high Cd content (59 ppm) which is likely to remain constantly high, as the REACH regulation does not specifically prohibit Cd for colorization of ABS. Using sorted, non-colorized secondary ABS could be feasible solution to that problem, but better sorting results in higher secondary material costs.

The energy$_{el}$ was an environmental hotspot of 1.82 m^2 ECOLIFE® as well, which is used for compounding wood, plastic and the additives to WPC pellets (9.4 MJ/FU) and for the co-extrusion processes (57 MJ/FU). In both product LCA studies, the energy$_{el}$ was modelled by a generic LCA dataset reflecting the country-specific energy$_{el}$ grid mix. At first glance, using energy$_{el}$ from renewable resources could improve the environmental profile. The viability of utilization need to be studied in detail on a regional basis in context of land use competition, indirect land use effects and biodiversity issues (Alexandratos & Bruinsma 2012; Lauri et al. 2014; Searchinger 2013; Searchinger et al. 2015) as well as logistics and storage challenges of renewable energies (Brinker 2011).

The transport of raw and semi-finished materials from Germany to China was not influencing the overall environmental parameters of ECOLIFE®, but specifically it was the environmental hotspot in the supply chain of the specialized wood powder from industrial roundwood. Using co-products from regional suppliers should be considered as an alternative. Co-products derived from wood sawing or planing are not classified as waste, when generating revenues. A sensitivity analysis in Table 17 exhibits the comparison of the LCIA for 1 kg of alternative wood material sources. WPC based on wood co-products are very likely to result in same physical and mechanical properties as specialized wood powder from industrial roundwood or the like. The environmental burdens of processes generating co-products were allocated based on economical values. The environmental parameters of secondary wood from mixed waste wood (A II) and co-products were respectively considered to the environmental parameters of virgin wood particles as the ecological reference.

Significant improvements in GWP, EP, AP, ADPE and ADPF can be achieved by substituting wood co-products in WPC. Comparing co-products to waste wood, the savings in GWP, ADPE, and POCP become less important whereas the regional environmental parameters EP and AP were greatly affected.

Table 17. LCIA (cradle-to-gate) comparison of 1 kg wood particle (< 1 mm) acc. to EN 16485; virgin wood particles from Norway spruce, co-products from sawmill processes and secondary wood from mixed waste wood (A II). x… inside system boundary

Wood particles from…	**System boundary**			**A1–A3**						
	A1	**A2**	**A3**	*GWP*	*ODP*	*EP*	*AP*	*POCP*	*ADPE*	*ADPF*
Norway spruce	x	x	–	1	1	1	1	1	1	1
Co-products	x	x	x	-61%	–[2]	-12%	-32%	0%	-74%	-56%
Waste wood	*[1]	x	x	-79%	-80%	-71%	-71%	+3%	-80%	-81%

[1] wood inherent properties only
[2] not assessed due to inconsistent data for energy provision of nuclear energy in upstream processes (Diederichs 2014)

The CML-IA (CML 2001, Apr. 2015) contains the abiotic depletion potential for non-fossil resources (ADPE) expressed in antimony equivalents (kg Sb-eq.) and abiotic depletion potential for fossil resources (ADPF) expressed in mega joule (MJ). Development of indicators expressing resource efficiency is currently a hot topic in policymaking and in research (eurostat 2016; Huysman et al. 2015). For example, Fritz (2014) studied existing indicators to develop an evaluation of resource efficiency in terms of sustainable building assessments for which the LCA results can be further transferred for evaluation purposes in the application context such as the building context and certification schemes like the German "Sustainable and green building" system DGNB. However, the ADPs are not (yet) reflected in the "Guideline for Sustainable Building" (BMUB 2016).

6.3.2 Identification of an environmentally sound end-of-life treatment

Recycling of post-consumer WPC to yield secondary WPC material was ecologically beneficial compared to incineration with energy recovery, based on the basket-of-benefits LCA methodology. This was observed for all WPC formulations that determined further the NCV in context of energy recovery potential and the provision of complementary $energy_{th}$ and $energy_{el}$.

The environmental profile of WPC recycling was sensitive to the amount of virgin WPC in the formula and thereof the virgin HDPE. Therefore, a sensitivity analysis was conducted with higher virgin WPC content in the recycled WPC formula. If the share of virgin WPC in the composite formula is increased to 50%, the environmental profile of WPC recycling almost equals the incineration system in the environmental parameters GWP (-6%), AP (-6%) and becomes even worse in EP (+7%), ADPF (+10%) and ADPE (+31%). As a result, if virgin WPC material is needed or used in the secondary WPC formula the use of secondary plastics should be considered.

