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Dissertation

Environmental Assessment of New Electrical and Electronic Appliances and Energy Supply Systems

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Affidavit

I declare in lieu of oath, that I wrote this thesis and performed the associated research myself, using only literature cited in this volume. If text passages from sources are used literally, they are marked as such.

I confirm that this work is original and has not been submitted elsewhere for any examination, nor is it currently under consideration for a thesis elsewhere.

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(Date) (Signature)

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Abstract

Electrical and electronic products account for more than ten percent of the global industrial production. They are characterised by a high degree of diversification of materials and substances in their composition and dimensions from the macro to the nanoscale. A simultaneously cross-continent and highly segmented production, supply, and disposal chain leads to an almost unmanageable complexity. Another particularity compared to other goods is the consumption of electrical energy in the use phase, which is problematic for the climate and the availability of fossil resources, and which overshadows further environmental influences of other aspects and life cycle phases.

The work addresses the question on how electrical and electronic products can be assessed in their entirety for their environmental compatibility. It is investigated whether the life cycle assessment methodology and the availability of life cycle inventory data can provide sufficient answers to environmental questions, provide a basis for identifying future environmental problems of new technologies, and support the development of solution strategies.

The work investigates which aspects of the different life cycle phases significantly affect the environment, which interrelationships have been unnoticed, and which methodological modifications are necessary for a comprehensive environmental assessment. This also includes the future development of the market and associated new environmental pressures. On the basis of examples from the fields of new materials for electrical engineering, electronic devices and modules as well as systems for the production of electrical energy from renewable sources, it will be shown which further environmental aspects should be taken into consideration and how methodological approaches must be improved in order to identify regional and global ecological interactions on spatial and temporal level. Future developments will be outlined in order to derive methodological input for decision support for the design of devices and the overall ecological impact of the electrical and electronic market.

Kurzfassung

Elektrische und elektronische Produkte stellen mehr als zehn Prozent der globalen Industrieproduktion dar. Sie sind gekennzeichnet durch eine hohe Diversifizierung der Materialien und Inhaltsstoffe in ihrer Zusammensetzung und Dimension vom Makro- bis in den Nanobereich. Eine gleichzeitige Kontinente übergreifende, stark segmentierte Produktions-, Zuliefer- und Entsorgungskette führt zu einer nahezu unüberschaubaren Komplexität. Eine weitere Besonderheit gegenüber anderer Güter stellt der klima- und ressourcenproblematische Verbrauch an elektrischer Energie in der Gebrauchsphase dar, der eine weitere Hinterfragung von Umwelteinflüssen anderer Aspekte und Lebenszyklusphasen in den Hintergrund treten lässt.

Die Arbeit geht der Frage nach, wie elektrische und elektronische Produkte in ihrer Gesamtheit auf ihre Umweltverträglichkeit bewertet werden können. Es wird untersucht, ob die Methode der klassischen Ökobilanzierung und die Verfügbarkeit von Sachbilanzdaten eine ausreichende Beantwortung ökologischer Fragen, eine Grundlage für die Identifizierung von zukünftigen Umweltproblemen neuer Technologien bieten und das Entwickeln von Lösungsstrategien unterstützen kann.

Es wird genauer untersucht, welche Aspekte der verschiedenen Lebenszyklusphasen die Umwelt beeinflussen, welche Zusammenhänge unbeachtet geblieben sind und welche methodischen Modifizierungen für eine umfassende ökologische Bewertung vorgenommen werden müssen. Dies beinhaltet zudem die Entwicklung des Marktes und die zu erwartenden zukünftigen Umweltdrücke. Anhand von Beispielen aus den Bereichen neuer Materialien für die Elektrotechnik, elektronischer Geräte und Module sowie Systeme zur Gewinnung von elektrischer Energie aus erneuerbaren Quellen soll gezeigt werden, welche Umweltaspekte weitere Berücksichtigung finden und methodische Zugänge genauer hinterfragt werden müssen, um regionale und globale ökologische Wechselwirkungen auf räumlicher und zeitlicher Ebene zu erfassen. Es werden zukünftige Entwicklungen streiflichtartig dargestellt, um methodische Beiträge für die Entscheidungsfindung für das Design von Geräten und die ökologische Gesamtwirkung des elektrischen und elektronischen Marktes abzuleiten.

Abbreviations

%	Percent
€	Euro
+	Plus [math.]
<	Less than [math.]
>	Greater than [math.]
±	Plus/minus (tolerance, math.)
≙	Correspond to [math.]
°C	Degree Celsius
‰	Per thousand
a	Anno; year
AC	Alternating current
Al_2O_3	Aluminium oxide
A–LCA	Attributional life cycle assessment
ALO	Agricultural Land Occupation
ARD	Abiotic resource depletion
a–Si	Amorphous silicon
AT	Austria
bar	Bar [metric unit of pressure]
BE	Belgium
BG	Bulgaria
cap	Capita
CC	Climate change
CdSe	Cadmium selenide
CdTe	Cadmium telluride
CED	Cumulative energy demand
cf.	Confer [compare with]
CFL	Compact fluorescent lamp
CH	Switzerland
CIGS	Copper indium gallium diselenide
C–LCA	Consequential life cycle assessment
cm^2	Square centimetre
CMH	Ceramic metal halide (lamp)
CML	Centrum voor Milieukunde (University Leiden, NL)
CN	China
CO_2	Carbon dioxide
CO_2 eq	Carbon dioxide equivalent
CO_2–e	Carbon dioxide equivalent
CO_2–eq	Carbon dioxide equivalent
c–Si	Crystalline silicon
CY	Cyprus
CZ	Czech Republic
D^2PAK	Package code for SMD semiconductors
DB eq	Dichlorobenzene equivalent
DC	Direct current
DE	Germany
Ded-LED	Dedicated LED (lamp)
demonstr.	Demonstrator [Prototype that serves as demonstration and proof–of–concept model for a new technology]
DK	Denmark
DPSIR	Driving forces, Pressure, State, Impacts, Response
e.g.	Exempli gratia [for example]
EC	European Commission
EC PEF	European Commission product environmental footprint
EDIP	Environmental Design of Industrial Products [LCIA method]
EDP	Electronic data processing
EE	Estonia
EEE	Electrical and electronic equipment
EI99	Ecoindicator 99
el.	Electrical
El. energy	Electrical energy
E–LCA	Environmental life cycle assessment
E–LCC	Environmental life cycle costing
em.	Emission
e–mobility	Electro mobility
env.	Environmental
EPA	Environmental Protection Agency [US]
EPBT	Energy payback time
EPS	Environmental Priority Strategy
EPTA	European Parliamentary Technology Assessment network
eq.	Equipment
eq.	Equivalent
EROI	Energy return on energy invested
ES	Spain
et al.	Et alii [and others]
ETAG	European Technology Assessment Group
EU	European Union
EU27	27 member states of the European Union 2007-2013
EU28	28 member states of the European Union as of 2013 (incl. Croatia cf. EU27)
EU28+2	EU28 plus Norway and Switzerland
e-waste	Electric and electronic waste
FAETP	Freshwater aquatic ecotoxicity potential
ff.	And the following
FI	Finland
FL, FLL	(Long) Fluorescent lamp
fossil–C	Fossil carbon
FR	France
FR4	Flame retardant 4 [composite material composed of woven fiberglass cloth with an epoxy resin binder that is flame resistant]

g	Gramme
GHG	Greenhouse gas
GHI	Global horizontal irradiance
GLO	Global
GR	Greece
GW_p	Gigawatt peak (peak: nominal power)
GWh	Gigawatt hour
GWP	Global Warming Potential
GWP 100	Global Warming Potential to a time scale of 100 years
h	Hour
ha	Hectare = 10,000 m2
HEC	Hydroxyethylcellulose
HID	High-intensity discharge lamp
HPS	High–pressure sodium lamp
HR	Croatia
HT	Human toxicity
HTP	Human toxicity potential
HU	Hungary
HWL	Hazardous waste landfilled
i.a.	Inter alia
i.e.	Id est [that is, in other words]
I/O	Input/output [LCA method]
ICT	Information and communication technology
IE	Ireland
IGBT	Insulated-gate bipolar transistor
ILCD	International Reference Life Cycle Data System
IMS	Insulated metal substrates
Int-LED	Integrated LED (lamp)
ISO	International Organisation for Standardization
IT	Italy
IT	Information technology
IT & T	Information technology & telecommunication
ITA	Institute of Technology Assessment (Austria)
ITAS	Institute for Technology Assessment and Systems Analysis (Germany)
IUP	Intermediate unit process
K	Kelvin (colour temperature of lamps)
k	Kilo
kg	Kilogramme
kg/E	kg/EinwohnerIn (kg/capita)
km	Kilometre
km^2	Square kilometre
kW	Kilowatt
kWh	Kilowatt hour
kWp	Kilowatt peak (peak: nominal power)
l	Litre [unit of volume]
LCA	Life cycle assessment
LCC	Life cycle costing
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LCM	Life Cycle Management
LCP	Liquid crystal polymer
LCSA	Life cycle sustainability assessment
LED	Light emitting diode
LFL	Long fluorescent lamp
LIME	Japanese life cycle impact assessment method based on endpoint modeling
lm	Lumen (unit), the SI unit of luminous flux
LT	Lithuania
LTCC	Low temperature co–fired ceramics
LU	Luxembourg
LUCAS	LCIA method Used for a CAnadian-Specific context
LV	Latvia
m	Metre
m^2	Squaremetre
m^2.a	Squaremeter and year
math.	Mathematics
MEEuP	Methodology study for Ecodesign of Energy-using Products
MFA	Material flow analysis
MH	Metal halide (lamp)
min	Minutes
MIPS	Material input per unit of service
mix	Country specific electrical energy mix (also referred as "electricity mix")
MJ	Megajoule
µm	Micrometre
mm	Millimetre
MOSFET	Metal-oxide-semiconductor field-effect transistor
MT	Malta
MW	Megawatt
MW_p	Megawatt peak (peak: nominal power)
n/a	Not available; not applicable
NACE	Nomenclature statistique des activités économiques dans la Communauté européenne (Statistical classification of economic activities in the European Community)
NL	The Netherlands
n-layer	n-type semiconductor (n-negativ)
NREL	U.S. National Renewable Energy Laboratory
NUTS	Nomenclature des unités territoriales statistiques [European Union]
OECD	Organisation for Economic Co-operation and Development

ÖNACE	Österreichische Systematik der Wirtschaftstätigkeiten (Austrian version of NACE)
OTA	Office of Technology Assessment [US.]
PA	Polyamide
Pb	Lead
PBT	Polybutylene terephthalate
PCB	Printed circuit board
PCC	Pollution Control Cost assessment
PE	Polyester
PET	Polyethylene terephthalate
pH	Scale used to specify how acidic or basic a water-based solution is
pH11	Basic solutions
pH3	Acidic solutions
PH7	Neither acidic nor basic solution
PL	Poland
p-layer	p-type semiconductor (p-positive)
PMMA	Polymethyl methacrylate
PP	Polypropylene
ppm	Parts per million
PT	Portugal
PTFE	Polytetrafluorethylen [Teflon]
PV	Photovoltaic
PVA–S	Polyvinyl acetate
PWB	Printed wiring board = printed circuit board
ReCiPe	Life cycle impact assessment method; provides a 'recipe' to calculate life cycle impact category indicators; the acronym also represents the initials of the institutes that were the main contributors to this project and the major collaborators in its design: RIVM and Radboud University, CML, and PR Consultants
REE	Rare earth element
REO	Rare earth oxide
RER	Europe
RO	Romania
RoHS	Restriction of the use of certain hazardous substances [in electrical and electronic equipment]
RoW	Rest of the World
RWL	Radioactive waste landfilled
S	Sum [math.]
SE	Sweden
SETAC	Society of Environmental Toxicology and Chemistry
SI	Slovenia
SK	Slovakia
SLCA	Social life cycle assessment
SLCA	Streamlined life cycle assessment
S–LCA	Social and socio–economic life cycle assessment
SMD	Surface mount device
SMT	Surface mount technology
SPI	Sustainable Process Index
SPOLD	Society for Promotion of LCA Development
STAN	Substance flow analysis [Free software]
STOA	Scientific and Technological Options Assessment
t	Tonne [unit of mass]
T5	Type of fluorescent lamp; diameter: 15.9 mm
TA	Technology Assessment
TAB	Office of Technology Assessment at the German Bundestag
TE	Terrestrial ecotoxicity
TJ	Terajoule
tkm	Tonne-kilometre [unit of transportation measurement]
TRACI	Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts [Impact Assessment methodology]
TV	Television
TWh	Tera-Watt hours
U.S.	United States [of America]
UCTE	Union for the Co-ordination of Transmission of Electricity (Europe)
UK	United Kingdom
UNEP	United Nations Environment Programme
US	United States
USA	United States of America
UV	Ultraviolet
W	Watt
WEEE	Waste electrical and electronic equipment
WE–LCA	Working environment life cycle assessment
Wp	Watt peak (peak: nominal power)
wt%	Weight percent [percentage by mass]
x	Multiplied by [math.]

Table of Contents

1. Introduction

From the perspective of electrotechnical engineering the broad discipline of *environmental engineering* is covered by various scientific research fields. They can be divided into the following categories:

- *Environmental sensors:* detection and monitoring of pollutants
- *Control engineering*: management of energy, material, and pollutants flows
- *Holistic life cycle assessment*: impact assessment for manufacturing, use, and end–of–life of products and their framework conditions
- *Eco–design:* development of energy efficient systems, environmental friendly materials, disassembility, and long lifetime
- *Renewable energy sources*: sustainable electrical energy generation
- *Technology assessment*: analysis and trends for the development of technology

Besides the traditional electrotechnical disciplines dealing with environmental technology, such as measuring techniques, control engineering, sensor technology, and energy efficient circuit design, the interdisciplinary environmental investigation of the electrical and electronic products themselves, along with the full life cycle, is a key factor for environmental engineering. This subject area is included in the *holistic life cycle assessment* which is the scope of this work.

The evaluation of global environmental impacts of the electrical industry calls for a holistic view. The investigation of international framework conditions is also required, as well as regional realities.

Electrical and electronic devices consume electrical energy and its generation represents a large portion of the environmental impact in the product life cycle. However, there are further environmental relevant factors to be considered, like factors arising during manufacturing, the end–of–life of the product, and the consumer behaviour.

A comprehensive environmental assessment of an electrical device must be based on the concept of the full life cycle, from raw material acquisition, manufacturing, use phase to the end–of–life management. The method of life cycle assessment (LCA) has become a key concept for environmental evaluations, based on complex models and different evaluation methods with different weightings of the environmental impact categories, such as global warming potential, human toxicity, or eco–toxicity.

There is a number of difficulties in carrying out an environmental assessment of electric and electronic products. Electronic devices consist of a large variety of components and materials with a large value chain of manufacturing processes and locations and dispersed supply chains. Therefore, a complete environmental evaluation with a single life cycle assessment study is almost impossible to perform. Compared to more homogeneous materials such as building materials or agricultural products, electronic devices can only be assessed by applying a number of simplifications, generalisations, and omissions. A comprehensive environmental assessment of electronic products needs, besides traditional life cycle assessment case studies, further information such as consumer behaviour, spatial distribution of products or the effects of the manufacturing energy consumption ("grey energy"), which consider a larger global context.

Current European research focuses on the reduction of the energy consumption of individual electrical devices. Nevertheless, in many cases the total energy balance shows a rebound effect. This means that actions to increase the energy efficiency do not achieve their objectives, but in the worst case result in a substantial increase of the total electrical energy consumption. This phenomenon cannot be detected by the attributional approach of a life cycle assessment study.

The manufacturing and waste management standards, as well as environmental regulations and the ban of hazardous substances, are different on each continent or even between countries. This leads to location dependent environmental pollutions, which are not sufficiently captured by the Europe focussed life cycle inventory databases for electronic products.

The sector of the electrical industry and electronics for automotive application holds a share of about one–sixth of the total global industrial production. Therefore, besides the electrical energy consumption, electrical and electronic equipment represent in the form of material consumption, manufacturing processes, and electronic waste generation, a significant environmental and resource factor. Due to various sophisticated designs of electronic devices and the continuous change of technologies, there is a large research gap in the environmental assessment of new materials and products.

2. Objectives and Organisation of the Research

The environmental assessment of electrical and electronic products is perceived in public mainly by the electrical energy consumption in the use phase, the content of hazardous substances, and a required high recycling rate of valuable materials. In the field of electrical energy generation, the focus is on the reduction of carbon dioxide emissions and utilisation of fossil fuels as well as the expansion of energy generation based on renewable energy sources. These mission–oriented objectives are reflected in a broad European research landscape, which, however, ignores many further environmental pressures of electrical and electronic products in their life cycle.

2.1. Problem Definition

The electronics sector is characterised by a highly diversified variety of materials, components, devices, and their relating fabrication processes. The fabrication is associated with a considerable amount of highly pure materials with their need for sophisticated production processes. The electrical sector is of a fast evolving nature of technological processes and it has one of the largest globally distributed production networks and supply chains. It concerns all life cycle stages, ranging from the manufacturing phase, going to the use phase, and ending with the electrical and electronic waste (e–waste) management. It includes all continents with their different contributions and framework conditions in terms of material and energy supply, quality, working conditions, environmental standards, and electronic waste management. The end–of–life treatment of electronic waste is becoming an ever greater global problem, as a considerable amount of waste is transferred from Europe and North America to Africa and Asia, where unregulated informal waste treatment harms the workers and the environment. There are manifold environmental aspects, which have to be considered on a case-by-case basis.

The international climate negotiations, resource scarcity and European resource dependency, and increasing waste problems are the drivers for technology development of electric products and their circular economy. The European legal regulations that followed particularly concern the electrical energy consumption during the use phase and the end–of–life phase of the life cycle, omitting the importance of the other upstream life cycle stages and varying global as well as local boundary conditions. Beyond that, the material composition, the manufacturing expenses, the market development, and the consumer behaviour significantly influence the environmental impact of electrical and electronic products. Although there is a lot of literature on single case environmental studies for specific product groups, especially the environmental performance of new technologies is not yet fully explored.

The reduction of the specific energy consumption of an electrical product is not a sustainability criterion, as it does not take into account the quantity of electrical products circulating in parallel, the intensity of use and the manufacturing or disposal costs. A holistic life cycle assessment shows actual environmental impacts and the possibility of new strategy developments for sustainable economic activities.

This work is based on the following theses: Firstly, the availability of data for the environmental evaluation of highly sophisticated single processes of new electrical products as well as of the entire electrical sector is insufficient. Secondly, there is no methodical assessment procedure available that comprehensively addresses the specificities of the electrical sector.

2.2. Objectives of this Thesis

The aim of the thesis is to analyse the electrical and electronic sector from a holistic environmental perspective. New environmental assessment results shall give a better understanding of the environmental performance of the sector and the results shall further indicate existing research gaps concerning new electrical and electronic equipment and energy systems based on renewable energy sources. Further objectives are to identify appropriate methodological approaches to fully describe the environmental impact of electrical systems and which environmental challenges are facing the future.

2.3. Research Questions

There are three key research questions to be answered in this work:

1. What is the dimension of the global electrical sector, what kind of markets and market segments are defined and which material flows result from this?
2. Which methodological approaches are required and applicable for describing the environmental performance of the electrical sector over the entire life cycle on a spatial and temporal level?
   - Which functional units are suitable?
   - Which system boundaries must be taken into consideration?
   - Which assessment methods can be applied and/or have to be extended?
3. What future challenges will arise for society by the high dynamics of the electrical sector?
   - How is the volume, the material composition and the spatial distribution of electrical waste changing?
   - What impact does the growing market for renewable energy systems have?
   - How do data uncertainties influence future environmental pressures?

2.4. Organisation of the Research

The investigated materials and devices are chosen in the scope of research projects and existing infrastructure at the TU Wien in which material and process data were accessible. In detail, the research subjects of the case studies are selected from the following electrical related categories:

- Materials (subject: low temperature co–fired ceramics – LTCC)
- Electronic components (subjects: power modules for automotive application)
- Electronic devices (subjects: rear camera systems, LED lamps)
- Electrical energy generation (subjects: photovoltaic, hydro power, and biogas plants)

The thesis initially introduces in Chapter 3 to the historical development of material flow analysis, technology assessment, impact assessment approaches, and life cycle assessment, followed by the state of the art of environmental studies for electrical products. Thereafter, the work is divided into two parts. The first part deals with market–relevant questions and methodological approaches to investigate electrical systems by holistic environmental evaluation approaches apart from classical life cycle assessments. This includes topics such as market development, material flow of hazardous substances, regional energy balance, rebound effect, and grey energy, uncertainty of technology and energy predictions, end–of–life considerations, and well–being of the end–user. Chapters 4 to 7 include newly conducted case studies described as follows.

Chapter 4 presents a new comprehensive review of the global electrical market including its classifications and sectors, manufacturing countries, material fractions, and e–waste generation.

Chapter 5 deals with the market development of photovoltaic implementation in Europe. The study addresses the relations of urban and rural PV expansion and energy production on local level as well as hazardous material flows and future environmental pressures of PV modules.

Chapter 6 investigates roof–top photovoltaic plant expansion of a small town and the related material turnover. The questions of regional energy balance and hazardous material flows are investigated.

Chapter 7 includes the topic of LED lighting and related environmental considerations on global market expansion, implications on users, and future environmental pressures related to end–of–life treatment. The study addresses material and technical data uncertainties and material flows of valuable substances for LED–lamps. Further, local energy balance and rebound effect are investigated associated to the consumer behaviour.

The second part deals with the application of different methodological approaches of standardised life cycle assessments. Chapter 8 gives a comprehensive introduction to the methods and standards for life cycle assessment from the viewpoint of electrical engineering followed by a critical discussion of diverse aspects of the applicability on electronic products. The subsequent chapters 9 to 14 include different newly conducted case studies addressing midpoint and highly aggregated life cycle assessment indicators, utilisation of different functional units, consequences of simplifications and data uncertainties, applicability of laboratory data and local infrastructure projects, and further data requirements.

Chapter 9 includes midpoint assessment methods and material turnover of three different commercial low–temperature co–fired ceramic (LTCC) substrates under laboratory conditions. The study addresses the selection of an appropriate functional unit, the availability of life cycle inventory data, and the transferability of laboratory data to industrial processes.

Chapter 10 presents a comprehensive review and comparison of published life cycle assessment studies for LED lamps. The study addresses the utilisation of different functional units, environmental parameters of rare earth oxides as well as simplifications, omissions, and uncertainties of LCI data and their consequences.

Chapter 11 deals with the environmental evaluation of energy systems and addresses different findings of highly aggregated and midpoint assessment methods and their application to local infrastructure projects. The Sustainable Process Index (SPI) is compared with the Carbon Footprint and the Agricultural Land Occupation, and the respective data situation and applicability are discussed.

Chapter 12 determines the Sustainable Process Index of a new design for an electronic power discrete package fabricated in embedding technology compared to a commercial power component for e–bike application. The study addresses the application of highly aggregated assessment methods as a tool in the product design phase and investigates the environmental trends for the fabrication of miniaturised electronic components.

Chapter 13 investigates the environmental performance of two different newly developed power modules for e–mobility application conducted by the Sustainable Process Index. The study addresses the application of highly aggregated life cycle assessment methods to identify environmental hot spots in the product design phase, to evaluate further data requirements, and to show optimisation options.

Chapter 14 deals with the comparison of a commercially available and a newly developed rear camera system for automotive application conducted by the life cycle assessment method ReCiPe. The study addresses the application of midpoint life cycle assessment methods and discusses the different results of the indicators Climate Change, Human Toxicity, and Ecotoxicity as well as consequences of process data uncertainties.

The final Chapter 15 provides a general conclusion of the findings of the case studies and summarises the answers to the research questions of the thesis.

3. State of the Art

In the 1960s, a broad international discussion on sustainable development began. Three schools of thought of environmental evaluation methods emerged that evolved in parallel. The different methods influenced each other, which led to new methodological approaches through the decades. The following sections present the historical development of material flow analysis (MFA), technology assessment (TA), impact assessment approaches, and life cycle assessment (LCA). Subsequently, the state of the art of environmental evaluation of new electrical and electronic products is presented. The three methodological approaches are essential for this work, as MFA is the basis of the strategic assessment of resource availability and the waste situation, LCA is applied as a method for environmental evaluation and optimisation of individual products, and both together contribute to the technology assessment of the development of an industrial sector.

3.1. Development of Material Flow Analysis (MFA)

The history of MFA dates far back, where the basic principle of mass balances in terms of "input equals output" had already been recognized. In the 18th century, the French chemist Lavoisier experimentally proved that the mass of matter cannot be altered by chemical processes. In the 20th century, the method of MFA was applied at different times and different places especially in chemistry. In the 1930s, the American economist of Russian origin Wassily W. Leontief created input-output tables, which were later developed to solve economic problems. The tables allowed systematically quantifying the interrelationship between goods, demand, production processes, and deliveries. (Brunner & Rechberger, 2017)

In the 1970s, MFA was first applied to environmental management and resource conservation studies. Two fields of application developed, metabolism of cities and regional pollutant emissions. In the following decades, MFA evolved towards solid waste streams, wastewater, material recycling and application for LCA. (Brunner & Rechberger, 2017)

The *metabolism of cities* was first described in 1965 by Abel Wolman. In a study input and output flows per capita of a hypothetical city with one million inhabitants were calculated by means of available average production and consumption data of goods. (Wolman, 1965).

Another study from 1975 deals with the urban metabolism of the city of Brussels in Belgium. A detailed total balance of imports and exports was compiled including natural energy, pollutants, water, goods and waste. The results show i.a. that the entire energy need of the city is provided by the hinterland, although theoretically it could be covered by the natural energy input by the sun (Duvigneaud & Denayer-De Smet, 1975). In 1991, the MFA was expanded by introducing the concepts of activities and the *metabolism of the anthroposphere*. The aim is to analyse and control the dynamics of regional material flows to describe the metabolism of complex biological and cultural systems for resource and waste management of environmental impact studies. The results from the MFA complement the findings of other disciplines such as environmental sciences, e.g. ecotoxicity, life sciences, e.g. human toxicity, or social sciences. MFA aims to deliver complete and consistent information about all flows and stocks of particular materials within a defined system in space and time. (Baccini & Brunner, 1991).

In 2001, a Swedish study was published that quantified diffuse emissions during the use of heavy metal containing goods in Stockholm. The type and mass of goods containing heavy metals and their emission potential were determined, excluding road-wear. The total stock of copper-containing goods was approximately 123,000 tonnes, of which 70% was accounted for electrical and electronic equipment. In 1995, the total amount of lead-containing goods was 51,000 tonnes, of which 30,000 tonnes were accounted for power and telecommunication cable shielding and 500 tonnes for electronic equipment. (Sörme et al., 2001)

The method of MFA extended over the years by, for example, new social science approaches. The new term *colonization* was coined, which describes the treatment of nature by human societies to render better exploitability (Fischer-Kowalski & Haberl, 1993).

In 2008, the OECD published a 4-volume guidance for measuring material flows and resource productivity. The first Volume describes approaches and measurement tools for MFA. The emphasis is on national level and on areas in which practicable indicators can be defined. The second Volume includes a theoretical and technical description of methodologies and concepts for accounting material flows. The third Volume includes activities related to the measuring and analysing of natural resources and material flows. It describes the main characteristics and the extent of activities for environmental decision making and reporting. The fourth Volume includes a practical guidance to implement national material flow accounts (OECD, 2008). The OECD is continuously developing and extending sustainability indicators, such as the *Green Growth Indicators* based on MFA. The aim is to rise productivity for economic growth and higher living standards while keeping low e.g. land consumption, air pollution, and CO_2 emissions. Currently, ten economic and environmental indicators are incorporated and correlated (OECD, 2011/2017).

A book published by Brunner and Rechberger includes a comprehensive presentation of the methods for MFA from the basics to the evaluation of material balances of high complex systems and a number of case studies. The standardised, precise methodology and the incorporation of new developments such as treatment of data uncertainties is a tool to generate new data and increase knowledge in the fields of waste and resource management, industrial ecology and environmental protection. The results of MFA can be applied to evaluation methods such as *Material Intensity on Service Unit* (MIPS), *Swiss Ecopoints, Sustainable Process Index* (SPI), *Life Cycle Assessment* (LCA), or *Statistical Entropy Analysis* (SEA) (Brunner & Rechberger, 2017; Cencic & Frühwirth, 2018). Parallel to the further development of MFA, the software tool *STAN* was developed at the TU Wien. The freeware helps to carry out material flow analysis according to the Austrian standard ÖNorm S 2096 (Material flow analysis - Application in waste management) (STAN, 2019).

3.2. Development of Technology Assessment (TA)

The formation of technology assessment is based on three constitutive elements: (1) Crisis of the technical progress optimism, that is, the new is automatically also the better; (2) The need to make decisions in increasingly complex modern societies, in some cases with high levels of uncertainty; (3) A democratic claim for a formative role in the handling of scientific and technical progress; this created a divergence between the perspectives of the decision makers and those affected. Thus, in the formation of the technology assessment and in the context of the technical progress to which it must react, there are quite heterogeneous elements that make it difficult to give a clear role definition. (Friedl & Mihaliy, 2019)

The idea of technology assessment arose in the U.S. as a tool for parliamentary policy advice. In 1967, the term was first used by the Democratic Representative Emilio Q. Daddario in the Subcommittee on Science, Research, and Development of the House Science and Astronautics Committee of the United States Congress. Problems related to the development and use of technology should be evaluated through a technology assessment. Short- and long-term societal, economic, ethical and legal consequences of the use of technology should be explored. The aim was to provide information for policy makers on policy alternatives. The TA's very broad field of expertise includes not only the evaluation, but also the diffusion of technology and technology transfer. Furthermore, TA should investigate the role of technology and society and the identification of factors that lead to a faster acceptance of new technologies. (Banta, 2009; Wong, 2015)

Years of work in the Subcommittee of the Congress led to the founding of the U.S. Congress Office of Technology Assessment (OTA) in 1972. The foundation was not without controversy within the Congress, as it would result in government intervention in industrial innovation activities. Already in the 1980s arose the first public criticism on the TA agency, that it would be an unnecessary instance. By the help of a Republican majority in the Congress, in 1995, the funding for OTA was withdrawn and the Agency was closed. At this time OTA had 143 staff members and an annual budget of $ 21.9 million. (Banta, 2009)

In the 1970s, OTA's working methodology was that, at the request of Congress Committees, compiled TAs and technology-related policy reports were prepared related to drafting laws. The reviewed documents included contributions from stakeholders from academia, industry, consumers, and the public and private sector. No recommendations were made, but a compiled presentation of different views and policy options, including their costs and consequences was submitted to the clients. (Wong, 2014)

Also in the 1970s, technology assessment reached Europe. A number of conceptual, methodological and institutional types of TA were established. Since the 1990s, the originally expert-oriented TA has been extended to participatory procedures to avoid conflicts later on and to give concerned parties a voice, which are usually not involved. These methods are known as *participatory technology assessment* and *constructive technology assessment* (ITA, 2019a).

In the 1980s, TA institutions were set up in five European countries, Denmark, France, Germany, The Netherlands, and UK with different conceptual and organizational models. The European Parliament established TA agencies which were modelled after OTA. In the 1990s these agencies formed the European Parliamentary Technology Assessment network (EPTA). Vig and Paschen investigated in detail the different developments of these national European TA agencies on issues such as why did different countries adopt different concepts of TA, how do different methodologies affect the outcomes of TA, and how might TA develop in the future. The different cultures and governmental functions of the countries can lead to substantial methodological and conceptual innovation and a better insight in the potential for transnational social learning and institutional transfer. (Vig & Paschen, 2000)

The Scientific and Technological Options Assessment (STOA) is an institution of the European Parliament. Until 2004, reports were made by temporary or external staff, resulting in a work not going deep enough and without uniform profiling. Since 2005 the reports are made by the European network of parliamentary TA, the European Technology Assessment Group (ETAG). In 1990, the Office of Technology Assessment at the German Bundestag (TAB) was established in Germany. Since 1995 the office is located at the Institute for Technology Assessment and Systems Analysis (ITAS) at the Karlsruhe Institute of Technology. In Austria, the Institute of Technology Assessment (ITA), founded in 1994 as a follow-up to the Research Center for Technology Assessment founded in 1988, is part of the Austrian Academy of Sciences. The interdisciplinary technology assessment of ITA is based on three aims: (1) to understand the complex interplay between technology and society from multiple perspectives; (2) to concomitantly analyse technology development, and; (3) to contribute to socially responsible technology policy by advising policy-makers and society. (Friedl & Mihaliy, 2019)

Currently, the principle methodological procedure to perform a TA is based on following steps: (1) Problem definition; (2) Description of the technology; (3) Prediction of future technology development; (4) Description of society and persons affected; (5) Prediction of social developments; (6) Identification, analysis and evaluation of consequence; (7) Analysis of political options; (8) Communication of the results in a generally accessible form. (ITA, 2019b)

Technology assessment has been accompanied by methodological challenges since its beginning. A key question is, at what time and at what stage of development a technology assessment should be involved. If a technique is already applied, its consequences can be studied well and confirmed with statistics. However, damaging effects can hardly be prevented or can only be reduced with considerable financial expenditure, respectively. Although it is easier to modify technology developments at an early stage, few social and environmental statements can be made about future applications. Other problems can arise when a technology is not related to a specific product. An example of this is nanotechnology, which extends the functionality of various product groups. However, the application of these product groups is very different. One strategy is to apply the method of *roadmapping* which involves in its investigations all current and future stakeholders and relates them to each other. The method can also be used to distinguish legitimate expectations from exaggerated hopes of new technologies. (Fiedeler et al., 2004)

Another discussion within technology assessment draws attention to the stage before the beginning of technology development, namely the education of engineers. Sotoudeh deals with the question of the extent to which technical education fits in with today's society. For two centuries, technical universities were expected to transfer engineering knowledge and thereby develop new technologies for industrial production. The challenge of the 21st century is to address socio-economic and environmental threats to sustainable development. Today`s engineers should be able to understand how to set new objectives, develop measures and assess the success or failure of sustainable development through their technology developments. A key requirement for technical universities is the development of a long-term perspective and the transfer phase of complex systems. (Sotoudeh, 2009 / 2010)

3.3. Development of Impact Assessment Approaches

The methods of material flow analysis were the first approach to measure environmental parameters of a region. However, they were unable to give a statement on the consequences of specific material flows on the environment. This circumstance called for the development of methods to identify the impact and define indicators for a measure of environmental impact.

The first methodological guide for LCA was published by Heijungs et al. in the frame of the development of the internationally agreed *Code of Practice* for performing a structured life cycle assessment. The report presents the definition and type of calculation of 13 independent *classification factors* which were the basis for the today still used *characterisation factors* of the CML method. This method was named after the Institute of Environmental Sciences of the University of Leiden (CML). The classification factors include for example biotic and abiotic depletion, greenhouse effect, human and ecotoxicological classification factors, nutrification, aquatic heat, noise, and others. The results then are compiled to an *evironmental profile* of the investigated product. (Heijungs et al., 1992)

In the same year as the *Code of Practice* was presented, in parallel, a project ran including a different methodological approach for evaluating a high aggregated indicator system for sustainable development. In 1993, a three-year EU-project was launched with the objective to enhance a set of indicators which can be used to measure and monitor progress towards sustainability and to assess the applicability of four sustainable indicators. The indicators included MIPS (*Material Input per unit of Service*), SPI (*Sustainable Process Index*), PCC (*Pollution Control Cost assessment*), and *Exergy*. As a prerequisite, it was assumed that sustainability can only be reached by decreasing the consumption of energy, materials, land, and other limited resources in different economic activities. Concurrently, the output of these activities that is emissions to air, water, and soil, and solid waste would be reduced. (EC, 1996)

The method of *Material Input per unit of Service* (MIPS) quantifies the mass flows of displaced materials that are moved in order to manufacture a product or provide a specific service. It does not differentiate between qualities and flows between ecosphere and anthroposphere. The pressure on the environment is assumed to be only the relocation of masses. It is a measure of "how much nature" is used (Schmidt-Bleek, 1993). MIPS was developed at the Wuppertal Institute to estimate the environmental impact potential of products and services. The aim is to develop longterm strategies for dematerialization and to reduce the so-called "ecological backpack" (Schmidt-Bleek, 1994).

The method of *Sustainable Process Index* (SPI) indicates the area which is needed to embed anthropogenic input and/or output flows of materials for providing a certain service sustainably into the ecosphere. The method is based on the idea, that the solar energy is the only external resource to the closed system of the earth (Narodoslawsky & Krotscheck, 1995). The method of *Pollution Control Cost* (PCC) assessment does not measure the pressure on the ecosphere, but economically quantifies the efforts to keep the pressure below a certain limit by avoiding the release of pollution into the ecosphere. The method of *Exergy* indicates the level of thermodynamic disequilibrium. That means it indicates the available work that is used for an activity. (EC, 1996)

In a number of case studies in den fields of transport, packaging, and geographical regions, these different impact assessment methods were applied to the same study object. The results showed that the four investigated indicators do not point in the same direction. All indicators confirm that the physical basis of resource use and production is the key for environmental sustainability. The economical indicators have similar trends as the SPI method. In some cases, MIPS showed different trends and therefore the indicators are rather complementary to each other. Due to the different assumptions and targets of the indicators, any common result means a higher level of robustness than the outcome based on one indicator alone. (EC, 1996).

A further new method is the determination of the *ecological footprint* based on MFA. It was recognised that the regions of affluent societies are very much dependent on the co-use of their hinterland and their ecological footprint must be reduced (Wackernagel et al., 2002). The World Resource Institute compared national flows of goods from different countries. Physical indicators of material flows were introduced that complement national economic indicators such as gross domestic product (GDP) (Matthews et al., 2002).

The decision for a tool to assess the environmental performance of a product or process should be made in awareness of the normative basis of the chosen measures and indicators, in other words, the chosen impact assessment method. Examples of three different normative concepts are the following: *Efficiency oriented measures* are based on the concept that using less input of resources leads to lower environmental impact for the same product or service. Efficiency can also be expressed by *Emergy*, which means to keep the quality loss of anthropogenic material and energy flows as low as possible for a time period as long as possible. *Problem oriented measures* are based on the normative concept of different, incomparable environmental problems such as global warming, human or water toxicity. For each problem a predominant cause-effect chain can be identified and the cumulative impact can be evaluated. *Sustainability based measures* are based on the normative concept of strong sustainability. Each anthropogenic activity is linked to a surface area on earth whose regenerative capacity is dependent on the only unlimited natural income of solar radiation and time. Consequently, all production systems compete for the limited area on earth. (Narodoslawsky & Shahzad, 2015)

Social objectives in the direction of sustainable development are initially formulated in general terms. The objectives must be made concrete and measurable in order to achieve these goals. *Indicators* are important tools that together with formulated environmental *objectives* and *actions* form an appropriate environmental governance system. An example of an indicator systems approach is the DPSIR. This system combines and correlates indicators of the categories *driving forces* (e.g., energy demand), *pressure* (e.g., air pollution), *state* (e.g., increased concentration of pollutants), *impacts* (e.g., impact on climate change), and *response* (e.g. new regulatory requirements). (Kaltschmitt & Schebeck, 2015)

There are basic differences between midpoint indicators that are applied for example in the CML and ReCiPe methods, and endpoint or high aggregated indicators, such as Ecoindicator`99 and SPI. Midpoint indicators are problem oriented and useful for monitoring and reporting of products already in use by one or a few numbers of characterisation factors. Typically for today is the monitoring of the characterisation factor *Global Warming Potential*. The disadvantage of midpoint indicators is the fact that they are not comparable among each other and therefore they are not applicable for weighting. High aggregated indicators can also be applied at an early stage of product development. The weighting decision of the indicator is made before the aggregation. Therefore it is much easier to identify materials, processes or activities that generate a high environmental pressure in the product development phase. (Narodoslawsky & Stoeglehner, 2010)

Technology decisions for sustainable development strongly depend on the value system of the individuals of the decision-making body. Technology assessment examines effects of technologies from an ecological, economic, and social perspective using qualitative and quantitative analysis methods. Life cycle assessment has been used as a quantitative standardised method to predict the expected environmental impact. However, the choice of the system boundary, the applied assessment method and the considered environmental impact indicators is dependent on the local specific value system in which the deciding individual person is situated. The objectives of the value system can be for example reducing green house gas emissions or reducing air pollution. Value decisions made by committees instead of individuals increase the *intersubjective* quality of the value system. (Sotoudeh & Narodoslawsky, 2018)

3.4. Development of Life Cycle Assessment (LCA)

Parallel to methodologies for Material Flow Analysis and Technology Assessment, the method of Life Cycle Assessment (LCA) has been developed and evolved. The history of life cycle assessment dates back to the 1960s. At the time, there were first scientific concerns about the limited availability of energy sources and raw materials. One of the first life cycle inventory calculations was conducted by Harold Smith, who evaluated the cumulated energy demand for the manufacturing of chemical products and presented them at the World Energy Conference in 1963 (Curran, 1994).

The first global simulation studies on the relationship between a changing world population and the demand for energy and raw materials were made in the late 1960s. The results were published in *The Limits to Growth* (Meadows, 1972) and *A Blueprint for Survival* (Goldsmith et al., 1972). The first predictions of future climate change led to further more accurate calculations of energy and resource consumption of industrial processes in both the US and Europe. After the oil crisis, until the 1980s, the environmental interest shifted from energy consumption to hazardous and household waste (Curran, 1994). In 1974, one of the first life cycle inventory studies was carried out for the U.S. Environmental Protection Agency EPA including beverage container alternatives (Hunt et al., 1974). In Europe, in 1973, a study on the contribution of degradable plastics was carried out to contribute to the solution of the waste problem (Battelle, 1973). Further LCA studies of this period were carried out by Gustav Sundström, Sweden, Ian Boustead, UK, and by EMPA St. Gallen, Switzerland (Klöpffer& Grahl, 2014).

In 1973, also the first Environmental Department was created in the European Commission and in 1981, a Directorate-General for the Environment was set up, which developed and carried out various successful environmental action programs parallel to further developments of the LCA methodology (Knill & Liefferink, 2007; Schön-Quinlivan, 2012). Since then, the EU has become an increasingly influential actor in environmental policy and legislation, not only within the EU but also globally, as chronologically documented in a review article (Selin & VanDeveer, 2015).

The first international organisation that recognised the potential value of developing a methodology for LCA was the Society of Environmental Toxicology and Chemistry – SETAC. As of 1990, a number of LCA workshops were held in the USA and Europe and one of the first outcomes was the LCA triangle, which was the basis of the still valid structure of a LCA. The triangle, extended in 1993, is stretched around the "Goal and Scope" with the three arms "Inventory Analysis", "Impact Assessment", and "Improvement Assessment". In 1990, the first LCA workshop was held at Smugglers Notch, Vermont, USA. In 1991, the SETAC Europe founding conference was held in UK at the University of Sheffield. In the same year, a two-day SETAC LCA workshop was held in the Netherlands at the University of Leiden, which was organised by the Centre of Environmental Science (CML) (Jensen & Postlethwaite, 2008).

In the course of a further meeting in Brussels, in 1992, the Society for Promotion of LCA Development (SPOLD), an association of industries, was created, which was independent of SETAC. SPOLD mobilised financial resources for the development of sound LCA methodology and developed the SPOLD format for data exchange. SPOLD supported the integration of LCA into a comprehensive "environmental toolbox" together with other methods such as Material Flow Analysis, Environmental Risk Assessment. In 2001, SPOLD terminated its activities (SPOLD, 2019).

SETAC is still the forum for interdisciplinary communication between environmental scientists, managers and engineers who are interested in environmental issues. The Society has two administrative offices, the one in Pensacola, Florida, USA and the other in Brussels, Belgium. SETAC publishes two scientific journals, Environmental Toxicology and Chemistry and Integrated Environmental Assessment and Management and convenes annual meetings around the world (SETAC, 2019).

The first scientific papers focusing on LCA appeared i. a. in Environmental Science & Technology, in the Journal of Cleaner Production, in the International Journal of LCA, in Resources, Conservation and Recycling, and in the Journal of Industrial Ecology. Life Cycle Sustainability Analysis (LCSA) is getting more attention which is reflected by a UNEP-SETAC working group, a subject section within the International Society for Industrial Ecology (ISIE), and an increasing number of scientific papers (Guinée, 2011). Since 1996, the best-known forum for scientists developing LCA and Life Cycle Management (LCM) methodologies is The International Journal of Life Cycle Assessment, which is devoted entirely to LCA and closely related methods (Springer, 2019).

The first few decades from 1970 to 1990 were the conceptual phase, with the main research focus on energy and resource consumption, emissions and waste generation. In the decade from 1990 to 2000, the main focus was on the international harmonization and standardisation of the various methods. During this period, the evaluation methods still used today were developed, namely the midpoint method CML 1992 as well as endpoint or damage approaches. Links to other disciplines were established and the idea of consequential LCA was developed and its effect on allocation methods. In the first decade of the 21st century, the United Nations Environment Program (UNEP) and SETAC launched the Life Cycle Initiative as an international partnership. The aim was to put *life cycle thinking* into political and social practice. (Guinée, 2012).

Since 1994, the International Organization for Standardization (ISO) has been involved in the discourse. This resulted into an ISO series of standards in 2006, namely the standards for life cycle assessment ISO 14040 and ISO 14044 (ISO, 2006a, ISO 2006b). Since the introduction of the standard, which merely sets a framework and does not define any methodological details, there is again an increasing methodological *divergence* observable relating to methods in terms of system boundaries, allocation methods or spatially differentiated LCAs. New is the life cycle costing (LCC) and the social life cycle assessment

(SLCA). The current discourse includes broadening the questions of an LCA towards the three pillars of sustainability, people, planet, and prosperity, but also including scarce resources and land or user behaviour (Guinée, 2012).

3.5. Selected Environmental Assessment Studies of Electrical and Electronic Products

In the broad field of electrical engineering a large number of environmental case studies for electrical components, products, and electrical energy generation have been conducted. They can be classified in studies published in scientific journals, e.g. (Int J Life Cycle Assess, 2019), studies by manufacturers, e.g. (Osram, 2009), preparatory studies for supporting governmental decision–making for national and international regulations, e.g. (Oeko–Institut, 2014), and studies in connection with NGOs, concerning manufacturing pollutions, working conditions, and e–waste treatment in countries of Asia, Africa, and South America, e.g. (Caravanos et al, 2011; Greenpeace, 2014; Step, 2013).

Electronic devices contain a large number of components and materials. To structure and categorise the needed life cycle inventory data, the study by Andrae et al. presents a generic model for life cycle inventory data collection applied to electronic devices. The electronic components are divided into main groups and sub-groups and their process steps are classified in similar intermediate unit processes (IUPs). The results show that the IUPs for integrated circuits and printed circuit boards have a significant higher global warming potential per kg than other components and device parts. (Andrae et al., 2005)

Further studies investigate the applicability of life cycle assessment methods for microelectronics and the environmental impact of materials for interconnection technologies for electrical engineering, lead and lead–free high– and low–melting point solders based on nano–particles, substrate materials, coating techniques, and wave soldering. The results of the comparative attributional LCA of Sn-Pb and Pb-free solder shows that the manufacturing of lead-free solder increases the global warming potential by 10%. From the consequential point of view the eliminated lead from the solder life cycle is partly offset by increased Pb use in e.g. batteries. (Andrae, 2009/2014).

The study conducted by Chen et al. investigates in a comparative and simplified environmental assessment the greenhouse warming potential (GWP) and abiotic resource depletion (ADP) of different substrate materials (FR4, LTCC, LCP, PTFE) for system–in–a–package applications. The LCA methods Eco-Indicator 99 and Environmental Priority Strategy (EPS) were chosen for weighting. The results show that the liquid crystal polymer (LCP) material had the best environmental performance for almost all considered environmental impact categories. (Chen et al., 2004)

The environmental impact of semiconductor devices was investigated by Boyd. The studies provide life cycle inventory data and midpoint LCA results for the manufacturing of semiconductors for several product generations, including wafer production, high pure chemicals, and energy consumption. The book provides evaluation results of most common types of integrated circuits of logic and memory types from many technology generations of the last decades. (Boyd, 2014)

The publication by Andersen et al. presents the results of four independently conducted comparative LCA studies of the categories laptop computers, mobile phones, interconnect technologies in electronics micro–integration, and solar cells. In the case study of solar cells, a new etching process is compared with the current applied wet chemical etching process of the crystalline silicon wafers. The results show that the new dry texturing process has the potential to reduce the water consumption by 86% to 89% and the greenhouse gas emissions are reduced with 63% to 20% dependent on the wafer type. (Andersen et al., 2014)

A number of LCA studies of LED lamps were conducted comparing the environmental performance with other efficient lamps as well as the relations of the characterisation factor results between manufacturing and use stage. Dependent on the assumed forecasted realistic life time and detailed manufacturing life cycle inventory data the results differ in a wide range. In case of the global warming potential, the study by (Osram, 2009) shows that the manufacturing impact share is about 1.5% whereas the study of (Tähkämö et al., 2013) determined values between 15% and 40%.