Despite of the technically feasible recycling of WPC based on laboratory scale results (iVTH 2015a, 2015b, 2015c; Schirp & Hellmann 2013), a current challenge of recycling is the increasing market share of composites. The German Waste Wood Directive (German Government, 2003) regulates recycling of waste wood of composites with more than 50% wood. Quality-labelled German WPC decking contain > 50% wood content and could be classified as waste wood in accordance with AltholzV (German Government 2003). However, this act was released in 2003, where WPC decking were either in an early market state and not considered as an engineered wood product. Therefore, post-consumer WPC is often wrongly sorted as waste wood or WPC is collected through bulky waste systems. In both ways, WPC is treated as a residual and incinerated (EUWID 2015; Lampel 2015), which was found to be the least preferable EoL treatment from an environmental point of view as discussed before.

Crucial for WPC recycling activities in context of a circular economy is an established market (EC 2008), characterized by demand and supply of the secondary material. This market exists to some extent for post-consumer plastics and wood, but up until now not for post-consumer WPC. Although some manufacturers claim to provide a take-back system for post-consumer WPC decking, a google search resulted in only one result. In this case, the discarded WPC has to be individually transported to the manufacturer who takes back the post-consumer WPC on a voluntary basis (NATURinFORM 2016). A system wide take-back system for WPC is lacking so that the most dominant disposal route of WPC is incineration in Germany and most probably landfill in other European countries, where almost 50% of post-consumer plastics are still landfilled on average (BIO et al. 2011; Plastics Europe 2015). The financial subsidies for using secondary materials addressed in the "Roadmap to a Resource Efficient Europe" (EC 2011b) will therefore likely play an essential part for a viable bioeconomy in terms of a projected growing market share of WPC products and a resulting amount of post-consumer WPC in our society.

7 Conclusions

Coniferous roundwood and wood co-products thereof are likely to become a vital resource in the bioeconomy. This study examined the potential use of secondary wood as material substitution in WPC. Material flow analysis showed post-consumer waste wood of category A II (mixed category A II) and post-industrial particleboard (A II) to contain substitutable secondary wood particles. Both secondary materials essentially consist of softwoods (spruce and pine) and their use in WPC resulted in physical and mechanical properties comparable to WPC made of virgin Norway spruce or co-products thereof. However, both secondary wood materials are likely to contain abrasive materials, plastics and hazardous inorganic content, linked to insufficient sorting and recycling techniques due to economic reasons. Abrasive materials can easily damage WPC processing machinery and plastics may influence the processability of the polymer matrix. There is no noticeable ecological advantage to using either virgin or secondary wood in WPC, as both materials consume almost the same amount of energy to process into a workable particle size.

Secondary thermoplastic polyolefins (HDPE, PP) from recycled post-consumer packaging waste and WEEE (ABS, PS) are a practical alternative for WPC, especially for ecological and economic reasons. These secondary materials are readily available on the recycling market as high quality re-granulates, or at least have been sorted and recycled with accompanying data sheets. Furthermore, plastic recyclers continue to search for new value-added applications. From a technical perspective, secondary plastics present a feasible option for WPC due to their excellent mechanical and physical performances. Though secondary ABS and PS are more expensive than secondary polyolefins, WEEE-WPC do not necessarily need a coupling agent (SMA) and result in much higher technical performance than polyolefins-WPC. This would reduce the total production costs and benefit the environmental profile of the composites accordingly. The challenges presented by using secondary materials, such as the presence of inorganic hazardous substances in pigments or dark colour hues of WPC made from these secondary materials, could be avoided by co-extrusion with an outer layer from virgin materials. An innovative window system made of a co-extruded WPC window profile presented in the case study already uses secondary HDPE that is wrapped in aluminium sheets. An alternative should be considered to aluminium as its raw material supply chain causes significant environmental impacts. Future studies, however, will need to examine the assessment of organic hazardous materials as well and consider more LCA indicators with respect to resource efficiency.

Results from the "basket-of-benefits" LCA showed that recycling of WPC to yield secondary WPC is the preferable EoL alternative compared to incineration with energy recovery in waste-to-energy plants. In practice, WPC is an impurity either in waste wood recycling or in bulky waste recycling due to a non-existing demand for secondary WPC material and, therefore, rejected as a residual that is disposed in incineration plants. A vital take-back system for WPC is needed what could be achieved by better implementation of policy concepts, such as the "polluter pays principle", the EPR and especially the Waste Framework Directive 2008/98/EC, which offer great potentials to become a meaningful resource efficient bioeconomy.

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