A review of methodological approaches of life cycle assessments of consumer electronics is conducted by Andrae. In a case study of a smartphone, the upstream electrical energy consumption is 47 kWh per piece. Thereof, the use phase energy consumption is 2-6 kWh per year dependent on the type. The comparison of 23 LCA methods shows that the *European Commission product environmental footprint* (EC PEF) method gives the smallest errors. (Andrae, 2016).

The global electrical energy consumption of the communication technology sector was investigated by Andrae & Edler. The study includes three scenarios for production and use of communication consumer devices and networks, as well as data centers. The trends for all scenarios show that the share of the energy consumption of the use phase per device will decrease and will be transferred to data centers and networks. In 2030, the expected total share of the global electrical energy consumption of this sector is 21%, the worst case is estimated with 51%; the share of the electrical energy consumption of the production stage of devices, network and data centers is 11% for the expected case and 21% for the worst case in relation to the total electrical energy demand of this sector. (Andrae & Edler, 2015; Andrae, 2017)

Concerning electrical energy generation, comprehensive life cycle inventory data for all photovoltaic technologies were collected and prepared by (De Wild–Scholten, 2014). A further source for environmental data of renewable energy generation is the U.S. National Renewable Energy Laboratory (NREL), which evaluates published life cycle assessment studies for energy technologies in a harmonisation project, including wind, solar, photovoltaics, nuclear, coal, and natural gas (NREL, 2019).

A further article addresses the regional context dependency for renewable energy generation. All renewable energy sources have the common property that they compete for the limited land area with food and feed production, and the preservation of natural habitats. While the generation of wind, water, and solar energy is dependent on annual and daily weather volatility, energy based on biomass is base-load capable, i.e. it can be stored. Therefore, bio–energy is available "on demand" which makes it attractive for a future key role for sustainable energy generation. However, energy production from biomass requires new logistics and structure for the transport within the energy supply chain. Furthermore, besides the global CO_2 balance, the environmental assessment must take additional emissions to air, water, and soil into account on local level. Renewable energy production has to be integrated into a spatial planning context, for which new rules have to be developed. (Kettl et al., 2009; Narodoslawsky, 2017)

The environmental impact of electronic waste is investigated by a number of studies. As an example, a life cycle assessment of e–waste treatment in China was conducted by Hong et al. The study compares two scenarios of e–waste management, the one is a formal treatment by a professional dismantling enterprise and the other one is an informal treatment in the largest e–waste site in the world, in Guiyu. The results show that the informal recycling generates a significant local environmental pressure by direct emissions of toxic materials into air, water, and soil. As examples, the environmental impact indicator human toxicity was 20 times higher, freshwater ecotoxicity was 50 times higher, and terrestrial ecotoxicity was 35000 times higher than for formal recycling (Hong et al., 2015). The Chinese restriction of hazardous substances in electronic products is not a ban but a declaration requirement (China–RoHS, 2019).

Another study about e–waste management investigated the environmental impact of a new recycling process for silicon photovoltaic panels. The results show that the environmental pressure is concentrated on the incineration process of the encapsulating layers and the recovery treatment of silicon, copper, silver, and aluminium. It is noted that the transport of the PV modules to the treatment site shows in almost all categories a significant impact share. (Latunussa et al., 2016)

The conduction of a standardised life cycle assessment study requires matching detailed life cycle inventory data of the referring materials and processes. The main database for electronic devices and components is provided by Ecoinvent (Hischier et al., 2007). The comprehensive database includes life cycle inventory data for electronic components (microchips, transistors, diodes, capacitors, inductors, resistors, transformers, switches), electronic modules (printed wiring boards, cables, batteries, toner, drives, power supply unit), electronic devices (desktop computer, LCD screen, keyboard, mouse, laptop, power adapters, printer, ancillaries, and waste treatment), use (use of a desktop computer, use of a laptop, use of a laser printer), and disposal of e–waste (dismantling, treatment of fractions).

The Ecoinvent database further includes life cycle inventory datasets for photovoltaics, transport, electrical energy generation, metals, chemicals, plastics, etc. The documentations are available on the website (Ecoinvent, 2019) and provide transparent descriptions of the data backgrounds, assumptions, and data gaps. Based on this it is possible to decide, if the available dataset can be applied to the respective research question.

Further datasets for electronics are provided by the GaBi–database, which includes 247 datasets for different types of batteries, electronic components, such as resistors, capacitors, integrated circuits, LEDs, and printed wiring boards (GaBi, 2019).

Outside the scope of life cycle assessments, the impact of renewable electrical energy generation on animals is exemplary investigated by Wang et al., where the possible causes of collision between wind power plants and birds are reviewed. The authors summarise that the reasons why and how birds are killed in the area of wind power plants is insufficiently investigated. They elaborated six hypotheses as basis for systematic further research involving location and topography of wind power, lighting of turbines, turbine design characteristics and layout of wind farm, bird activities, flying conditions, and habitat disturbance. (Wang et al., 2015)

Considerations about negative consequences of wind farms were already published in 2011 in the Australian Journal *Nature & Society*. The article addresses open research questions on sleep disturbances of people living near wind farms, seismic vibrations disturbing insect and earthworm population around onshore wind parks, as well as fish population around offshore wind parks. It is also addressed the enhancing negative impact from not synchronously running generators of plants close to each other and the disturbance of seismic instruments. The author calls for systematic scientific research on thousands of reports from doctors about stress, hypertension, and sudden bursts of tachycardia of their patients. (Whisson, 2011)

In 2014, the German Environment Agency published the "Feasibility study on effects of infrasound – Development of investigation designs for determining the effects of infrasound on humans through different sources". The study identifies the most relevant sources for harassment of infrasound, that is heat pumps, biogas plants, thermal power stations, wind power plants, and building ventilation systems. A comparison of test results showed that negative effects of infrasound in the frequency range below 10 Hz cannot be excluded even at sound pressure levels below the threshold of hearing. Most of the documented diseases concern the cardiovascular system, concentration and reaction time, organs of balance, the nervous system, and auditory sensory organs. Protection against infrasound is de facto impossible. The required masses or volumes are inversely proportional to the frequencies and therefore, noise protective walls would have to be several meters thick. The study explicitly emphasizes that the literature does not present a coherent picture how to determine and assess low frequency sounds. (UBA, 2014)

The existing literature shows that there is generally a very small number of published LCA studies of new electronic components, modules or devices available compared to the market share of the electrical sector. The case study objects are according to the highly segmented sector of selected and isolated nature and the system boundaries and functional units are likewise highly diverse, making it difficult to compare the studies.

The assessment method is almost exclusively an attributional LCA. This is probably due to the fact that the electrical sector has a large variety of products and is subject to rapid technology changes. Almost all authors complain that there is insufficient life cycle inventory data available for the manufacturing processes, and it seems that they are more concerned with getting them as close as possible. Another explanation is that the environmental focus of electrical devices is on the use phase, that is, the consumption of electrical energy. Although the applied respective regional energy mix of manufacturing and use phase is distinguished the global consequences of the "grey energy" are insufficiently illuminated in the course of the European and U.S. energy efficiency action plans.

LCA studies on new electrical energy generation systems also have an attributional approach. Life cycle inventory data for photovoltaic modules seems to be better captured. The environmental pressure is mainly focused on the level of energy balances and recycling technologies. There are a few important studies addressing the consequences for human and animal health, and natural habitats outside of the scope of a LCA approach. All these studies indicate that there is an urgent need for further research.

The key challenges for the environmental assessment of new electrical equipment are the evaluation of better LCI data, the expansion to more meaningful and comparable functional units, and the conduction of consequential LCA studies and technology assessments, which incorporate a larger spatial, temporal, and global context and also consider aspects outside an LCA.

The highly dynamic electrical market will grow faster and scientific environmental research is caught between the individual assessment of new changing technologies and the simultaneously evaluation of their global environmental pressure which depends on the availability of market data. Actual global economic market data and forecasts on electronics are provided by industrial lobby groups such as Reed Electronics Research (Reed Electronics, 2019), which, however, is expensive for scientific researchers and they can only give an indication of global mass flows. Further market data for printed circuit boards are commercially available by the Association Connecting Electronics Industries–IPC (IPC, 2019). Summaries and excerpts of market data are provided by the German Electrical and Electronic Manufacturers' Association–ZVEI (ZVEI, 2019).

3.6. References

Andersen, Otto; Hille, John; Gilpin, Geoffrey; Andrae, Anders S. G. (2014). Life Cycle Assessment of Electronics. IEEE Conference on Technologies for Sustainability, IEEE Xplore, DOI: 10.1109/SusTech.2014.7046212.

Andrae, Anders S.G.; Andersson, Dag R.; Liu, Johan (2005). Significance of intermediate production processes in life cycle assessment of electronic products assessed using a generic compact model. Journal of Cleaner Production 13 (2005) 1269–1279.

Andrae, Anders (2009). Usefulness and Precision of LCA in Microelectronics Packaging. VDM Verlag.

Andrae, Anders S. G. (2014). Global Life Cycle Impact Assessments of Material Shifts. Springer, 2014, 208p.

Andrae, Anders S. G.; Edler, Tomas (2015). On Global Electricity Usage of Communication Technology: Trends to 2030. Challenges 2015, 6, 117-157; doi:10.3390/challe6010117.

Andrae, Anders S. G. (2016). Life-Cycle Assessment of Consumer Electronics. A review of methodological approaches. IEEE Xplore, IEEE Consumer Electronics Magazin, January 2016, p. 51–60.

Andrae, Anders S. G. (2017). Total Consumer Power Consumption Forecast. Presentation at the Nordic Digital Business Summit, Helsinki, Finland, October 5, 2017.

Baccini, P.; Brunner, Paul H. (1991). Metabolism of the anthroposphere. Springer-Verlag, 1991.

Baldé, C.P.; Wang, F.; Kuehr R.; Huisman J. (2015). The global e-waste monitor–2014. United Nations University, IAS SCYCLE, Bonn, Germany.

Banta, David (2009). What is technology assessment? International Journal of Technology Assessment in Health Care, 25:Supplement 1(2009), 7–9.

Battelle-Institut e.V. (1973). Studie über den Beitrag abbaubarer Kunststoffe zur Lösung des Müllproblems. Report to the Bundesministerium für Forschung und Technologie, Bonn, (BMFT/III B 3). Project Nos.: NTÖ 14 (BMFT); R 61.751 (Battelle). Frankfurt/Main, June 1973.

Boyd, Sarah B. (2014). Life-Cycle Assessment of Semiconductors. Springer, 2014, 256p.

Brunner, Paul H.; Rechberger, Helmut (2017). Handbook of Material Flow Analysis: For Environmental, Resource, and Waste Engineers. Second Edition. CRC Press, Taylor & Francis Group, 2017.

Caravanos, Jack; Clark, Edith; Fuller, Richard; Lambertson, Calah (2011). Assessing Worker and Environmental Chemical Exposure Risks at an e-Waste Recycling and Disposal Site in Accra, Ghana. Journal of Health and Pollution: February 2011, Vol. 1, No. 1, pp. 16-25.

Cencic, O.; Frühwirth, R. (2018). Data reconciliation of nonnormal observations with nonlinear constraints. Journal of Applied Statistics, 45 (2018), 13; S. 2411 - 2428.

Cencic, O.; Rechberger, H. (2008). Material Flow Anaylsis with Software STAN. Journal of Environmental Engineering and Management (JEEM), 18 (2008), 1; S. 3 - 7.

Chen, L.; Andræ, A. S. G.; Zou, G.; Liu, J. (2004). Characterization of Substrate Materials for System-in–a–Package Applications. Journal of Electronic Packaging 126 (2): 195-201.

China–RoHS (2019). Management Methods for the Restriction of the Use of Hazardous Substances in Electrical and Electronic Products. http://www.chinarohs.com/ [04/2019].

Curran, Mary Ann (Ed.) (1994). Life-Cycle Assessment: Inventory Guidelines and Principles. Battelle / EPA contract No. 68-C0-0003, prepared for the U.S. Environmental Protection Agency (EPA). Lewis Publishers, CRC Press Inc. 1994.

De Wild–Scholten, M. J. (2014). Life Cycle Assessment of Photovoltaics Status 2011, Part 1. Data Collection, report published by SmartGreenScans, The Netherlands.

Duvigneaud, P.; Denayer–De Smet, S. (1977). L'écosystème Urbs: L'écosystème urbain bruxellois, Scope, 1975.

EC–European Commission (Ed.) (1996). Schelleman, F.; Krozer, J.; Heijnes, H.; Narodoslawsky, M.; Eder, T.; Dielacher, T.; Schmidt-Bleek, F.; Malley, J.; Stiller, H.; Spangenberg, J.; Ayres, R.U.; Martinas, K.: Operational Indicators for Progress Towards Sustainability. Final report of an EC study aimed at developing a set of sustainability indicators based on LifeCycle approaches. Commissioned by: European Commission DG XII Project no.: EV5V-CT94-0374, 30 September 1996.

Ecoinvent (2019). http://www.ecoinvent.org/ [03/2019].

Fiedeler, Ulrich; Fleischer, Torsten; Decker, Michael (2004). Roadmapping als eine Erweiterung des „Methoden-Werkzeugkastens“ der Technikfolgenabschätzung?. Technikfolgenabschätzung – Theorie und Praxis Nr. 2, 13. Jg., Juni 2004.

Fischer-Kowalski, Marina; Haberl, Helmut (1993). Metabolism and Colonization. Modes of Production and the Physical Exchange between Societies and Nature. Innovation-The European Journal of Social Science Research 6(4):415-442, 1993.

Friedl, Anton; Mihaliy, Bettina (2019). Design and assessment of sustainable processes - Technology Assessment. Lecture notes. Institute of Chemical, Environmental and Bioscience Engineering, TU Wien, Austria.

GaBi (2019). Thinkstep GaBi–Extension database XI: Electronics. http://www.gabi-software.com/ [02/2019].

Goldsmith, Edward (1972). A Blueprint for Survival. The Ecologist, January 1972.

Greenpeace (2014). Maddy Cobbing and Tom Dowdall: Green Gadgets: Designing the future. The path to greener electronics. http://www.greenpeace.org/international/Global/international/publications/toxics/2014/Green%20Gadgets.pdf [10/2016].

Groß, Rita; Bunke, Dirk; Gensch, Carl-Otto; Zangl, Stephanie; Manhart, Andreas (2008). Study on Hazardous substances in electrical and electronic equipment, not regulated by the RoHS Directive; Final Report. Institute for Applied Ecology, Öko-Institut e.V., Germany, 273 pages.

Guinée, Jeroen B. (2012). Life cycle assessment: past, present and future. International Symposium on Life Cycle Assessment and Construction July 10-12, 2012, Nantes, France.

Guinée, J. B., Heijungs, R.; Huppes, G.; Zamagni, A.; Masoni, P.; Buonamici, R.; Ekvall, T.; Rydberg, T. (2011). Life cycle assessment: past, present and future. Environ. Sci. Technol. 45 (1) (2011) 90–96.

Heijungs, R.; Guinée, J.B.; Huppes, G.; Lankreijer, R.M.; Udo de Haes, H.A.; Wegener Sleeswijk, A.; Ansems, A.M.M.; Eggels, P.G.; Duin, R. van; Goede, H.P. de (1992). Environmental life cycle assessment of products: guide and backgrounds (Part 1). CML, Leiden.

Hischier, Roland; Classen, Mischa; Lehmann, Martin; Scharnhorst, Wolfram (2007). Life Cycle Inventories of Electric and Electronic Equipment: Production, Use and Disposal. Data v2.0 (2007). ecoinvent report No. 18.

Hong, Jinglan; Shi, Wenxiao; Wang, Yutao; Chen, Wei; Li, Xiangzhi (2015). Life cycle assessment of electronic waste treatment. Waste Management 38 (2015) 357–365.

Hunt, Robert G.; Franklin, William E., Welch, Richard O.; Cross, James A.; Woodall, Alan E. (1974). Resource and Environmental Profile Analysis of Nine Beverage Container Alternatives: Final Report. Report No. SW-91c under Federal solid waste management contract no. 68-01-1848. U.S. Environmental Protection Agency EPA/530/SW-91c, 1974, 185 pages. Online available: https://nepis.epa.gov/ [03/2019].

Int J Life Cycle Assess (2019). The International Journal of Life Cycle Assessment. Springer. ISSN: 0948-3349. https://www.springer.com/environment/journal/11367 [02/2019].

IPC Association Connecting Electronics Industries (2019). http://www.ipc.org/ [04/2019].

ISO 14040 (2006a). Environmental management – Life cycle assessment – Principles and framework.

ISO 14044 (2006b). Environmental management – Life cycle assessment – Requirements and guidelines.

ITA-Institute of Technology Assessment of the Austrian Academy of Sciences (2019a). What is TA – Geschiche der TA. Austrian Academy of Sciences, Vienna. Online: https://www.oeaw.ac.at/ita/de/ueber-uns/was-ist-ta/geschichte-der-ta/ [03/2019].

ITA-Institute of Technology Assessment of the Austrian Academy of Sciences (2019b). What is TA – Methods and Approaches. Austrian Academy of Sciences, Vienna. Online: https://www.oeaw.ac.at/ita/en/about-us/what-is-ta/methods-and-approaches/ [03/2019].

Jensen, Allan Astrup; Postlethwaite, Dennis (2008).SETAC Europe LCA Steering Committee – The Early Years. Int J LCA 13 (1) 1 – 6 (2008), Springer-Verlag.

Kaltschmitt, Martin; Schebek, Liselotte (2015). Umweltbewertung für Ingenieure: Methoden und Verfahren. Springer Vieweg, 2015.

Kettl, Karl-Heinz; Eder, Michael; Sandor, Nora Katalin; Narodoslawsky, Michael (2009). Ecological impact of renewable resource based energy technologies. Chemical Engineering Transactions 18, January 2009, pp. 611 - 616; DOI: 10.3303/CET0918099.

Klöpffer, Walter; Grahl, Birgit (2014). Life Cycle Assessment (LCA), A Guide to Best Practice. Wiley-VCH Verlag GmbH & Co. KGaA, Weinheim, Germany.

Knill, C.; Liefferink, D. (2007). Environmental politics in the European Union. Manchester University Press, Manchester.

Latunussa, Cynthia E.L.; Ardente, Fulvio ; Blengini, Gian Andrea; Mancini, Lucia (2016). Life Cycle Assessment of an innovative recycling process for crystalline silicon photovoltaic panels. Solar Energy Materials and Solar Cells, Vol. 156, Nov. 2016, Pages 101–111.

Matthews, E.; Amann, C.; Bringezu, S.; Fischer-Kowalski, M.; Hüttler, W.; Kleijn, R.; Moriguchi, Y.; Ottke, C.; Rodenburg, E.; Rogich, D.; Schandl, H.; Schütz, H.; Van der Voet, E.; Weisz, H. (2002). The Weight of Nations. Material Outflows from Industrial Economies. World Resources Institute, Washington, DC.

Meadows, Dennis (Ed.) (1972). The Limits to Growth: A Report for the Club of Rome's Project on the Predicament of Mankind. Universe Books, New York, 1972.

Narodoslawsky, Michael (2017). Bioenergy provision: utilizing contextual resources. Current Opinion in Chemical Engineering Volume 17, August 2017, Pages 93-97.

Narodoslawsky M.; Krotscheck C.; (1995). The sustainable process index (SPI): Evaluating processes according to environmental compatibility; J. of Hazardous Materials, 1995, 41 (2+3), 383-397.

Narodoslawsky, Michael; Shahzad, Khurram (2015). What Ecological Indicators Really Measure – The Normative Background of Environmental Evaluation. Chemical Engineering Transactions, vol. 45, 2015.

Narodoslawsky, Michael; Stoeglehner, Gernot (2010). Planning for Local and Regional Energy Strategies with the Ecological Footprint. Journal of Environmental Policy & Planning Vol. 12, No. 4, December 2010, 363–379.

NREL (2019). Life Cycle Assessment Harmonization. U.S. Department of Energy – National Renewable Energy Laboratory, http://www.nrel.gov/ [03/2019].

OECD–Organisation for Economic Co-operation and Development (2008). Measuring Material Flows and Resource Productivity. Volumes I-IV. Volume I: https://www.oecd.org/ environment/indicators-modelling-outlooks/MFA-Guide.pdf [03/2019].

OECD–Organisation for Economic Co-operation and Development (2011). Towards Green Growth. OECD Publishing. https://dx.doi.org/10.1787/9789264111318-en [03/2019].

OECD–Organisation for Economic Co-operation and Development (2017). Green Growth Indicators 2017. http://www.oecd.org/environment/indicators-modelling-outlooks/Highlights_Green_Growth_Indicators_2017.pdf [03/2019].

Oeko-Institut, BIO by Deloitte, ERA Technology. Preparatory study to establish the ecodesign working plan 2015–2017 implementing directive 2009/125/ec. task 2. Draft final report prepared for the european commission (dg entr). Technical report, 2014.

Osram, Siemens Corporate Technology (2009). Life-cycle Assessment of Illuminants – a Comparison of Light Bulbs, Compact Fluorescent Lamps and LED Lamps, Executive Summary.

Reed Electronics Research (2019). http://www.rer.co.uk/ [04/2019].

Schmidt-Bleek, Friedrich (1993). MIPS—A universal ecological measure. Fresenius Environmental Bulletin, 1 (1993), pp. 306-311.

Schmidt-Bleek, F. (1994) MIPS: Ein neues ökologisches Maß. In: Wieviel Umwelt braucht der Mensch?. Birkhäuser, Basel.

Schön-Quinlivan, Emanuelle (2012). The European Commission, In: Jordan, A.J. and Adelle, C. (Eds) Environmental Policy in the European Union: Contexts, Actors and Policy Dynamics (3e). Earthscan: London and Sterling, VA.

Selin, Henrik; VanDeveer, Stacy D. (2015). EU Environmental Policy Making and Implementation: Changing Processes and Mixed Outcomes. 14th Biennial Conference of the European Union Studies Association, Boston, Massachusetts, March 2015.

SETAC–Society of Environmental Toxicology and Chemistry (1993). Guidelines for Life Cycle Assessment: A „Code of Practice“. Edition 1. From the SETAC Workshop held at Sesimbra, Portugal, 31 March - 3 April 1993. Brussels, Belgium, and Pensacola, Florida, USA, Aug. 1993.

SETAC–Society of Environmental Toxicology and Chemistry (2019). History of SETAC. https://www.setac.org/page/History [03/2019].

Sörme, L.; Bergbäck, B.; Lohm, U. (2001). Goods in the Anthroposphere as a Metal Emission Source. A Case Study of Stockholm, Sweden. Water, Air, and Soil Pollution: Focus1: 213–227, 2001. Kluwer Academic Publishers.

Sotoudeh, Mahshid (2009). Technical Education for Sustainability–An Analysis of Needs in the21st Century. Peter Lang-Verlag, 2009, 244 pages. ISBN 9783631585948.

Sotoudeh, Mahshid (2010). Introduction: Technical Universities for Sustainable Development Learning to Deal with Complexity. GAIA - Ecological Perspectives for Science and Society, March 2010, Vol.19(1), pp.33-36.

Sotoudeh, Mahshid; Narodoslawsky, Michael (2018). Ökobilanzen und normative Urteile. Internationale TA18-Konferenz: Technikfolgenabschätzung und Normativität – An welchen Werten orientiert sich TA?, Institute of Technology Assessment of the Austrian Academy of Sciences ,Wien, 11. Juni 2018.

SPOLD–Society for Promotion of LCA Development (2019). https://lca-net.com/spold/whatis.html [03/2019].

Springer (2019). The International Journal of Life Cycle Assessment. https://link.springer.com/journal/11367 [03/2019].

STAN–(2019). Software for substance flow analysis. http://www.stan2web.net/ [03/2019].

Step (2013). Deepali Sinha Khetriwal, Claudia Luepschen, Ruediger Kuehr (Eds.). Solving the e–waste problem: An interdisciplinary compilation of international e–waste research. United Nations University Press. http://www.step-initiative.org/ [10/2016]

Tähkämö, L., Bazzana, M., Ravel, P., Grannec, F., Martinsons, C., Zissis, G. (2013). Life cycle assessment of light–emitting diode downlight luminaire – a case study; Int. Journal of Life Cycle Assessment, 18, 1009-1018, 2013.

Vig, Norman J.; Paschen, Herbert (Eds.) (2000). Parliaments and Technology: The Development of Technology Assessment in Europe. State University of New York Press, Albany.

Wackernagel, M.; Monfreda, C.; Deumling, D. (2002). Ecological footprint of nations. Redefining Progress, Sustainability Issue Brief November 2002.

Wang, Shifeng; Wang, Sicong; Smith, Pete (2015). Ecological impacts of wind farms on birds: Questions, hypotheses, and research needs. Renewable and Sustainable Energy Reviews 44(2015) 599–607.

Whisson, Max (2011). Wind power and ecology. Nature and Society, October-November 2011; ISSN: 1038–5665.

Wolman, Abel (1965). The Metabolism of Cities. Scientific American, Volume 213, Pages 179-190. https://irows.ucr.edu/cd/courses/10/wolman.pdf [03/2019].

Wong, Jennifer (2014). The History of Technology Assessment and Comparative Effectiveness Research for Drugs and Medical Devices and the Role of the Federal Government. Biotechnology law report, 33(6), 221-248.

ZVEI German Electrical and Electronic Manufacturers' Association (2019). Online: https://www.zvei.org/ [04/2019].

Part I

4. Global Electrical Market and e–Waste Generation

The global electro industry is a highly diverse production sector from generators and power lines to electrical appliances, electronic components, and new materials containing rare earths or nano–particles. There are different views and sources to classify the electro industry. The respective product groups can overlap in a wide range between the different classifications, or exclude large sectors. To give a detailed overview of the entire electro industry, the following chapter provides a comprehensive data collection of classifications and the international electro market as well as electronic waste generation (e–waste or WEEE). The following topics are considered:

- Classification of electrical products and activities according to different sources (section 4.1.1.)
- Classification of electrical products by material and functional groups (4.3.2.)
- Global market of the electrical industry, electronic components, and semiconductors by sectors and by countries (4.2.1.)
- German market and sectors of semiconductors (4.2.2.)
- Austrian electro market and sectors (4.2.3.)
- Global generation of e–waste by countries and sectors (4.3.1.)
- EU electric and electronic equipment (EEE) put on the market and collected e–waste by countries (4.3.2.)
- Austrian EEE put on the market and e–waste generation by sectors (4.3.3.)

4.1. Classification of the Electrical Sector

The evaluation of the total global environmental impact of electric and electronic equipment as well as electricity generation depends on the system boundary definition of the respective considered electrical sector. Comparing the literature, varying definitions and classifications of electric equipment are commonly used. The following tables show a comparison of included product groups and activities in the electrical sector.

4.1.1. Classification of Economic Activities of the Electrical Sector

The European Union implemented a regulation on the classification of the economic activities in the European Community (NACE, 2006) with the current status of 2012, which corresponds to the International Standard Industrial Classification (ISIC) standards of the United Nations (UN Statistics, 2008). The NACE classification structures the entire global economic output in 99 divisions, which are subdivided in groups and further in classes. Table 4.1. shows a summary of the classifications concerning the electrical industry. The divisions 26 and 27 correspond to the traditional partitions of the electrical and the electronic sector.

Table 4.1. Classification of the electrical sector according to NACE (NACE, 2006).

Manufacture of computer, electronic and optical products (Division 26)
Manufacture of electronic components and boards
Manufacture of computers and peripheral equipment
Manufacture of communication equipment
Manufacture of consumer electronics
Manufacture of instruments and appliances for measuring, testing and navigation; watches and clocks
Manufacture of irradiation, electromedical and electrotherapeutic equipment
Manufacture of optical instruments and photographic equipment
Manufacture of magnetic and optical media
Manufacture of electrical equipment (Division 27)
Manufacture of electric motors, generators, transformers and electricity distribution and control apparatus
Manufacture of batteries and accumulators
Manufacture of wiring and wiring devices
Manufacture of electric lighting equipment
Manufacture of domestic appliances
Manufacture of other electrical equipment

The divisions 26 and 27 cover a large part of the manufacturing phase of "typical" electrical and electronic equipment. Table 4.2. list further activities of the NACE classification of other divisions, which also relate to the electrical industry sector. They concern the electrical energy supply, motor vehicles, repair and service activities, which are mainly associated with the using phase of the electric sector. Furthermore, there is one general division for waste management.

Table 4.2. Parts of other divisions of the NACE classification concerning the electrical sector.

Further divisions, groups and classes relating to the electrical sector:
Manufacture of electrical and electronic equipment for motor vehicles (Class 29.31)
Electric power generation, transmission and distribution (Group 35.1)
Steam and air conditioning supply (parts thereof) (Group 35.3)
Waste collection, treatment and disposal activities; materials recovery (parts thereof) (Division 38)
Construction of utility projects for electricity and telecommunications (Class 42.22)
Electrical installation (Class 43.21)
Retail sale of electrical household appliances in specialised stores (Class 47.54)
Retail sale of furniture, lighting equipment and other household articles in specialised stores (parts thereof) (Class 47.59)
Retail sale of music and video recordings in specialised stores (Class 47.63)
Software publishing (Group 58.2)
Motion picture, video and television programme production, sound recording and music publishing activities (Division 59)
Programming and broadcasting activities (Division 60)
Telecommunications activities (Division 61)
Computer programming, consultancy and related activities (Division 62)
Information service activities (Division 63)
Repair of computers and personal and household goods (part thereof) (Division 95)

There are some more goods not separately listed here, which also can include electrical components, such as music instruments, sports equipment, and toys.

The NACE classification is certainly the most comprehensive compilation, which covers the entire electrical sector. In the literature there are different notable publishers of economical data, which cover only partitions of the electrical sector. Tables 4.3. to 4.5. show different distinctions of the electrical sector, their corresponding economic data are presented in section 4.2.

Table 4.3. Global electrical market by sectors in 2010, according to ifo–Schnelldienst (Gontermann, 2012).

Global electrical market by sectors according to ifo–Schnelldienst:
Automation
Components
Consumer electronics
Domestic appliances
Electro medicine
Information and communication technology (ICT)
Light
Power engineering
Others

Table 4.4. Sector structure of the electrical industry according to Bank Austria Economics & Market Analysis Austria (Wolf, 2014).

Electrical industry sectors according to Bank Austria Economics & Market Analysis Austria:
Batteries
Communications equipment
Components
Consumer electronics
Data processing equipment
Domestic appliances
Installation material
Luminaires
Measurement technology
Medical technology
Motors, generators
Optical and photo equipment
Other electrical equipment

Table 4.5. Electronic sectors according to Reed Electronics Research (Reed Electronics, 2015).

Electronic sectors according to Reed Electronics Research:
Components
Consumer electronics
Electronic data processing (EDP)
Measuring & controlling
Medicine & industry
Office equipment
Radio & radar
Telecommunication

4.1.2. Classification of the Electrical Sector from the Point of View of Waste Management

In contrast to the economical classifications presented above, the e–waste Directive of the European Union classifies the considered sectors of the electrical industry by size and function. Table 4.6. shows the classification divisions of the WEEE Directive, valid as of 2018 (EU Directive, 2012).

Table 4.6. Classification according to the WEEE Directive.

Classification of electrical and electronic equipment according to WEEE:
1. Temperature exchange equipment
2. Screens, monitors (surface greater than 100 cm^2)
3. Lamps
4. Large equipment (any external dimension more than 50 cm)
5. Small equipment (no external dimension more than 50 cm)
6. Small IT and telecommunication equipment (no external dimension more than 50 cm)

4.1.3. Classification of Electrical Products by Materials and Functional Groups

Depending on further specific issues, materials and components of electrical products can be classified, for example, in following groups:

- Ferrous metals, non–ferrous metals, plastics, ceramics, glass, etc.
- Valuable and hazardous materials (copper, noble metals, rare earths, lead, etc.)
- Recycling materials and non–recycling materials
- Mechanical housing components, electro–mechanical parts, and electrical components

The Austrian coordination centre for e–waste management published a statistical distribution of material groups across the total waste of electrical and electronic equipment, see figure 4.1. (EAK, 2015). Ferrous metals have a share of 45 %, followed by plastics and glass with about 16 % each.

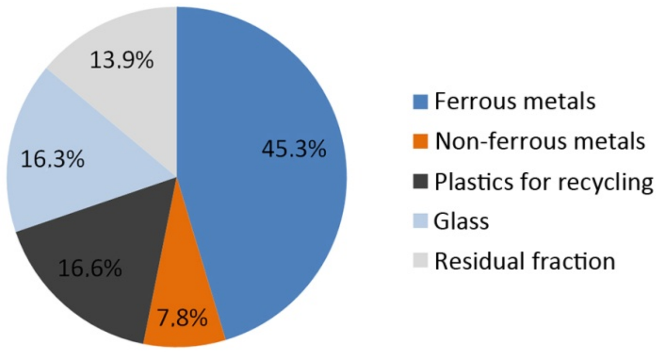


Figure 4.1. Material composition of the total e–waste in Austria.
(Own work; Data source: EAK, 2015)

The same literature provides further the average material composition of all collected batteries, which is shown in figure 4.2. More than 50 % of the materials are lead metal and 18 % are sulphuric acid. These relations result from the high mass share of vehicle batteries of about 85 %.

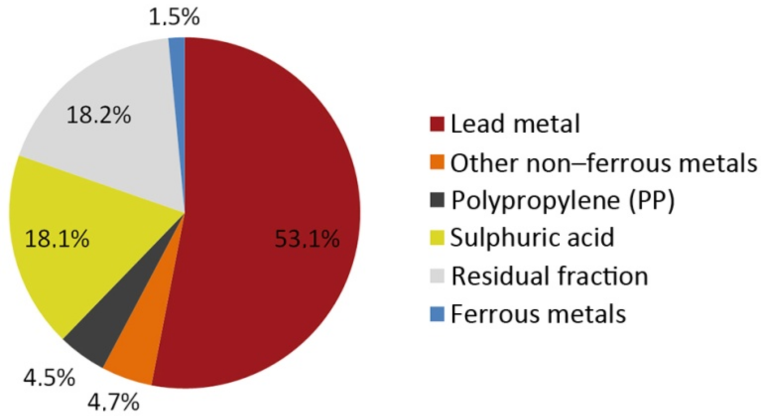


Figure 4.2. Average material composition of batteries in Austria.
(Own work; Data source: EAK, 2015)

4.2. Electrical Industry Market

4.2.1. Global Market

The total global production of all industrial sectors was 29.3 trillion Euro in 2010. Thereof, 11 % are represented by the electrical industry, followed by the chemical industry and the automotive industry. Figure 4.3. shows the global market situation for the technical industrial sectors.

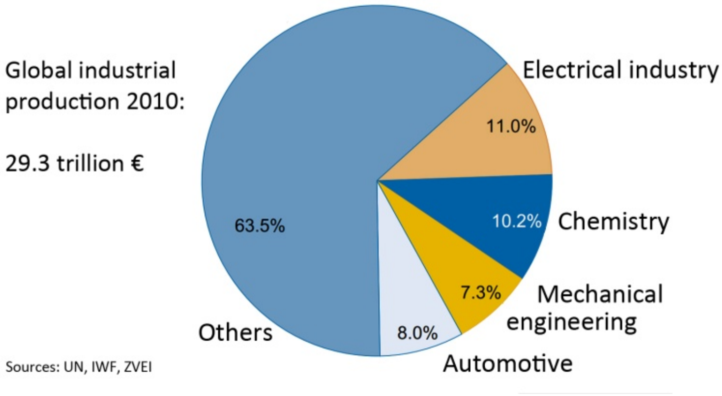


Figure 4.3. Global industrial production in 2010 (Gontermann, 2012).

According to the same literature source, the global electrical industry is divided in nine subsectors, whereby 45 % of the production market is shared by the components and information & communication sectors. Figure 4.4. shows the market share of all nine sectors with a total production of 3.4 trillion Euro in 2010. In 2016, the world electrical market increased to 3.99 trillion Euro (ZVEI, 2018).

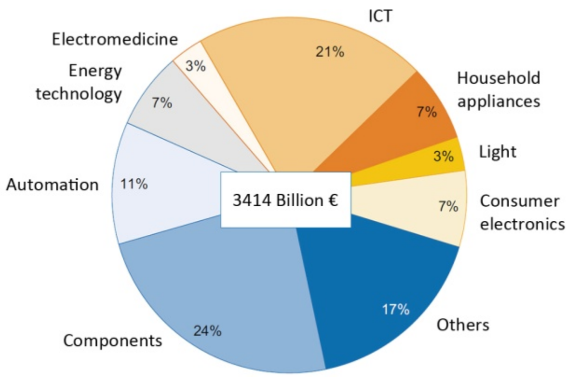


Figure 4.4. Global electrical market and its sectors in 2010 (Gontermann, 2012).

Compared to the wide–ranging market overview, Reed Electronics Research published market data including the subarea of the electronic market, which covers about 45 % of the global electrical market. Figure 4.5. shows that more than 50 % of the electronic market share includes equally components and electronic data processing equipment, followed by radio & radar. The European Union including Switzerland and Norway (EU28+2) produces 14 % of the global electronic market. The same countries consume one third more electronic products than they produce. (Reed Electronics, 2015).

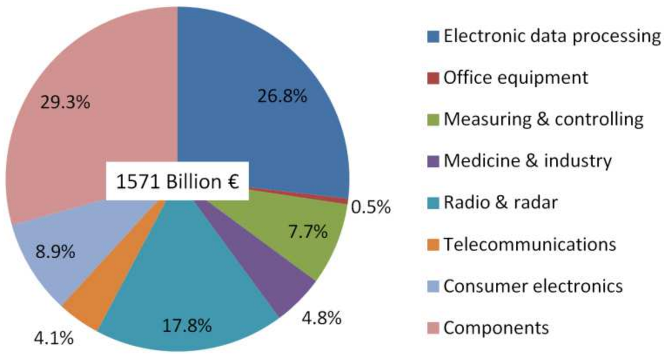


Figure 4.5. Global electronic market and its subsectors in 2011.
(Own work; Data source: Reed Electronics, 2015)

A comparison of figure 4.4. with figure 4.5. shows that despite of partly different defined market sectors, the respective total sales of the sectors have no consistent values. This can be seen from the components sector, the value shown in Figure 4.5 compared to Figure 4.4. should be about twice as much. Their allocation to the likewise considered final product groups, which include these electronic components, is not defined.

The market statistics so far addressed the global production share by subsectors. Figure 4.6. shows the global electrical market by regions and countries (Gontermann, 2012). Asia is the largest producer, followed by the Americas and then Europe.

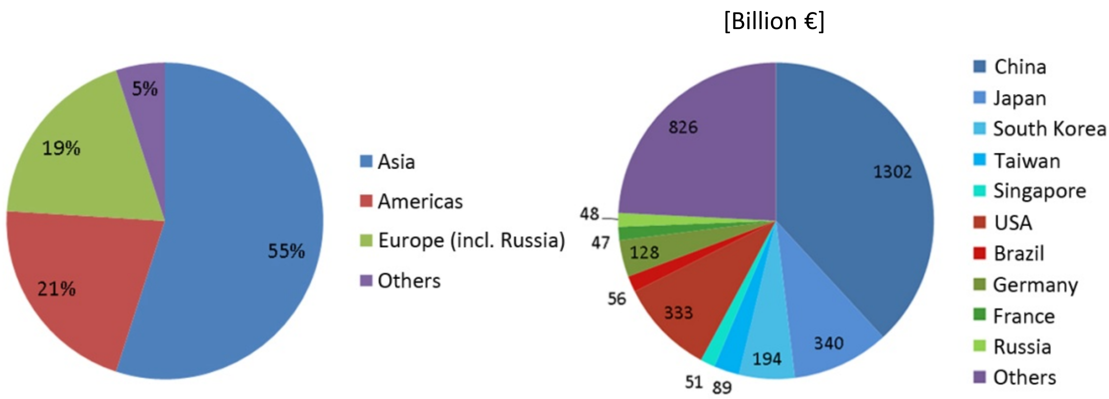


Figure 4.6. Global electrical production by regions and countries: market share by continent, 2011 (left), and market share of the top ten producer countries in billion Euro, 2010 (right).
(Own work; Data source: Gontermann, 2012)

The economy data do not provide, to which countries only the added values are related and which country is the real producer. For environmental considerations the manufacturing location is more of interest. In 2017, the electronics output of Asia increased to 63 % and that of Europe decreased to 12 % (Reed Electronics, 2019).

The following statistical data deal with the global components market. Figure 4.7. shows the global production of electronic components and printed circuit boards by regions and subsectors in 2015 (ZVEI, 2014a/2015a). In 2017, the total annual production volume of the electronics industry increased to 1791 billion US$ (Reed Electronics, 2019).

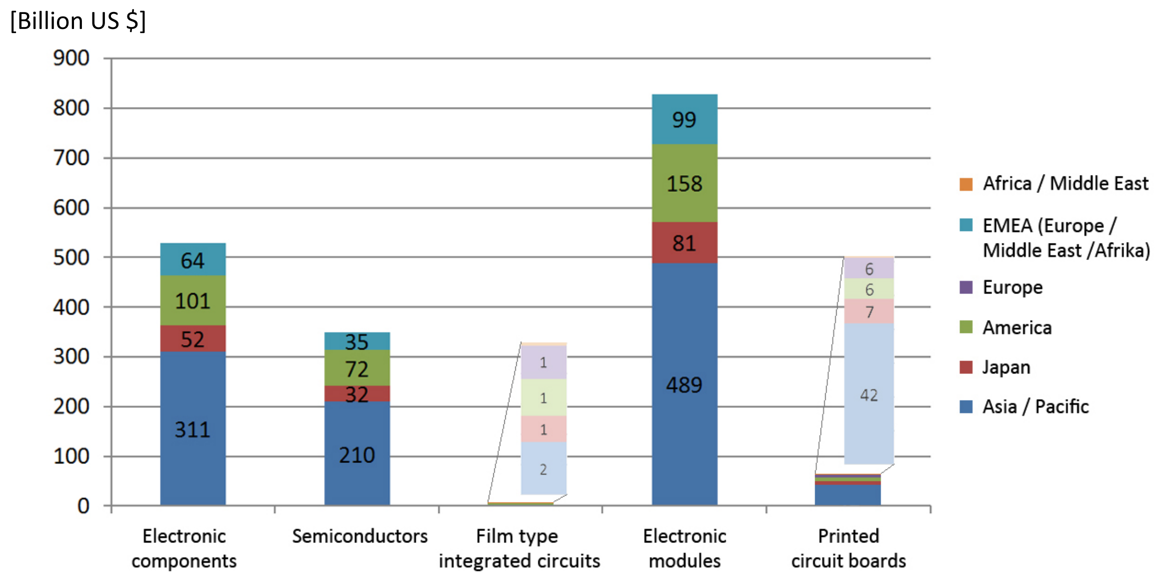


Figure 4.7. Global production of electronic components and printed circuit boards by sectors and regions in billion US $, 2015. (Own work; Data source: ZVEI, 2014a/2015a[1)])

The World Semiconductors Trade Statistics provides semiconductor market data for 2015 (WSTS, 2015). The total global semiconductor revenue is 343 billion US$. Figure 4.8. shows the market share by region. The Asia/Pacific region produces 60 % of the global manufactured semiconductors.

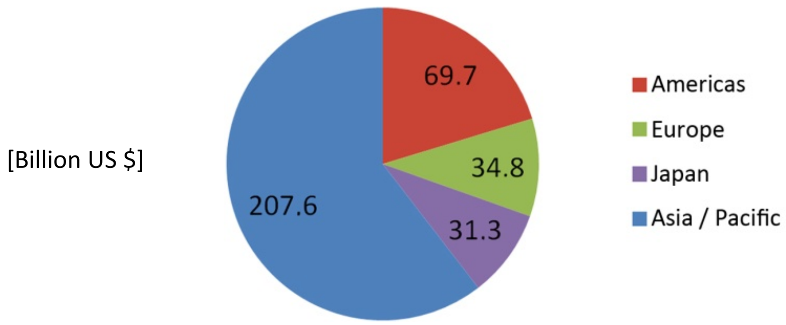


Figure 4.8. Global semiconductor market by regions in billion US $, 2015. (Own work; Data source: WSTS, 2015)

Figure 4.9. shows the global semiconductor market by sectors in 2015 (WSTS, 2015). More than 80 % of the semiconductor revenue is gained by integrated circuits.

[1)] Corrected data after consultation with ZVEI 22.05.2019.

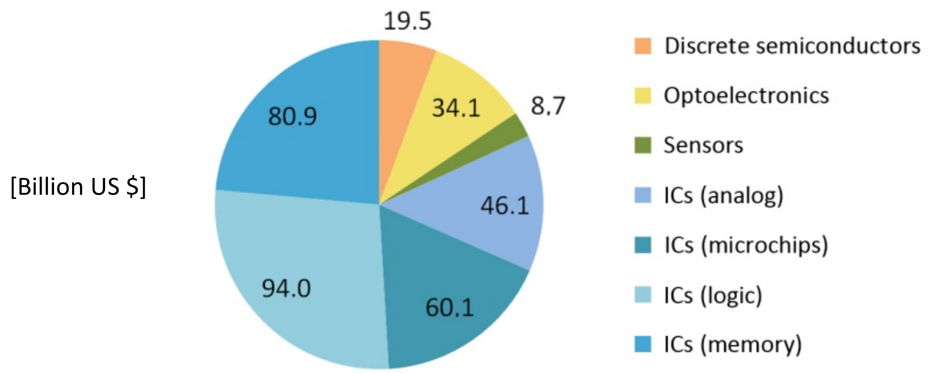


Figure 4.9. Global semiconductor market by sectors in billion US $, 2015 (WSTS, 2015).

4.2.2. Electronic Component Manufacturing in Germany

The statistical data so far considered the global market situation and sectors. European data are also available by mass in tonnes and they are presented together with e–waste data in section 4.3. Regional economic data are available for Germany. The Zentralverband Elektrotechnik- und Elektronikindustrie e.V. (ZVEI) provides detailed market data for the electrical industry. Figure 4.10. shows as an example the electronic component market in Germany by sectors and application (ZVEI, 2014a/2015a).

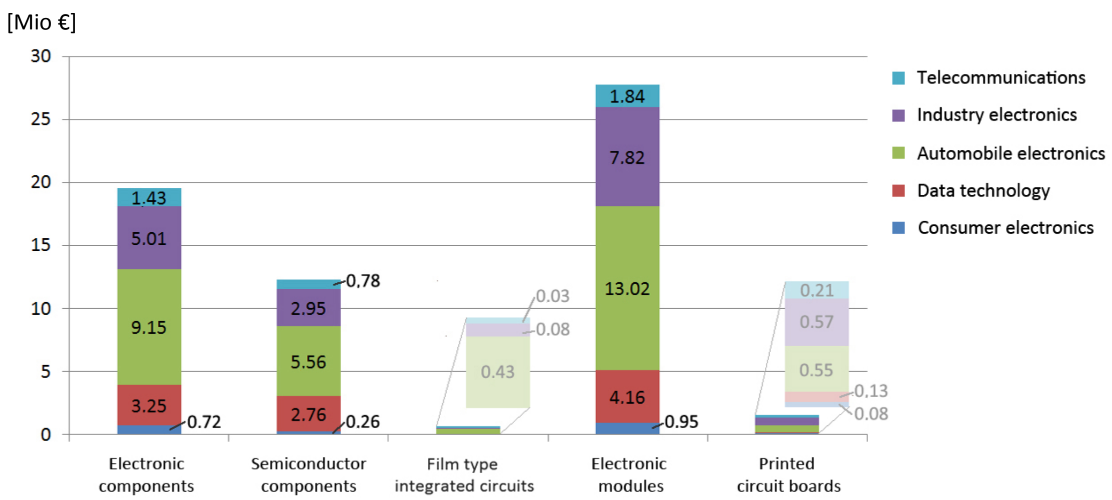


Figure 4.10. German production of electronic components and printed circuit boards by sectors and application in Mio. Euro, 2015. (Own work; Data source: ZVEI, 2014a/2015a)

The total electronic component revenue is 62 billion Euro in 2015. It is noticeable that the application of each sector is dominated by the automotive industry. Film type integrated circuits, such as thick film, thin film, or LTCC (low temperature co–fired ceramics) technologies are applied in the automotive industry in Germany by 80 %.

The market statistics so far refer to economical country data, which do not convey the relation of direct production data, import, or export. For environmental impact assessment the production sites and transports of goods are of interest. For Germany, there are comparable data of production, import, and export of the electrical industry available, as shown in table 4.7 for the years 2010, 2013, and 2014 (ZVEI, 2015b). The values show a high international transfer of goods.

Table 4.7. Economic performance of the electrical industry in Germany.

	2010 billion €	2013 billion €	2014 billion €
Production	143	138	144
Revenue	170	167	172
Export	160	158	166
Import	137	134	145

4.2.3. Electrical Industry Sectors in Austria

A further regional statistic is provided by the Bank Austria Economics & Market Analysis Austria. Figure 4.11. shows the Austrian electrical market according to the ÖNACE classification (Wolf, 2014; Statistik Austria, 2008). The figure shows that the subsector motors & generators has a share of 41 %, which is rarely outlined in statistics. The Austrian total revenue is 16.2 billion Euro in 2013, which corresponds to 3 % of the EU27 sector revenue.

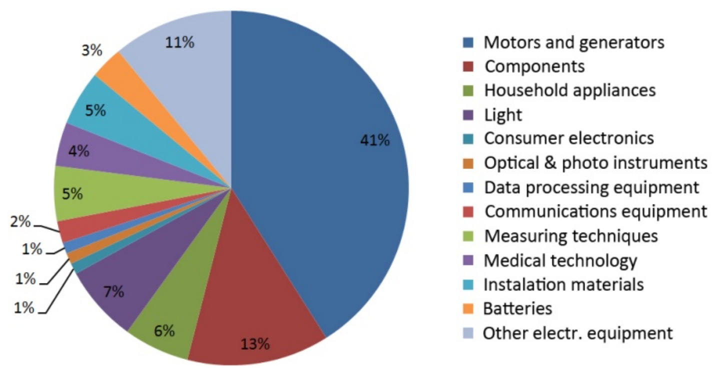


Figure 4.11. Sectors and market share of the Austrian electrical industry, 2013. (Own work; Data source: Wolf, 2014)

The statistics so far refer to economic data, which do not provide mass flows necessary for environmental assessment, but they reflect the global relations of production and they are closely related to mass transport flows and harmful substance flows.

4.3. Waste of Electric and Electronic Equipment – WEEE

4.3.1. Global e–Waste Generation

Concerning the end–of–life stage of electrical products, the Global e–Waste Monitor by the United Nations University provides detailed data of the international e–waste generation (Balde et al., 2015). The annual economic value of the total e–waste is estimated at 48 billion Euro. The gold content, as example for precious metals, is about 300 tonnes. Figure 4.12. shows the global distribution of e–waste in 2014.

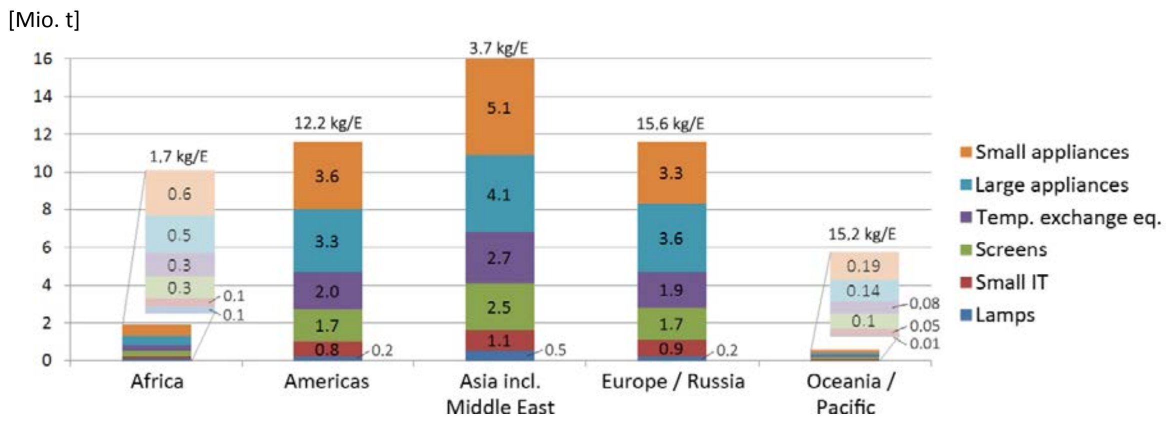


Figure 4.12. Global e–waste generation by regions and sectors in million tonnes and per capita in kg/E, 2014. (Own work; Data source: Balde et al., 2015)

The mass of the total global e–waste is estimated at 41.8 million tonnes, thereof, 6.5 tonnes are documented and recycled with the highest technical standard. The annual amount of harmful substances of e–waste is estimated at 2.2 million tonnes of lead glass, 0.3 million tonnes of batteries, and 4400 tonnes of ozone depleting substances.

A comparison with the producer countries according to figure 4.6. shows that, statistically, about 30 % of the Asian electrical industry is equally transferred to the European and American market. The import/export data of table 4.7. however suggests, that the international market transfers of electrical product are much higher than 30 %.

Figure 4.13. shows the global e–waste distribution by electrical sectors according to figure 4.12 (Balde et al., 2015).

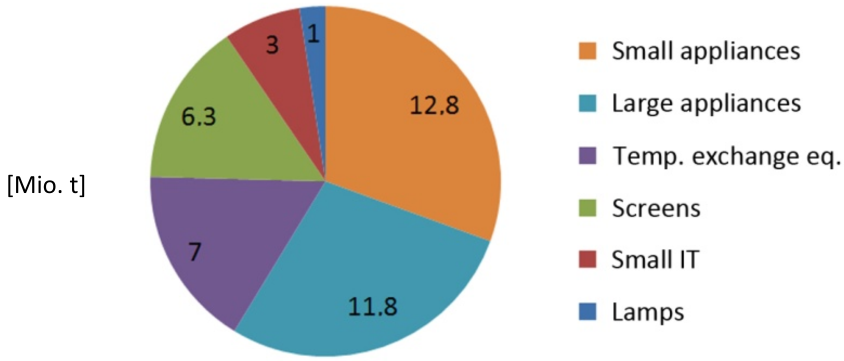


Figure 4.13. Sectors of the global e–waste in million tonnes.
(Own work; Data source: Balde et al., 2015)

4.3.2. European Union e–Waste Generation

The European Union provides detailed statistics for electrical and electronic equipment (EEE) put on the market and collected e–waste (WEEE) based on the reporting obligations according to the e–waste directive (EU Directive, 2012). Figure 4.14. shows the country data for the year 2016 (Eurostat, 2019).

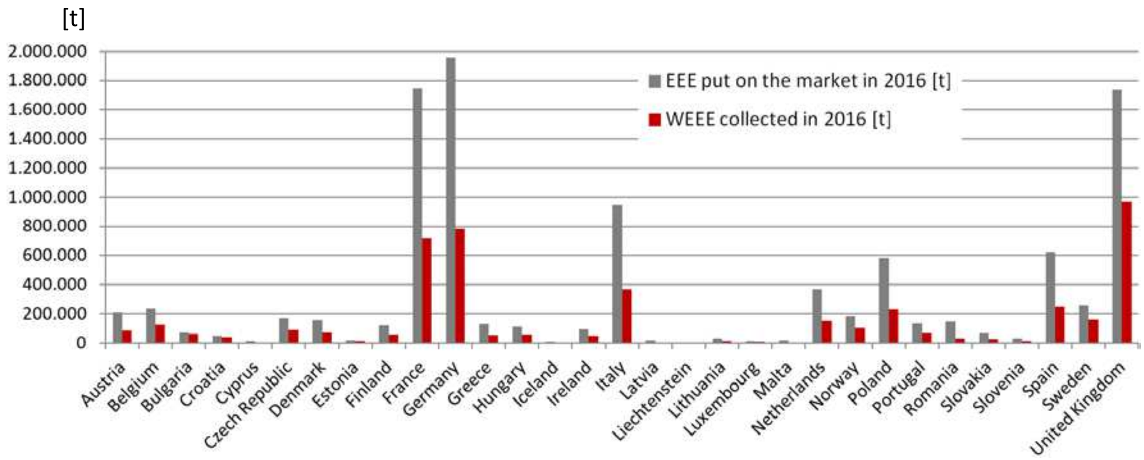


Figure 4.14. EEE put on the market and WEEE in the European Union in 2016.
(Own work; Data source: Eurostat, 2019)

In 2016, the collected e–waste in the EU28+Norway amounts to 4.5 million tonnes or 9 kg per capita. A comparison with figure 4.12. shows that in Europe, the collected e–waste is about 55 % of the estimated generated e–waste. The electrical equipment put on the market is 10.3 million tonnes or 20 kg per capita, i.e. twice as much as the collected and slightly higher than the estimated e–waste.

4.3.3. Austrian e–Waste Generation

Regional e–waste data for Austria are collected and evaluated by the Elektroaltgeräte Koordinierungsstelle Austria (EAK, 2019). Figure 4.15. shows electrical and electronic equipment put on the market and collected e–waste in Austria by sectors for the year 2016 (Eurostat, 2019).

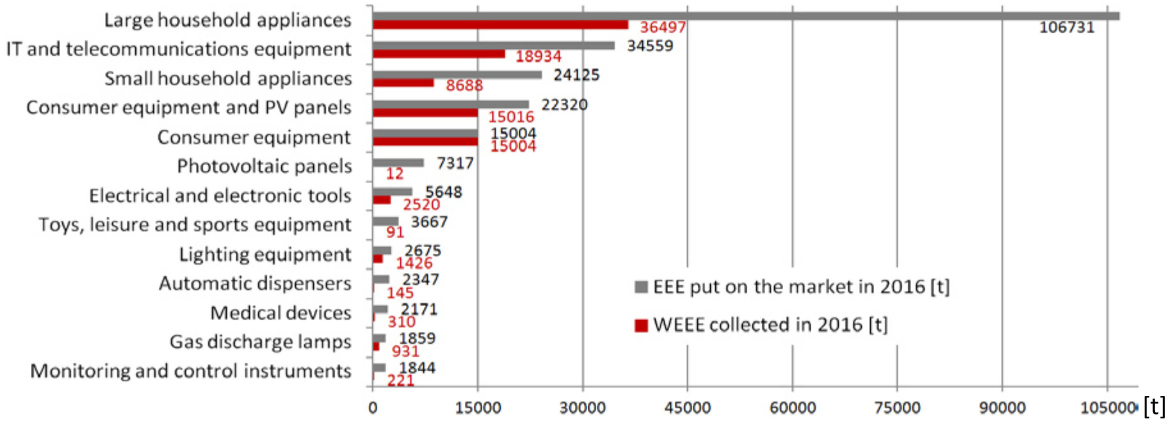


Figure 4.15. EEE put on the market and collected WEEE by sectors in Austria, 2016.
(Own work; Data source: Eurostat, 2019)

The total collected e–waste amounts to 112,665 tonnes or 13 kg per capita in 2017. During the same period 209,706 tonnes or 24 kg per capita are put on the market. Figure 4.16. shows the sectoral distribution of the e–waste in percent. Large and small household appliances have a total share of 67 %, IT&T and consumer equipment further 28 % (EAK, 2018). There was no collection of PV panels and automatic dispensers.

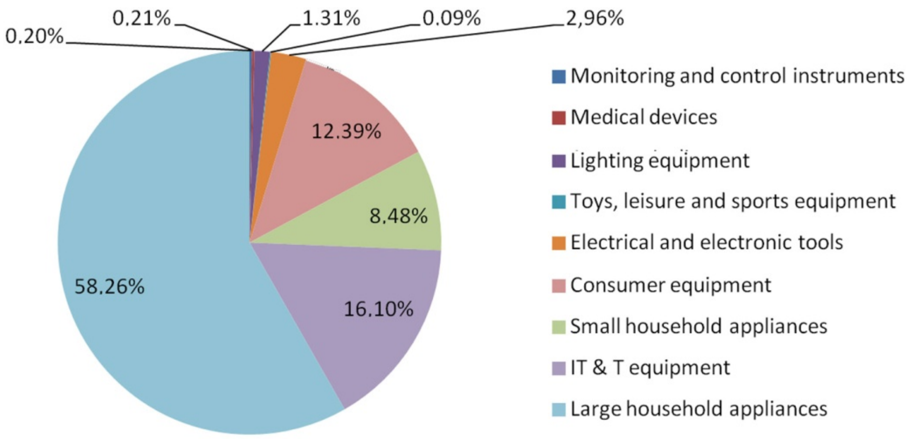


Figure 4.16. Sectoral distribution of e–waste in Austria in percent, 2017.
(Own work; Data source: EAK, 2018)

The e–waste is divided in "historical" and "new" WEEE. "Historical" e–waste includes electrical devices which are put on the market before the effective date of the substance restriction directive (RoHS) in 2006 (EU Directive; 2011). "New" WEEE are put on the market after 2006 and therefore, they only contain limited hazardous substances such as lead, mercury, and cadmium. Figure 4.17. shows the relations between "historical" and "new" WEEE in the collection of the year 2013. These data were evaluated for the last time in 2013. The graphic shows, that the share of "historical" equipment is very high even after seven years of legal force of the directive.

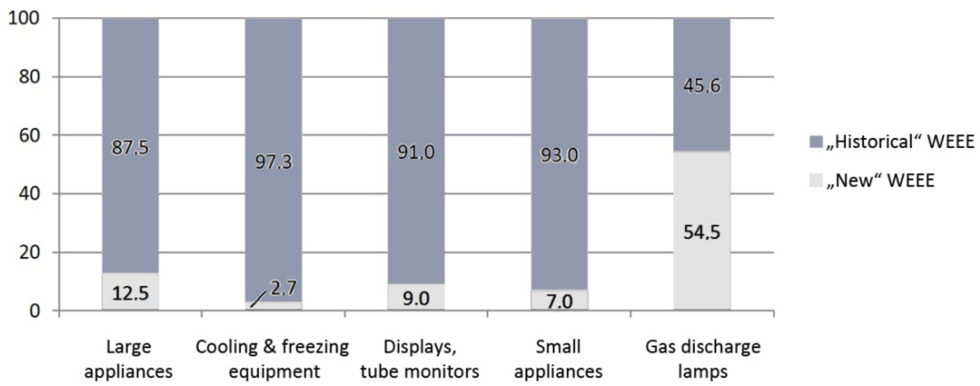


Figure 4.17. WEEE proportion containing hazardous substances in 2013. (Own work; Data source: EAK, 2014)

4.4. References

Balde, C.P.; Wang, F.; Kuehr, R.; Huisman, J. (2015). The Global e-Waste Monitor 2014. United Nations University, IAS– SCYCLE, Bonn, Germany, 2015.

BIO by Deloitte, Oeko-Institut and ERA Technology (2014). Preparatory Study to establish the Ecodesign Working Plan 2015–2017 implementing Directive 2009/125/EC. Task 2. Draft Final Report prepared for the European Commission (DG ENTR), 2014.

EAK (2014). Tätigkeitsbericht 2013. Elektroaltgeräte--Koordinierungsstelle Austria GmbH, 2014.

EAK (2015). Tätigkeitsbericht 2014. Elektroaltgeräte--Koordinierungsstelle Austria GmbH, 2015.

EAK (2018). Tätigkeitsbericht 2017. Elektroaltgeräte--Koordinierungsstelle Austria GmbH, 2018.

EAK (2019). Elektroaltgeräte--Koordinierungsstelle Austria GmbH. https://www.eak-austria.at/ [04/2019].

EU Directive (2000). Directive 2000/53/EC of the European Parliament and of the Council of 18 September 2000 on end-of life vehicles.

EU Directive (2009). Directive 2009/125/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for the setting of ecodesign requirements for energy-related products (recast).

EU Directive (2011). Directive 2011/65/EU of the European Parliament and of the Council of 8 June 2011 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (recast).

EU Directive (2012). Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE) (recast).

Eurostat (2019). Waste electrical and electronic equipment (WEEE) by waste management operations [env_waselee]. Last update: 27.03.19. Extracted on 20.05.19.

GlobeNewswire, Inc. (2019). Global Printed Circuit Board (PCB) Market to Witness a CAGR of 4.2 % during 2018-2024: Energias Market Research. Energias Market Research. January 03, 2019 01:56 ET. https://www.globenewswire.com/news-release/2019/01/03/1679894/0/en/Global-Printed-Circuit-Board-PCB-Market-to-Witness-a-CAGR-of-4-2-during-2018-2024-Energias-Market-Research.html [05/2019].

Gontermann, Andreas (2012). Die deutsche Elektroindustrie im globalen Vergleich. ifo Schnelldienst, 18, 2012, p. 22–26.

NACE (2006). Regulation (EC) No 1893/2006 of the European Parliament and of the Council of 20 December 2006 establishing the statistical classification of economic activities NACE Revision 2 and amending Council Regulation (EEC) No 3037/90 as well as certain EC Regulations on specific statistical domains.

Reed Electronics Research (2015). Yearbook of World Electronics Data 2014/2015/2016 Volume 1-4. Reed Electronics Research, UK, 2015.

Reed Electronics Research (2019). Yearbook of World Electronics Data 2018. Reed Electronics Research, UK, 2019. http://www.rer.co.uk/image/data/Downloads/Yearbook%2018%20%204%20Page%20-%20PPI.pdf [05/2019].

Statistik Austria, (2008). Systematik der Wirtschaftstätigkeiten. ÖNACE 2008.

United Nations Statistics Division (2008). International Standard Industrial Classification of All Economic Activities, Rev.4, 2008.

Wolf, Günter (2014). Branchenbericht: Elektroindustrie. Bank Austria Economics & Market Analysis Austria, 2014.

WSTS (2015). Historical Billings Report: total semiconductor revenues by the four regions Americas, Europe, Japan and Asia Pacific. World Semiconductors Trade Statistics, 2015.

ZVEI (2014a). Elektroindustrie weltweit. Branchenstruktur und Entwicklung. Zentralverband Elektrotechnik- und Elektronikindustrie– ZVEI, 2014, Frankfurt a. M.

ZVEI (2014b). Aktuelle Markttrends und Aussichten. Leiterplatten, integrierte Schicht-schaltungen, elektronische Baugruppen. ZVEI–Forum electronica 2014, 13. November 2014.

ZVEI (2015a). Wachstum bei elektronischen Bauelementen. Presseinformation, 19.05.2015. ZVEI– Zentralverband Elektrotechnik- und Elektronikindustrie e.V.; Frankfurt am Main, Germany, 2015.

ZVEI (2015b). Elektroindustrie in Zahlen. ZVEI – Zentralverband Elektrotechnik- und Elektronik-industrie, Frankfurt a. M., Mai 2015.

ZVEI (2018). Welt-Elektromarkt – Ausblick bis 2019. ZVEI – Zentralverband Elektrotechnik- und Elektronikindustrie, Frankfurt a. M., Juli 2018.

5. Market Development and Consequences on End–of–Life Management of Photovoltaic Implementation in Europe

Abstract: *The construction boom of photovoltaic (PV) plants in Europe has reached its peak from 2010 to 2012, when 50 % of the cumulated PV power in 2017 was installed. This work investigates the distribution of free–field PV plants in the EU28 by analysing of statistical data and Google mapping. In the region of South Moravia, Czech Republic, all free–field and additionally all roof–top plants in and around Brno are mapped and analyzed in detail to evaluate the distribution of PV modules between roof–top and free field installations. The study addresses the market development of PV plants in the EU28 and the relations of urban and rural PV expansion and energy production on local level. Further, hazardous material flows and future environmental pressures of PV modules are investigated. At the end of 2017, the cumulative installed PV power in the EU28 was 115 GW_p and the free-field plants are distributed throughout Europe not correlated with the solar irradiation. Particularly countries in southwest and southeast Europe have still a large expansion potential. The local in–depth study shows that the roof–top plants contribute less than 5 % to the total PV electrical energy generation. Hazardous substances in PV modules might lead to future widespread environmental problems particularly in rural areas and the end–of–life logistics requires greater attention.*

5.1. Introduction

The European 2009 Renewables Directive (EU Directive, 2009) obliges Member States to reach together a share of 20 % of energy from renewable sources by 2020. In 2017, a proposal for a recast of the Renewables Directive was published by the European Commission, which foresees a share of renewable energy sources of 27 % by 2030. Austria has a mandatory renewable energy target of 34 % of the national gross final energy consumption by 2020 (EC Proposal, 2017).

Besides wind energy, hydropower, and geothermal energy, solar energy is based on a high level of direct land consumption, which characterises the renewable energy market. A further significant part of land consumption as energy resource is caused by biomass in terms of the cultivation of corn for biogas, rapeseed for biofuel, or poplars in short-rotation forestry for block heat and power stations. (Dumke, 2017).

Since the EU Renewables Directive entered into force a considerable number of free–field PV solar parks using different technologies, scales and locations was built all over Europe over the last years. According to the actual Photovoltaic Barometer (Eurobserv'er, 2018), in total 106.6 GW_p photovoltaic power were installed and 114 TWh photovoltaic electrical energy were generated in the EU28 in 2017. The Fraunhofer Institute determined the cumulated photovoltaic power in Europe with 115 GW_p at the end of 2017 (Fraunhofer, 2019). The reported country specific cumulative values are not distinguished between roof top plants, facade integrated plants, and free–field plants.

In about 15 years, that is 2035, the first large scale generation of PV modules throughout Europe will reach its end of life. Since 2012 the waste management of PV modules is covered by the recast of the EU WEEE–Directive (EU Directive, 2012). In 2016, 12.3 million tonnes electronic waste (e–waste) was generated in Europe including Russia. Thereof, only 35 % were documented to be collected and recycled (Baldé et al., 2017). The predicted lifetime of PV modules is more than 25 years. In 2050 a total amount of 9.5 million tonnes end–of–life PV modules is expected in Europe (Monier & Hestin, 2011). Best practice treatment and recycling of photovoltaic panels including heavy metals are still under development and an economically viable waste management of all PV module technologies cannot yet be assessed.

Besides recovery and recycling of the collected modules, the deconstruction and collection logistics and their financial efforts for small scale roof–top plants with 5 kW_p or large scale solar fields with 100 MW_p are very different. There is no information about the numerical balance between PV modules in urban areas and modules installed in free–field plants. Figures 5.1. to 5.3. show examples of small, medium, and large scale free–field PV plants in different European regions.

Figure 5.1. Free–field photovoltaic plant with 3 MW_p in the mountains in the Peloponnese region, Greece (Google Earth, 2019; edited).

Figure 5.2. Small free–field and roof–top plant in the Tuscany region, Italy (Google Earth, 2019; edited).

Figure 5.3. Free–field photovoltaic plant with 145 MW_p installed on an area of 240 ha in Neuhardenberg, Germany (ENFO, 2019).

The aim of this work is to investigate the amount, location, scale and distribution of free–field PV plants in Europe by statistical data analysis and Google mapping to get an overview on the current environmental and future end–of–life situation. Furthermore, the relation between roof–top plants and free–field plants is investigated by a local detailed analysis in South Moravia, Czech Republic. The region is chosen as it has a representative size, an effective PV expansion, and the satellite images of the entire area were of the same actual date with high image resolution.

In the common frame of the thesis, the study addresses the market development of PV plants representing one type of renewable energy generation systems in Europe and its correlation to the regional energy harvesting potential. Further, the material turnover on local level in urban and rural areas is evaluated and compared. Future environmental pressures by material flows of toxic substances in PV modules and their consequences on end–of–life management are investigated and discussed.

5.2. Materials and Methods

5.2.1. Statistical Data of PV Plants in the EU

The statistical data of installed photovoltaic power in the EU countries from 2008 to 2017 are taken from the annual renewables reports of the EU supported Eur'Obersever (Eurobserv'er, 2018).

The cumulated installed power, the installed power per capita, and the installed power per surface area for each country are determined and graphically compared. For this case the country's land area and population data are taken from the European Statistical Office (Eurostat, 2019).

A further set of data is used to visualize the development of the installed power of each EU country from 2008 to 2017.

5.2.2. Mapping of Free–Field PV Plants in Europe

The free–field photovoltaic plants are mapped in Google Earth throughout Europe. In addition, strikingly large roof–top systems are mapped. Following sources are used to locate and map the free–field PV plants:

- Manufacturer's references by web search
- Open Street Map ITOworld (ITOworld, 2014)
- Energy Register of Greece (Energy Register, 2014)
- Direct search in Google Earth: The findings were dependent on the date of the satellite image. According to the statistical data the main search focus was in Greece, Italy, Spain, Czech Republic, Slovakia with large areas with image data of the year 2013/14. The images of Germany, Austria, France, UK, and Romania were only partly up–to–date and not representative for findings. The image resolution was high enough to clearly identify the PV plants with its typical shapes, paths and inverter stations. In few cases there could be a possibility of confusion with foil tunnels or greenhouses.

ITOworld and Energy Register do not include complete data sets of the regional PV distribution, but they are useful sources to search deeper in the respective region.

The distribution of the found free–field PV plants is compared with the map of Photovoltaic Solar Electricity Potential in European Countries provided by the European Union (JRC, 2014).

5.2.3. Free–Field and roof–top PV Plants in South Moravia, CZ

The Czech South Moravia region in Central Europe is investigated in detail to estimate the relation between urban roof–top plants and rural free–field plants and their respective contribution to PV energy production and the distribution of future waste modules. South Moravia is an EU NUTS 3 territorial unit (EC Regulation, 2018). The region is chosen as it has an effective PV expansion and the high resolution satellite images were of the same actual date.

The area is 7200 km^2 with a population of 1.2 million inhabitants, whereby one–third of the population lives in the region capital Brno. The free–field plants of the region South Moravia and also the roof–top plants in the city of Brno are mapped. Following data are collected and analysed by Google mapping:

- Total number of free–field solar plants in the region of South Moravia
- Total number of roof–top plants in Brno (city boundary) and number of modules per PV plant

The size of roof–top plants are divided into four categories, single–family houses up to 20 modules, multi–family houses or farm houses with 21–70 modules, mid scale business or communal buildings with 71–250 modules, and large scale plants with > 250 modules. The share of the categories is compared to the total amount of installed roof–top modules and free–field plants, respectively.

5.2.4. Environmental Pressure of Hazardous Substances in PV Modules

Photovoltaic modules contain a number of hazardous and toxic substances. Based on literature, the material composition particularly of heavy metals in different PV module technologies and the future consequences and environmental pressure for rural areas are discussed.

5.3. Installed Power and Distribution of PV Plants in the EU28

The statistical data of the Eurobserv’er Photovoltaic Barometer on the installed power of PV plants are graphically prepared (Eurobserv’er, 2018). Figure 5.4 shows the results of the EU28 per country in three expressions: total amount, per capita, and per surface area.

Germany owns the highest installed power in total as well as per capita. 86 % of the total installed power is shared by six countries, Germany, Italy, UK, France, Spain, and Belgium, in that order. The PV expansion is characterised by free–field PV plants, only Belgium has a remarkable high density of roof–top plants in all scales throughout the country, which is the result of a survey in Google Earth.

The installed power per capita shows a different picture. Germany again has the highest installed power per capita, followed by Belgium, Italy, Malta, Luxemburg, Greece, and the Czech Republic. The installed power per surface area is the highest by far in Malta, followed by Belgium, Germany, The Netherlands, and Italy.

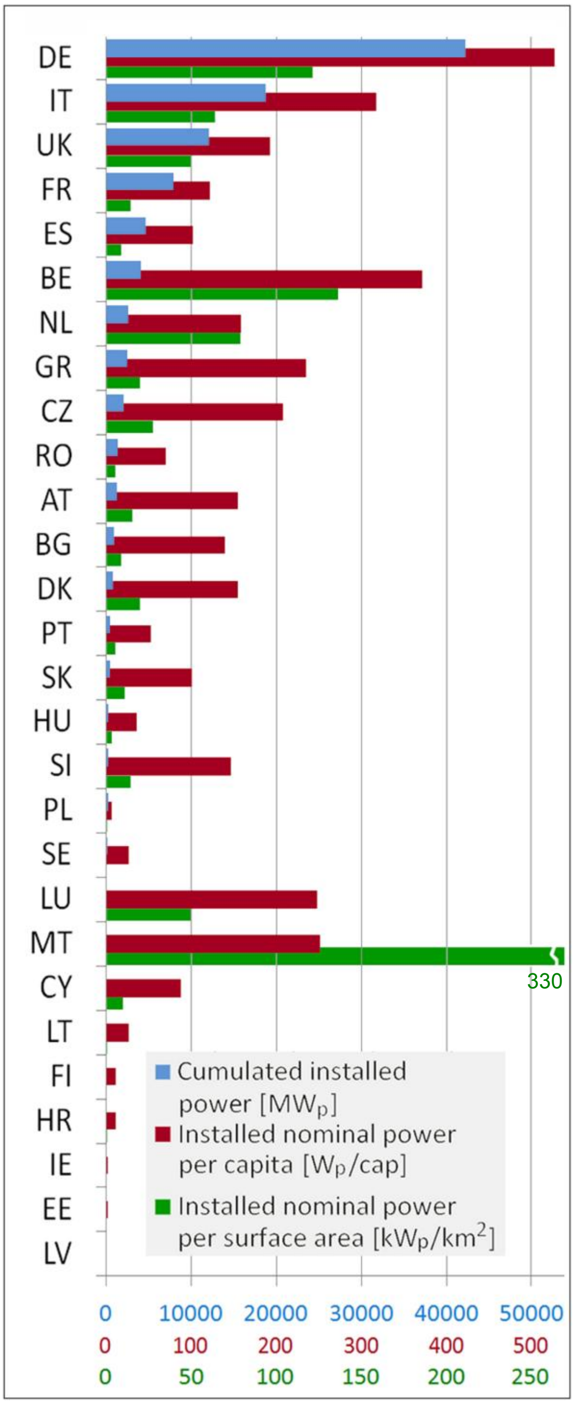


Figure 5.4. Cumulated installed photovoltaic power in the EU28 by country, as at Dec. 2017 (data source: Eur'Obsever, 2018).

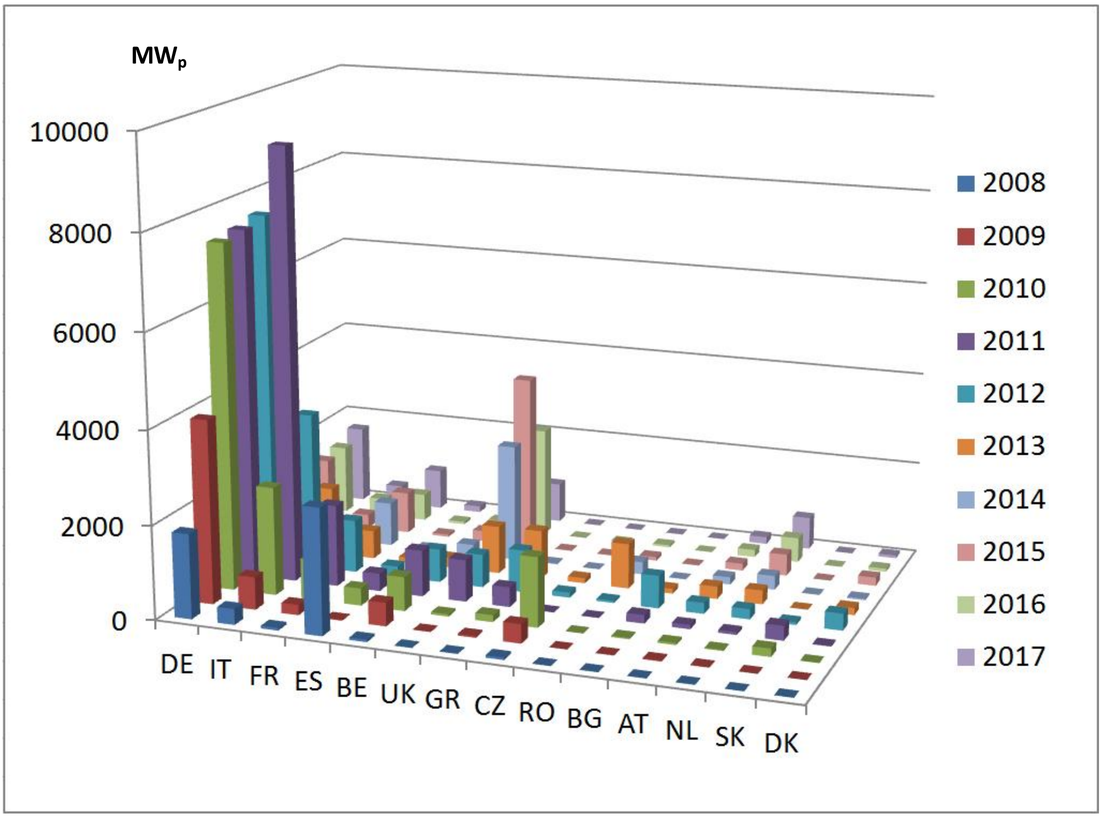
The total results are notable, because the global irradiation in Germany, Belgium, and The Netherlands is about one third less than in Italy, Greece, Spain and Portugal (see Figure 5.7.). Figure 5.4. together with the global irradiation map in figure 5.7. indicate, that there is still a high potential to exploit solar energy particularly in the Southwest European countries.

Figure 5.5. shows the course of the annual installations of PV plants for each EU country based on the statistical data of the Eurobserv'er Photovoltaic Barometer from 2008 to 2017 (Eurobserv'er, 2010-2018).

In Germany, the PV expansion peak was from 2010 to 2012, where 53 % of the national cumulative power was installed. In Italy, 49 % of the national cumulative power was installed within one year, in 2011. France had a slight expansion peak in 2011 and in the following years a continuous medium scale annual PV installation. Spain had an expansion peak in 2008, where 56 % of the national cumulative PV power was installed. Belgium installed 71 % of the national PV power from 2009 to 2012. Greece installed 58 % of the national PV power from 2011 to 2013. The Czech Republic had an expansion peak in 2010, where 68 % was installed within one year.

UK started a significant PV expansion in 2011 with a peak from 2014 to 2016, where 71 % of the national cumulative power was installed. Bulgaria installed 69 % of the national PV power in 2012 and Romania 68 % in 2013. The Netherlands started their significant PV expansion in 2012 with continuous increase until 2017. Austria shows a relatively constant PV expansion as of 2012 on a low level. All other countries have no significant contribution to the European PV expansion.

In most EU countries the installation activity declined due to the country specific changing support schemes. In 2011, Europe had a total annual PV installation peak with 22 GW_p and in 2012 with an annual installation of 16.7 GW_p. UK, Sweden, Hungary, Poland and Malta show significant national PV expansion activities as of 2014, whereby except of UK, the total European contribution is not significant.



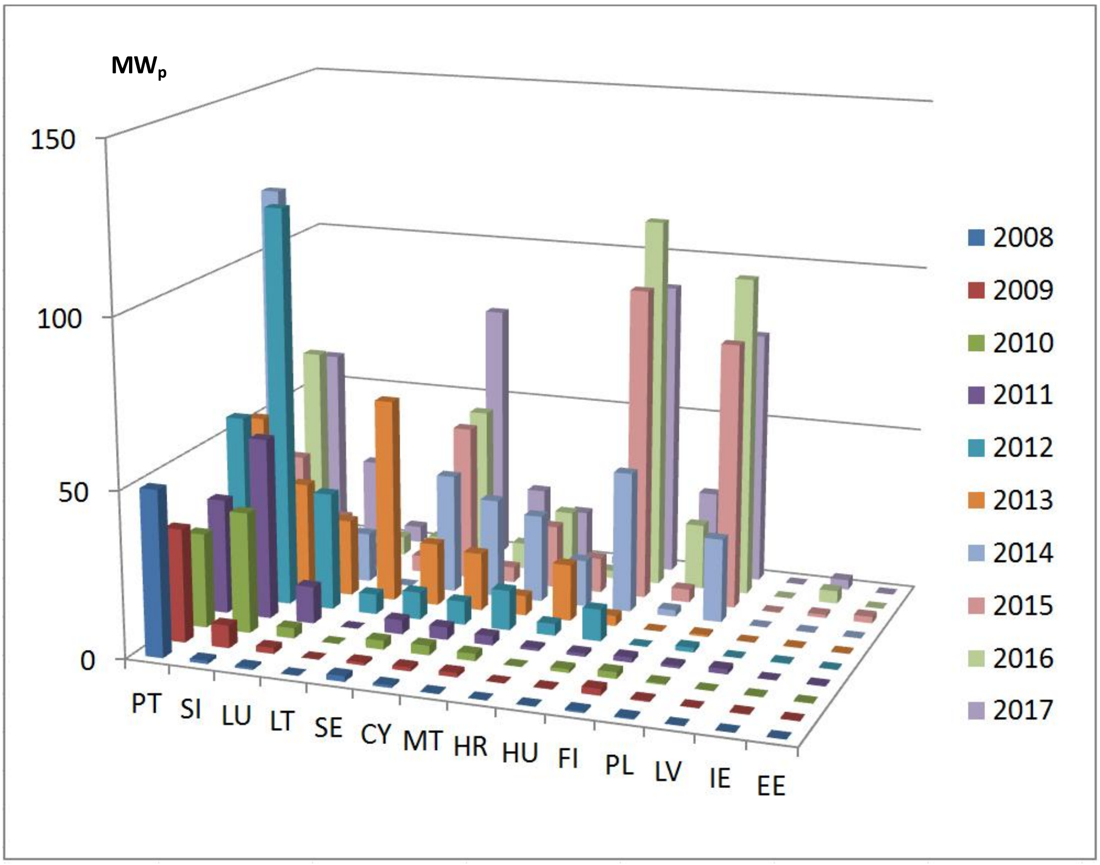


Figure 5.5. Annually installed photovoltaic power in [MW_p] in the EU28 by country from 2008 to 2017 (own work; data source: Eur´Obsever, 2018).

5.4. Free–Field PV Plants in Europe

Free–field PV plants are mapped in Google Earth all over Europe dependent on the actuality of the satellite image data. Figure 5.6. represents a snapshot of the situation in spring 2014. More than 4300 free–field solar plants are found in various scales (yellow pins). This number is significant, but there is still a large number of regions that remains unmapped. Large scale roof–top plants are also mapped which are marked with green pins. The image data of Central, East, and South Europe were actual and therefore the mapped areas show a representative PV expansion. The regions of western Germany, UK, and France are not representative compared to the data in figure 5.4. due to satellite images not being up–to–date. The mapped regions are compared to the PV solar electricity potential map in figure 5.7.

The distribution of the PV plants does not correspond to the solar radiation, but strongly to the country specific state funding. Hungary, Portugal and Romania have little activity while the solar energy exploitation could be high.

The size and locations of the free–field PV plants are different: Germany has a number of large scale plants which are located on conversion areas such as former military airports and military training areas. PV plants are also found on former landfill sites, along railways and motor ways. In Italy and Greece the PV plants tend to be smaller and thus larger in number. Middle and large scale plants are also located in not cultivated southern mountains. Nevertheless, the greater part of free–field PV plants is located on former cultivated farm land.

A review in Google Earth shows the timely trend that the first free–field PV plants were built far from settlement areas and getting closer and closer to the settlements over the years.

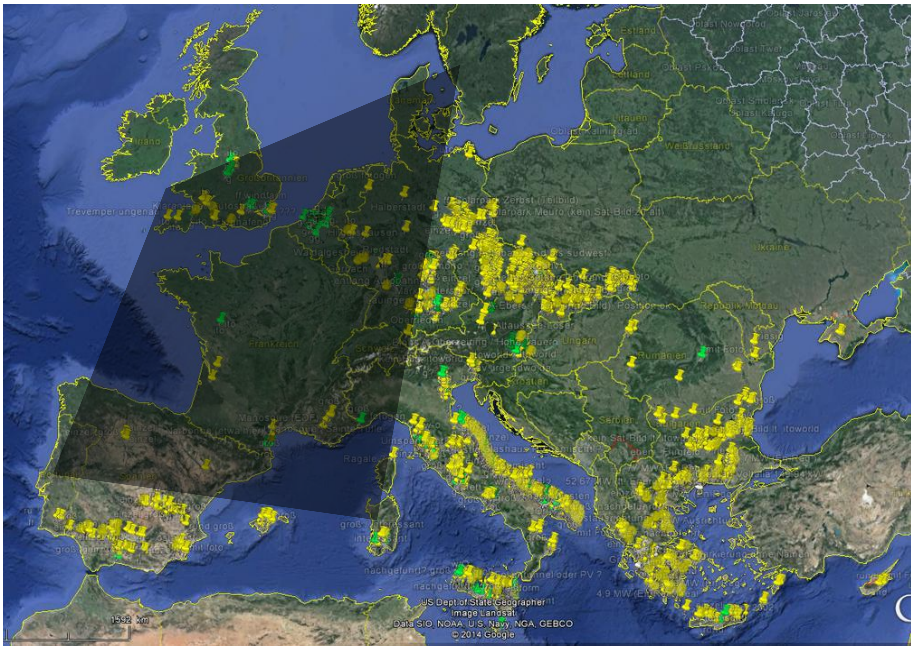


Figure 5.6. Overview of about 4300 mapped free–field PV plants (yellow pins) in Europe, dependent on the actuality of the satellite maps. The darkened area is not representative. (Google Earth, 2019, own work)

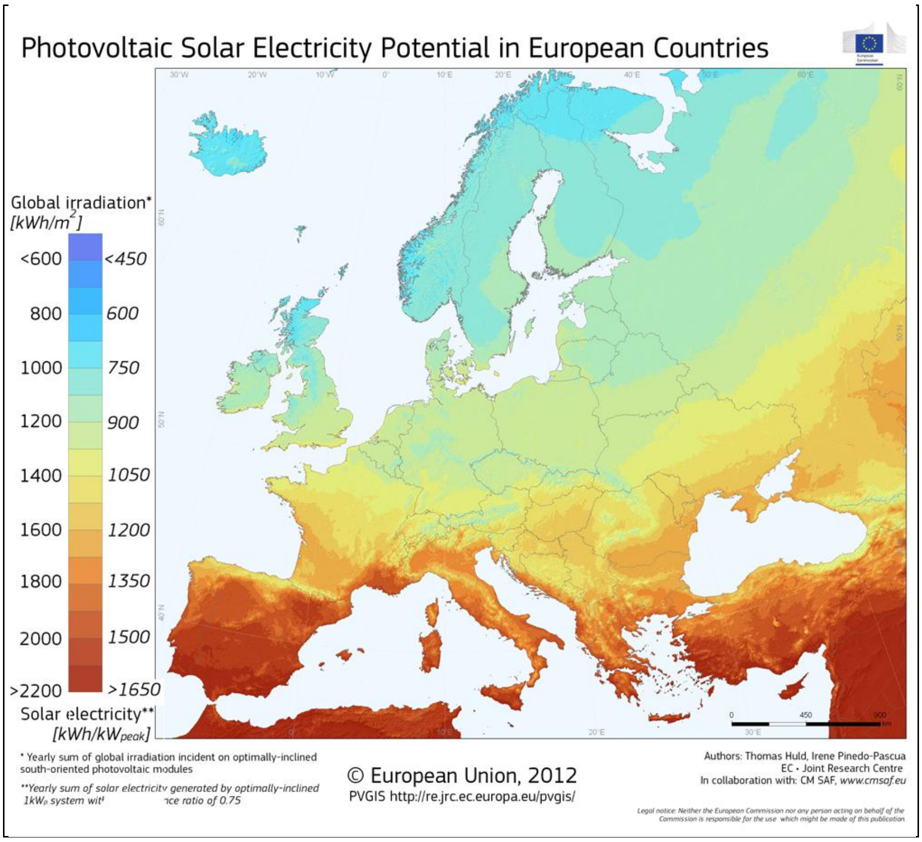


Figure 5.7. Photovoltaic solar electricity potential in European countries (Source: JRC, 2014).

5.5. Free–Field and Roof–Top PV Plants in South Moravia, CZ

In the Czech Republic, as relatively small country, the total installed PV power is small. However, South Moravia represents a region with high installation activity close to the average in Central Europe. Figure 5.8. displays all free–field PV plants in this region, and marked as dark yellow pins those of the neighbouring regions. In May 2014 the number of 185 free–field solar plants was counted. The largest one is located at Brno airport with an installed power of 20 MW_p.

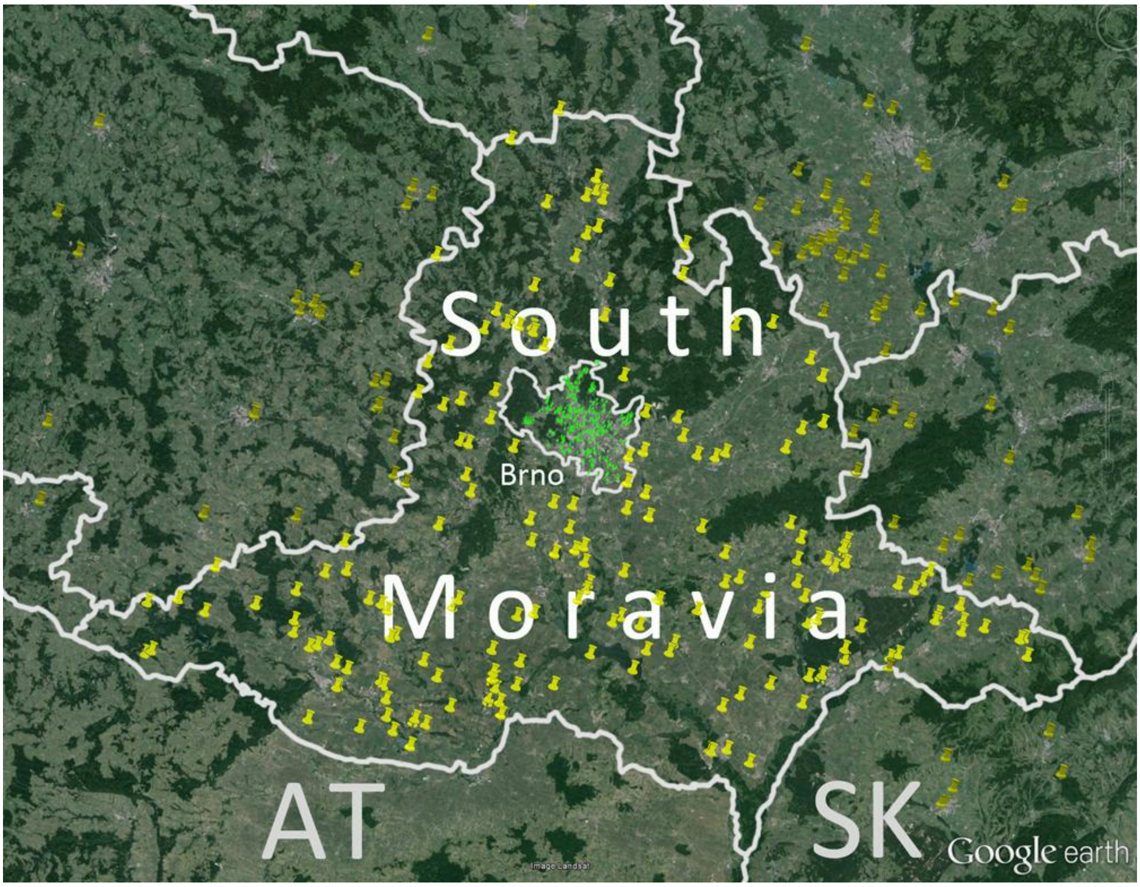


Figure 5.8. Distribution of free–field photovoltaic plants (yellow pins) and roof–top plants (green pins in Brno) in the region of South Moravia, Czech Republic, May 2014. (Google Earth, 2019; own work)

To compare the share of free–field PV plants with roof–top plants the installed solar modules in the city of Brno (385 000 inhabitants) are mapped and counted. Figure 5.9. shows the roof–top plants as of May 2014.

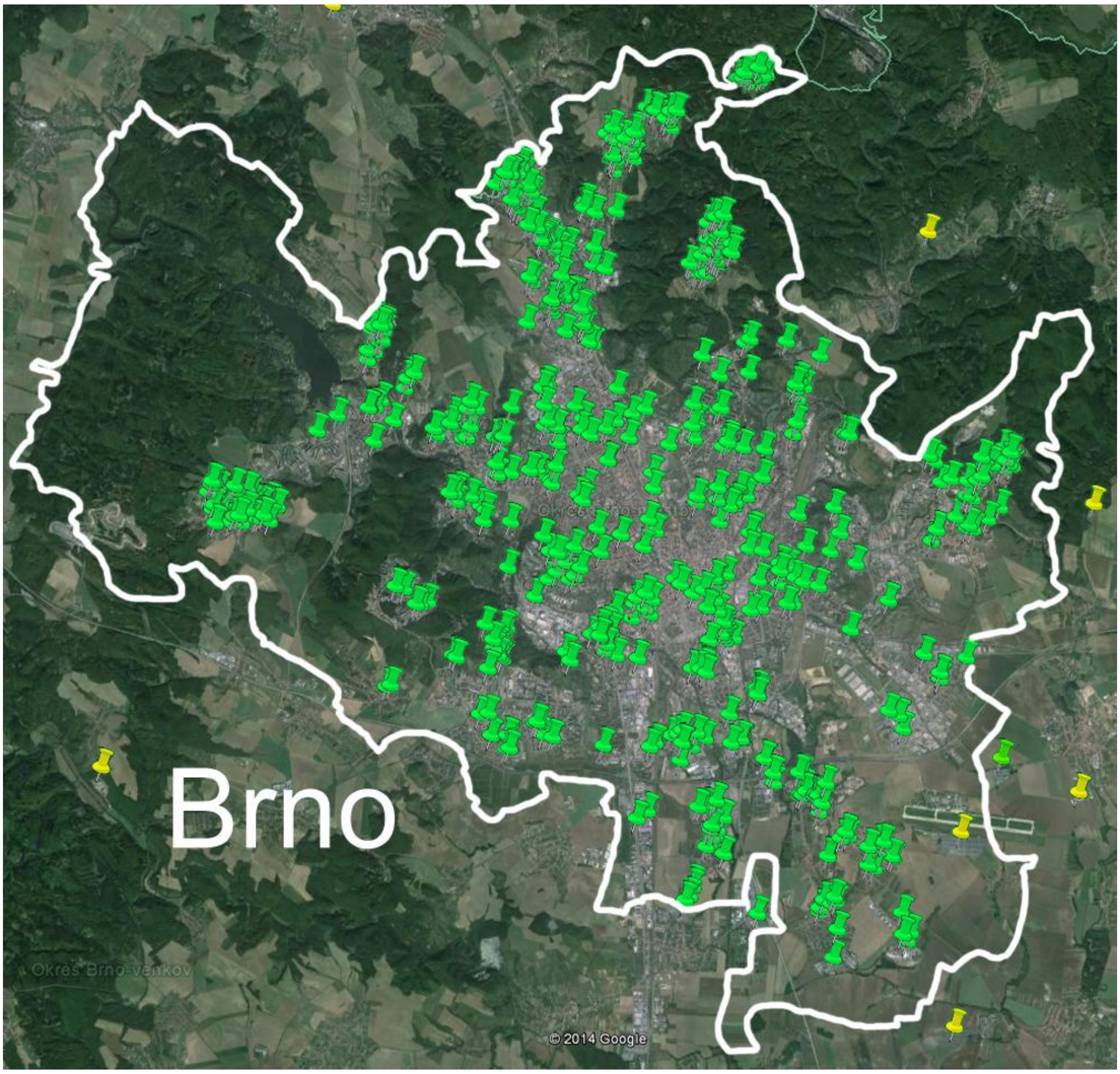


Figure 5.9. Roof–top solar plants (green pins) in Brno, May 2014; yellow pins: free-field PV plants. (Google Earth, 2014, own work)

The solar systems are distributed throughout the city. Besides the photovoltaic systems also solar thermal systems are found and partly mapped. For small scale roof–top plants it is difficult to distinguish between the two systems only by satellite image. There is no information available about the amount and location of solar thermal systems in this region. But for the entire country of the Czech Republic in 2013 the cumulated installed capacity of thermal solar collectors was 680.6 MW_{th} and 972,299 m^2, respectively (Eurobserv'er, 2014). The South Moravian share is however uncertain. Table 5.1. lists the relevant results of the data collection.

Table 5.1. Determined roof–top solar systems in Brno, May 2014.

Total number of solar modules	Number of PV systems with			
	2–20 modules	21–70 modules	71–250 modules	> 250 modules
45000	151	122	80	24

62 % of the counted modules belong to large scale roof–top PV systems and 23 % to the next smaller category. This means, that the large number of one/two family house roof–top plants have a share of only 15 % of the total roof–top PV electrical energy generation.

To determine the relation between roof–top plants and free–field plants the module number of the 20 MW_p PV plant in Brno Airport is evaluated. It is constructed by 85 000 modules of different sizes. Therefore, all roof–top plants of Brno with 45 000 modules hold almost half the installed power of one large free–field PV plant in the region. Based on 40 roughly measured areas of randomly selected free–field PV plants in South Moravia, the total proportion of roof–top PV installations and the respective electrical energy production is estimated. Compared with all 185 free–field PV plants in the region of South Moravia, the roof–top PV plants in Brno have an estimated energy contribution of 3–5 %. In this current status the urban PV electrical energy generation is insignificant.

5.6. Environmental Pressure of Hazardous Substances in PV Modules

Photovoltaic modules contain a number of hazardous and toxic substances. There is different information about the content of these substances per m^2. Silicon based PV modules contain 10–20 g toxic lead per module, which is part of the solder and the metallisation (Diermann, 2011). Therefore, for a standard module area of 1.6 m^2 the lead content is 6.25–12.5 g/m^2. CdTe modules contain 43.4 g CdTe and 3.52 g CdSe per m^2 (Jungbluth et al., 2009). Another publication indicates, that the content of CdTe amounts to 18 g/m^2 (Bayerisches Landesamt, 2011). The results of a further study show that c–Si modules contain 6.32–6.68 g/m^2 lead, CdTe modules contain 5.56–7.88 g/m^2 cadmium and 6.24–8.4 g/m^2 tellurium, and CIGS modules contain 5.64 g/m^2 indium (Nover et al., 2017a).

A study by the University of Stuttgart from 2017 investigated whether and to what extent metals leak out of damaged PV modules. Of the four major module technologies, 5 x 5 cm^2 module samples were cut out and put into aqueous solutions of three different pH values of pH3, pH7, and pH11 for varying lengths of time. The studied materials were crystalline silicon (c–Si), amorphous silicon (a–Si), CIGS (copper indium gallium diselenide), and cadmium telluride (CdTe) modules. The results show that for CdTe module samples, up to 100 % and for CIGS modules, up to 30 % cadmium dissolve out. In case of c–Si module samples, up to 5 % Pb dissolve out. The release of hazardous substances occurs at the open edges and the dissolving process starts independently of the state of delamination of the module sample. (Nover et al., 2017a/b)

During regular operation of the PV modules risks might be low for the user and the environment. In case of mechanical damage of the module laminate and non-removal for decades, these substances can be released uncontrolled and spread into the environment. The investigations about the PV plant distribution of section 5.4. shows that by far the largest share of PV modules are located in rural areas. That means, the environmental pressure of end–of–life modules is not locally concentrated but widely spread throughout Europe.

There are large logistical differences for the end–of–life management between roof–top PV plants and free–field plants. PV modules on roofs are included in already built–up areas and do not require additional land or other infrasturcture. Their end–of–life treatment can be included in construction waste disposal and fall under the concept of "urban mining". Different to urban areas, free–field PV plants have a considerably larger area and there is no logistics up to now to deal with waste modules. To leave them on the land is a worst case option that cannot be discounted for areas, which have no agricultural or forestry value. This might be the case in southern mountainous regions or already contaminated land and land fill areas.

There is reason to fear that in 25 years there will be no budget set aside to remove the PV plants, dig out the foundations of the racks and the cables which are buried in the ground for several hundred meters or kilometers. In any case, the removal will be cost intensive and could require more financial resources than the value of the land itself. The worst case scenario is, that this land will be contaminated with glass splinters and widely spread out low concentrated lead, cadmium and other immission. Besides the development of recycling technologies, which is an actual European research focus, the waste management *before* the treatment facility gate has to be moved into the focus and it must be considered, that not *one* applicable logistic strategy is required, but a number of different strategies as there are different regions in Europe, where thousands of free–field PV plants are located. As a counterpart to "urban mining" there should be a strategy like "land mining of anthropogenic resources" developed.

5.7. Conclusion

At the end of 2017, the cumulative installed PV power in the EU28 was 115 GW_p. The construction boom of PV plants had its peak in 2011. 50 % of the actual cumulated installed power was built from 2010 to 2012 and since then the annual installations were more stable on a lower level and slightly decreasing. 86 % of the EU cumulative installed power is located in Germany, Italy, UK, France, Spain, and Belgium, whereby Germany holds the largest share with 40 % followed by Italy with 18 % and UK with 12 %. In recent years, UK, The Netherlands, Sweden, Malta, Hungary, and Poland show increasing activities in PV plant expansion.

The major part of PV electrical energy is generated by free–field plants and the plants are wide spread distributed throughout the countries. Only in Belgium large scale roof–top plants dominate the PV energy production. An in-depth study in South Moravia, CZ, shows that the share of roof–top PV plants in urban areas contributes less than 5 % to the total regional PV electrical energy generation.

The distribution of the European PV plants shows that areas with high solar irradiation are not yet sufficiently developed per area basis. This concerns Hungary, Romania, Bulgaria, Spain, and Portugal. Outside the European Union, all successor states of former Yugoslavia and Albania have still a high PV expansion potential.

Future environmental pressure is expected to be caused throughout Europe by end–of–life free–field PV plants. All module technologies contain hazardous substances such as lead and cadmium. Even with the smallest damages of the PV module laminate the substances leak and spread into the soil. Dependent on the technology, the modules can contain up to 12.5 g lead or 23 g cadmium per m^2. It is estimated that in 2016, cumulative installed PV modules in Europe contained about 11,000 t lead and 800 t cadmium (Nover et al., 2017a).

Besides the development of sustainable recycling technologies for PV modules, waste management logistics and financial means for deconstruction and collection logistics have to be moved into the focus to avoid large numbers of not removed end–of–life free–field PV plants for decades.

The future PV expansion should focus on already sealed large areas such as farm buildings, halls, and other large roof areas. It should further be considered to prescribe lead-free Si–modules in the European Union, which is technically possible.

5.8. References

Baldé, C. P.; Forti, V.; Gray, V.; Kuehr, R.; Stegmann, P. (2017). The Global E-waste Monitor–2017. United Nations University (UNU), International Telecommunication Union (ITU) & International Solid Waste Association (ISWA), Bonn/Geneva/Vienna.

Bayerisches Landesamt für Umwelt (2011). Berechnung von Immissionen beim Brand einer Photovoltaik-Anlage aus Cadmiumtellurid-Modulen. Augsburg, 2011. [German]

Diermann, Ralph (2011). Blei bleibt beliebt. Photovoltaik – Solartechnik für Installateure, Planer, Architekten. Ausgabe 04-2011. [German]

Dumke, Hartmut (2017). Erneuerbare Energien für Regionen – Flächenbedarfe und Flächenkonkurrenzen. Dissertation. Technische Universität Wien, Austria.

EC Proposal (2017). Proposal for a Directive of the European Parliament and of the Council on the promotion of the use of energy from renewable sources (recast). Brussels, 23.2.2017 COM(2016) 767 final/2 2016/0382 (COD).

EC Regulation (2018). Regulation (EC) No 1059/2003 of the European Parliament and of the Council of 26 May 2003 on the establishment of a common classification of territorial units for statistics (NUTS). Consolidated version 18.01.2018.

Energy Register (2014). http://www.en.energyregister.gr/; Map; (accessed May 5, 2014).

ENFO–Energie GmbH (2019). Solarpark Neuhardenberg. Größter Solarpark Europas – ein Projekt der ENFO, Frankfurt / Oder, Germany. https://www.enfo-energie.biz/ [04/2019].

EU Directive (2009). Directive 2009/28/EC of the European Parliament and of the Council on the promotion of the use of energy from renewable sources.

EU Directive (2012). Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE) (recast).

Eurobserv'er (2014). Solar thermal and concentrated solar power barometer – Eurobserv'er, May 2014.

Eurobserv'er (2018). Photovoltaic Barometer 2010-2018. http://www.eurobserv-er.org [02/2019].

Eurostat (2019). https://ec.europa.eu/eurostat/ [04/2019].

Fraunhofer ISE (2019). Photovoltaics Report, updated: 14 March 2019. https://www.ise.fraunhofer.de/de/veroeffentlichungen/studien/photovoltaics-report.html [03/2019].

Google Earth (2019). https://www.google.com/earth/ [04/2019].

ITOworld (2014). http://www.itoworld.com/; Maps: Electricity generation (accessed May 5, 2014).

Jungbluth, N.; Stucki, M.; Frischknecht, R. (2009). Photovoltaics. In Dones, R. (Ed.) et al., Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. ecoinvent report No. 6-XII, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, 2009. [German]

JRC-Joint Research Centre (2014). Photovoltaic Geographical Information System (PVGIS). http://re.jrc.ec.europa.eu/pvgis/ (accessed May 5, 2014).

Monier, Véronique; Hestin, Mathieu (2011). „Study on photovoltaic panels supplementing the impact assessment for a recast of the WEEE Directive. Final report 14 April 2011, Bio Intelligence Service, France, 2011.

Nover, J.; Huber, S.; Zapf-Gottwick, R.; Werner, J. H.; Feifel, C.; Koch, M.; Metzger, J. W. (2017a). Schadstofffreisetzung aus Photovoltaik–Modulen, Abschlussbericht. Universität Stuttgart. Gefördert durch das Bundesministerium für Wirtschaft und Energie, Förderkennzeichen: 0325718. 138 pages. [German]

Nover, Jessica; Zapf-Gottwick, Renate; Feifel, Carolin; u. a. (2017b). Long–term leaching of photovoltaic modules. Japanese Journal of Applied Physics 56, pp. 08MD02-1 bis 08MD02-6, doi: https://doi.org/10.7567/JJAP.56.08MD02.

6. Roof–Top Photovoltaic Plants of a Small Town in Lower Austria: Expansion Potential and End–of–Life Considerations

Abstract: *The preparation of the 2015 climate conference of the United Nations in Paris resulted in different national renewable energy scenarios for the next 35 years. The Technology Roadmap for Photovoltaic in Austria has set the target to cover 15 % of the national electrical energy consumption by photovoltaic plants by 2030. This work investigates technical and environmental characteristics of photovoltaic plants and develops a system model for roof–top photovoltaic expansion, material turnover, and PV energy production by the example of the small town Eggenburg in Lower Austria with 3500 inhabitants. The study addresses material flows, market development and future environmental pressures of electric devices on local level. The questions of regional energy balance considerations and hazardous material flows are investigated. Under current conditions, the results show, that only 60 % of the photovoltaic climate target can be achieved by roof–top plants. Within the system model and in the state of full photovoltaic plant expansion in 2030, there are 170 tonnes of modules mounted on the roofs in Eggenburg with an annual end–of–life waste generation of 8.4 tonnes of materials, including 1.4 kg cadmium compound and 12 kg lead. There is need of accurately funding strategies and research on future material flows and local logistics for end–of–life roof–top PV systems.*

6.1. Introduction

The *Paris Convention 2015* addresses the global response to the threat of climate change, together with sustainable development and efforts to eradicate poverty, by:

"*Holding the increase in the global average temperature to well below 2 °C above pre-industrial levels and to pursue efforts to limit the temperature increase to 1.5 °C above pre-industrial levels, recognizing that this would significantly reduce the risks and impacts of climate change*" (United Nations, 2015).

To achieve this goal at European level, the *Renewable Energy Directive* requires that 27 % of the final energy consumption is provided by renewable energy sources by the year 2030 (EU Directive, 2009/2016).

On national level, the *Renewable Energy Action Plan for Austria* defines targets to be reached by 2020: The final energy consumption should reach 1,100 Petajoule (PJ), whereby 34 % of the gross final energy consumption is provided by renewable energies (BMWFJ, 2010). The Action Plan further defines funding strategies for photovoltaic (PV) plants: PV systems with a nominal power < 5 kW_p (typically single–family homes) and 5–50 kW_p (agriculture and forestry) are funded by the Austrian Climate and Energy Fund

(KLIEN, 2017). The funding of other PV systems > 5 kW_p is covered by the Green Electricity Act (Ökostromgesetz, 2012).

The *Technology Roadmap for Photovoltaic in Austria* presents the following energy scenario (BMVIT, 2016): In 2050, the national gross energy consumption shall decrease from currently 1400 PJ to 864 PJ based on 100 % renewable energy. Photovoltaic systems will cover 12.4 % of the total primary energy demand and 27 % of the electrical energy demand. As an intermediate target, the roadmap defines for the year 2030 that 100 % of the electricity generation will be covered by renewable energy sources and thereof 15 % will be provided by PV systems. Figure 6.1. schematically shows the required changes for the total energy consumption and energy sources from 2015 to 2050 based on the values above.

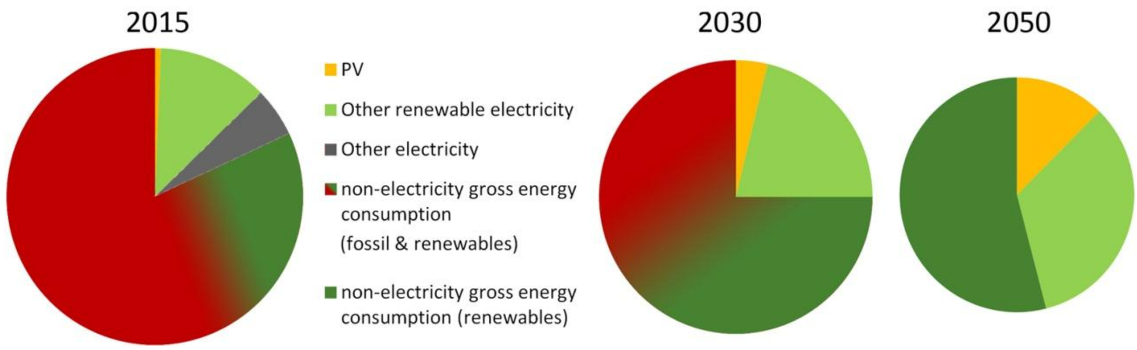


Figure 6.1. shows the Austrian energy scenarios by 2030 and 2050 compared with the situation in 2015 with respect to PV electricity generation (Statistik Austria, 2016; BMVIT, 2016; BMWFJ, 2010).

Based on these terms and conditions, this work investigates, if the PV climate target 2030 can be reached by means of roof–top PV plants by the example of the small town Eggenburg in Lower Austria. In the common frame of the research topic, the study addresses material flows of valuable and hazardous materials and substances, market development and future environmental pressures of electric devices on local level. The questions of reliable data for hazardous material flows and regional energy balance considerations are investigated. Environmental considerations are made concerning material turnover, content of toxic materials, and end–of–life management of PV modules.

6.2. Materials and Methods

The study framework includes the area within the city boundary of Eggenburg in Lower Austria. Eggenburg is a small town with about 3500 inhabitants (Statistik Austria, 2018). The building structure is characterised by one- and two-family houses, a historic city center with long farm buildings in the backyard both with mainly saddle roofs, and regional public buildings. The town has no industrial and business park with typically flat roofs.

The study is divided into the following sections:

Characteristics of Photovoltaic Systems

- State of the art of PV module technology
- Hazardous materials in PV modules

Estimation of PV Electricity Generation in Eggenburg by 2030

- State of expansion of solar roof–top plants in Eggenburg by 2015
- Solar radiation data for Austria and Eggenburg
- Monitoring data of roof–top plants in the region of the Waldviertel
- System model and expected electrical energy generation in Eggenburg by 2030
- Can the PV climate target be achieved?

End–of–Life Management of PV Modules

- Material fractions of silicon based and thin film PV modules
- End–of–life treatment of PV modules
- Future environmental pressure of end–of–life modules

The investigations process and combine literature data from scientific publications, governmental reports, and monitoring data of different stakeholders. Based on that, a realistic PV expansion scenario for Eggenburg is calculated by using Google mapping.

6.3. Characteristics of Photovoltaic Systems

6.3.1. State of the Art of PV Modules

PV module technologies can be divided in two main categories, silicon–based and thin film types. The statistical distribution of installed technologies is as follows (Fraunhofer ISE, 2016):

- Silicon–based technologies: 93 %
  – mono–crystalline silicon (mono–Si) 25 %
  – poly–crystalline silicon (poly–Si) 68 %
- Thin film technologies: 7 %
  – cadmium telluride (CdTe) 4.5 %
  – CuInGa diselenide (CIGS) 1.9 %
  – amorphous silicon (a:Si) 0.6 %

The state of the market for PV modules is determined by actual data sheets of the leading manufacturers. Reference data are shown in the following table.

Table 6.1. Best practice nominal power, efficiency, and technical data of different PV module technologies as on January 2017 (PV Datasheets, 2017).

Type	Company	Area per module	Nominal power	Nominal power/m^2	Module efficiency	Mass per module	Manufacturer comments
Mono–Si	SunPower X–Series	1.63 m^2	360 W	221 W/m^2	22.2 %	18.6 kg	Pb–free
Mono–Si	SunPower E–Series	1.63 m^2	327 W	201 W/m^2	20.4 %	18.6 kg	Pb–free
Mono–Si	Yingli Solar Panda Series 2	1.62 m^2	300 W	185 W/m^2	18.5 %	18.5 kg	
Poly–Si	Yingli Solar YGE Series	1.62 m^2 1.94 m^2	275 W 325 W	170 W/m^2 168 W/m^2	16.9 % 16.7 %	18.5 kg	
CdTe	First Solar Series 4	0.72 m^2	118 W	164 W/m^2	16.3 %	12.0 kg	
CIGS	Solar Frontier Avancis Power Max	1.05 m^2	140 W	133 W/m^2	13.3 %	16.0 kg	RoHS *) compliant

*) European RoHS–Directive (restriction of hazardous substances) for electric and electronic products

Using the data of table 6.1., the required average module area for an installed nominal power of 5 kW_p roof–top plant can be calculated as follows:

- Maximum nominal power of all technologies: 23 m^2
- Average mono–crystalline modules: 25 m^2
- Average poly–crystalline modules: 30 m^2
- CdTe modules: 30 m^2
- CIGS modules: 38 m^2

As calculation basis for the required roof area 10 % of the module area is added including installation distances.

6.3.2. Hazardous Materials in PV Modules

The European RoHS–Directive (restriction of hazardous substances) regulates the circulation of certain hazardous substances, that is lead, cadmium, mercury, hexavalent chromium, brominated flame retardants, and phthalates in electrical and electronic products. But the Directive explicitely does not apply to "*photovoltaic panels (...) installed by professionals (...) to produce energy from solar light*" (RoHS 2, 2011).

Silicon based PV modules contain up to 30 g toxic lead per module, which is part of the solder and the metallisation (Diermann, 2011). CdTe modules contain 43.4 g CdTe and 3.52 g CdSe per m^2 (Jungbluth et al., 2009). Another publication indicates, that the content of CdTe amounts to 18 g/m^2 with a mass ratio of 1:1 for cadmium and telluride (Bayerisches Landesamt, 2011).

During regular operation it might be no risk for the user and the environment. In case of fire or improper end–of–life treatment, including non-removal for decades, these substances can be uncontrolled released and spread into the environment.

6.4. Estimation of PV Energy Generation in Eggenburg by 2030

6.4.1. State of Expansion of Solar Roof–Top Plants in Eggenburg by 2015

The actual state of installed roof–top solar systems in Eggenburg was determined by google mapping. Figure 6.2. shows the distribution as of the satellite image date August 2015 (Map data ©2017 Google).

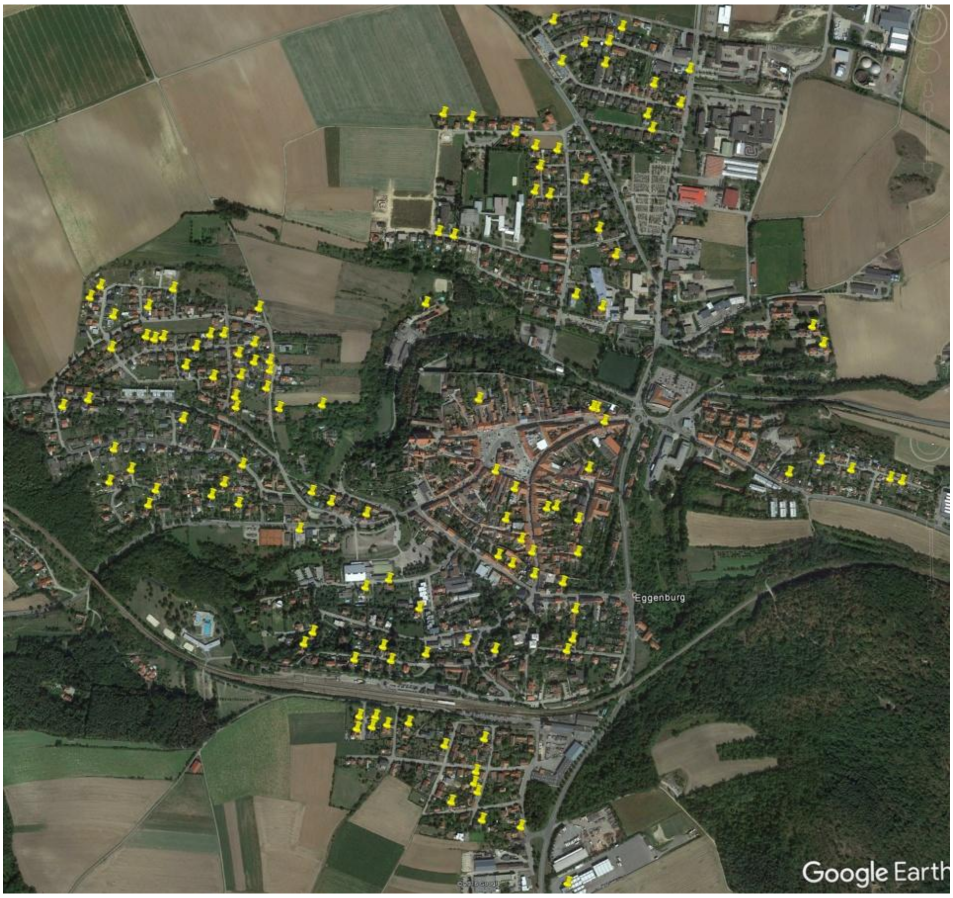


Figure 6.2. Installed solar systems in Eggenburg as of 2015.

It is noticeable that apartment buildings with large roof areas located in the eastern part of the city and large areas of the old inner city have no installed solar systems, whereby the city center is listed as historical monument. The main part of PV plants are located on single familiy houses all over the city quaters. The solar systems and their number of modules were counted: 124 solar systems are installed, whereby 56 plants are assumed to be photovoltaic plants due to the large number of modules. Two of them are ground based PV plants in house gardens. The top view gives no indication, whether it is a photovoltaic or a thermal solar plant. It is assumed that a system with < 10 modules is most likely a thermal system, however, this roof is "occupied". Table 6.2. shows the distribution of the determined PV plants between private and functional buildings.

Table 6.2. Distribution of the PV plants by dimension and building type.

Number of modules	Number of PV plants	Type of building
10 – 24	41	Single/two family houses
25 – 75	13	farm houses, public, business

6.4.2. Solar Radiation Data for Austria and Eggenburg

The project Solargis was funded by the European Commission in the frame of the FP3 research programme to determine the best sites for renewable energy systems on a regional level which is based on the use of a Geographical Information System (Cordis, 2017). Solargis provides a solar resource database for almost any location in the world. Figure 6.3. shows the distribution of the average annual sum of the global horizontal irradiance (GHI) for Austria.

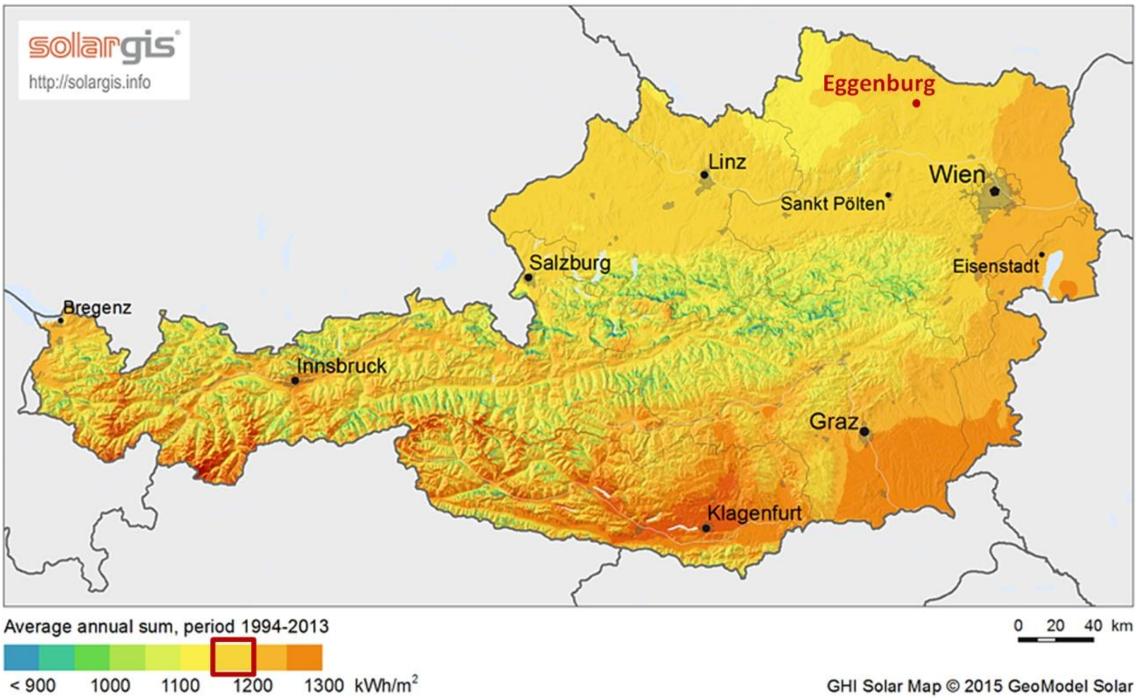


Figure 6.3. GHI Solar Map of Austria (Solargis, 2015).

The average annual GHI sum for Eggenburg is in the range of 1150–1200 kWh/m^2. In the following section, the average electrical energy production of existing PV plants of the same region is examined.

6.4.3. Monitoring Data of Roof–Top Plants in the Region of the Waldviertel

There are several initiatives providing monitoring data of the actual electrical energy production from photovoltaic systems. Solar–Log is operated by a private company that collects PV data including location, plant size, module technology, orientation, and roof pitch of more than 1000 PV plants in Austria (Solar–Log, 2016).

The photovoltaic company Sonnenstrom is located in the region of the Waldviertel and provides regional customer PV data including technical nominal data and yearly electrical energy production (Sonnenstrom, 2016). Thereof, the provided data of 42 PV systems, which are located in the same GHI region as Eggenburg (see figure 6.3.), were chosen to calculate the average electrical energy production dependent on the roof orientation and the roof pitch. Figure 6.4. shows the results of the samples in kWh per watt peak (W_p) and year.

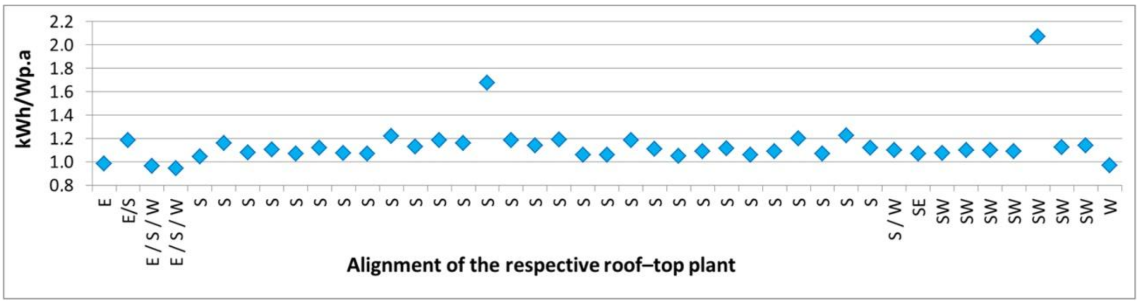


Figure 6.4. Annual electrical energy production of 42 roof–top plants, dependent on the orientation of the roof: E (east), S (south), W (west) or mixed, repectively.

The average specific electrical energy production for *south* oriented PV modules, excluding values with more than 20 % deviation, results to 1.119 kWh/W_p per year. *Southwest* oriented modules produce 1.25 % less electricity.

Figure 6.5. shows the results of the electrical energy production of the same samples, but sorted by the roof pitch in degree.

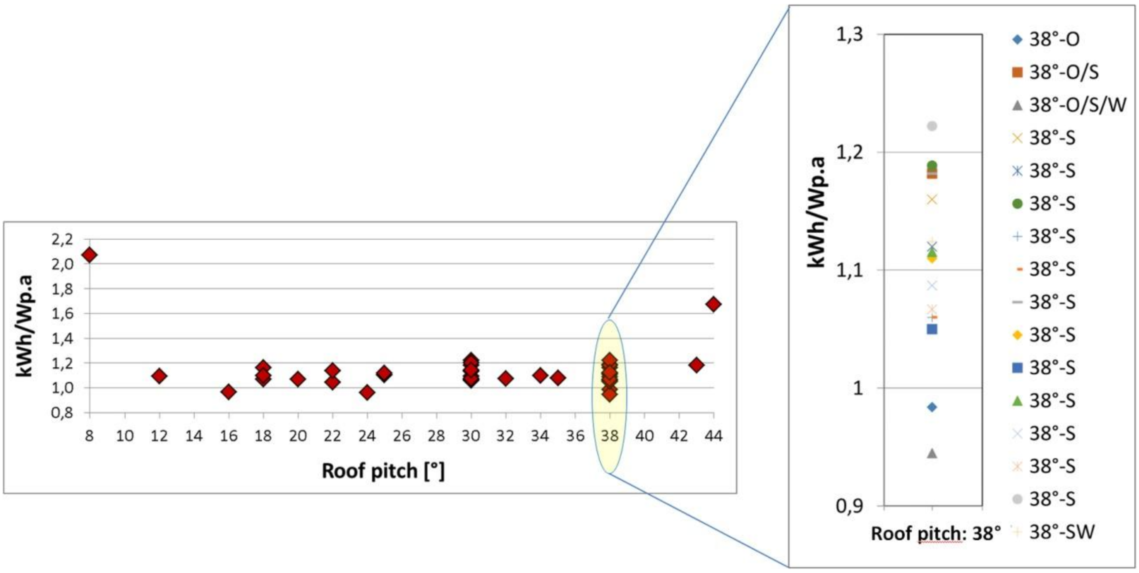


Figure 6.5. Annual electrical energy production of the same 42 roof–top plants as figure 6.6., but dependent on the roof pitch.

The most frequent constructed roof pitch is 38°, followed by 30°. The right part of the figure shows the enlarged results for 38° extended by the orientation of the respective PV system. The results show that there is no clear correlation between roof pitch and electrical energy generation. It is to be expected that the differences of the annual electrical energy production are more likely dependent on the manufacturing technology than on the roof pitch or the deviation from the ideal orientation of the solar radiation.

6.4.4. System Model and Expected PV Energy Generation in Eggenburg by 2030

The determined data so far constitute the basis for a system model to calculate the expansion, the material turnover, and the PV energy generation until 2030. The maximum nominal power per PV plant is assumed to be 5 kW_p according to the funding limit. The realistic maximum expansion potential is assumed to be four times as much roof–top PV systems as in the year 2015, that is about 500 plants. This value is based on a visual evaluation of the situation in Google Earth and the assumption, that not more than about 2/3 of the real estate properties will be made available for PV plants due to static and economic reasons, roof shape and orientation, solar thermal usage, etc. For a generally valid statement, the calculation base is one roof, which is equipped with modules with a statistical distribution of technologies. Materials and area usage for PV plants are calculated in the following.

Assumed boundary conditions: (see section 6.3.1.)

Table 6.3. Share of PV module technologies and their respective properties.

Share of module technology	**Nominal power**	**Mass per m^2**	**Hazardous substances per m^2**
25 % Si monocrystalline PV modules:	200 W_p/m^2	11.4 kg/m^2	18 g $lead/m^2$
68 % Si polycrystalline PV modules:	170 W_p/m^2	11.4 kg/m^2	18 g $lead/m^2$
7 % thin film PV modules (CdTe):	165 W_p/m^2	16.7 kg/m^2	27 g $CdTe,CdS/m^2$

Module area A per roof top plant (5 kWp), statistical distribution:

$A = P_{N,total} \cdot \Sigma\, p_i / P_{N,i}$ → $A = 28\ m^2$

Total mass M per roof top plant: (only modules)

$M = P_{N,total} \cdot \Sigma\, p_i / (P_{N,i} \cdot m_{s,i})$ → $M = 335$ kg

Hazardous substances per roof top plant and for Eggenburg in 2030, average:

$M_{Pb} = P_{N,total} \cdot \Sigma\, p_i / (P_{N,i} \cdot m_{s,Pb,i})$ → $M_{Pb} = 470$ g → $M_{Pb,Egg} = 235$ kg

$M_{Cd} = P_{N,total} \cdot \Sigma\, p_i / (P_{N,i} \cdot m_{s,Cd,i})$ → $M_{Cd,avg} = 57$ g → $M_{Cd,Egg} = 29$ kg

Roof with solely CdTe modules: → CdTe = 820 g

$P_{N,total}$...installed nominal power per roof
p_i...share of module technology
$P_{N,i}$... nominal power per module / m^2
$m_{s,i}$...specific mass / m^2

System model for PV Energy generation in Eggenburg

The annual electrical energy production of one roof top plant $E_{roof,yr}$ is calculated including the average electrical energy production 1.119 kWh/W_p (see figure 6.4.) and 5 kW maximum installed nominal power:

$E_{roof,yr} = 5.6$ MWh

The system model is divided in two timeframes, firstly, from 2016 to 2030 and secondly, as of 2030. The first scenario is defined as linear model with the starting point of 125 PV plants and an annual expansion of 25 plants until 2030. Figure 6.6. shows the system model with the input flows of the local solar irradiance and the PV materials, and the output flows of the produced electrical energy and waste.

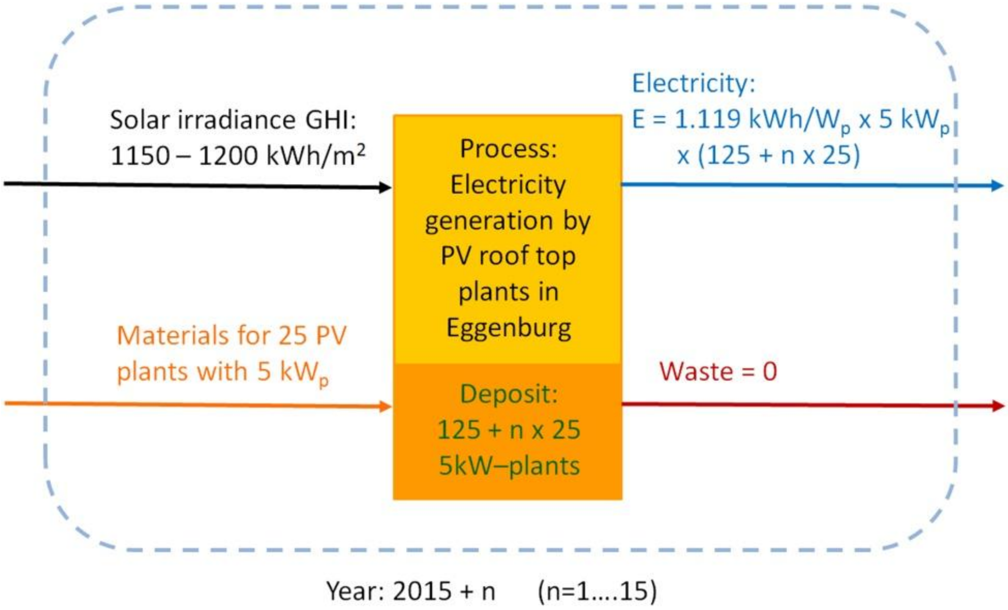


Figure 6.6. System model for the electrical energy generation scenario in Eggenburg between 2016 and the full expansion in 2030.

Until 2030 all PV systems are in operation and no waste is produced. The lifetime of a roof–top plant is assumed to be 20 years. As of 2030 the first generation PV plants reach their end–of–life and the modules become waste. From that year onwards the deposit of PV systems remains constant and it is assumend that 25 PV plant systems are rebuilt each year in accordance with the annual growth rate before. Figure 6.7. shows the system model as of 2030 with a stable electrical energy production.

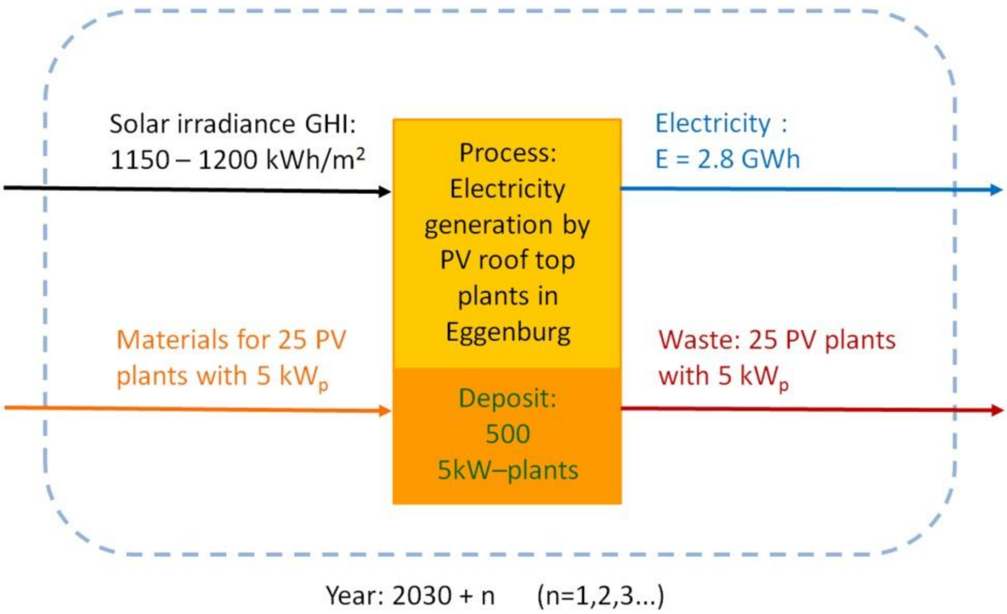


Figure 6.7. System model for the electrical energy generation scenario in Eggenburg as of 2030.

Considering the results of the mass calculations at the beginning of this section, as of 2030, the end–of–life modules produce an annual total waste of 8.4 tonnes, including 1.4 kg cadmium compound and 12 kg lead.

Figure 6.8. shows the graphical model for the electrical energy generation and the material turnover of all PV systems in Eggenburg as of 2016. The solid lines represent the calculations for a funding limit of 5 kW_p per PV system. The area calculations showed that in most cases the required roof area of 28 m^2 could be extended. Furthermore, it can be expected that the module efficiencies will increase and therefore, the required specific roof aera will even more decrease and the energy output increased, respectively.

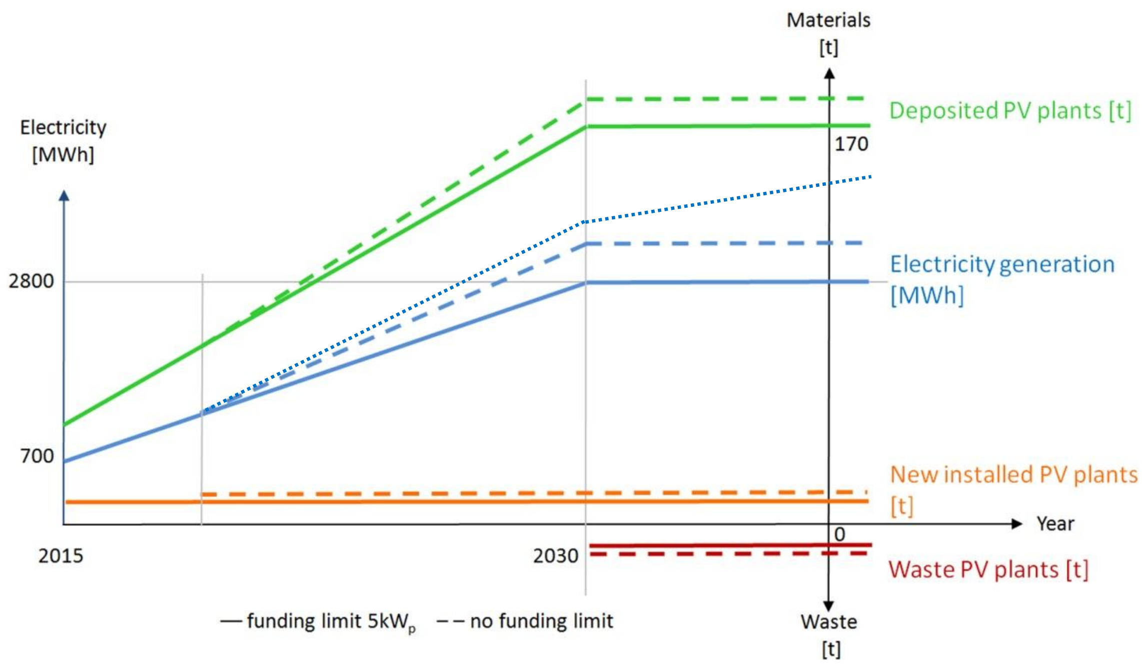


Figure 6.8. Graphical model for the electrical energy generation and material turnover scenario in Eggenburg as of 2015.

The dashed lines represent a possible better expansion situation, if there is no funding limit. In case of unlimited funding of the installed nominal power, steady efficiency increase of the modules, and full exploitation of the roof areas, the curve of the electrical energy generation (blue lines) will show a gradient straight line (blue dotted line) instead of the horizontal line.

The assumed initial condition of 125 PV plants in 2015 is not exact due to the number of thermal solar systems. This error would lead to a shift of the year of full PV expansion.

6.4.5. Can the Climate Target be Achieved?

PV energy production reduces greenhouse gas emissions. The amount depends on the regional electricity mix. In Germany, PV energy avoids 0.5 kg CO_2 per kWh (Fraunhofer, 2017).

The previous calculations showed that a realistic maximum achievable electrical energy production in Eggenburg based on roof–top PV plants is 2.8 GWh per year. As described in the introduction, the climate target for Austria is 15 % PV share of the electrical energy production in 2030. The following calculation investigates, if this target can be achieved.

Based on 2015, Austria consumed 70 TWh gross electrical energy with a total population of 8.6 Mio inhabitants, that is about 8,100 kWh per capita and year (Statistik Austria, 2016). It is estimated, that in 2030 the electrical energy demand is 82 TWh including transmission losses and consumption in the plant (Veigl, 2015). Including an increase of the Austrian population of 10 %, in 2030 the annual electrical energy demand per capita ist about 8,700 kWh. Based on these values, in 2030, Eggenburg with an assumed stable population of 3500 inhabitants shall produce 4.6 GWh PV electrical energy per year.

The PV expansion scenario shows, that in Eggenburg, only 60 % of the target, that is 9 % instead of 15 % of the electrical energy production, can be achieved in 2030. There is a need to amend the funding strategies for PV plants, first of all, the funding limits should be repealed to exploite the full roof area. New strategies of incentives and financing should be developed to exploite all flat roofs such as storage buildings, car park roofs, and other commercial and public facilities. The role of buildings under monumental protection should be rethought particularly concerning roof parts which are not directly visible in a historical ensemble. In case of larger cities all flat roofs of industrial parks should be exploited. However, the life time of commercial buildings could be a sensible factor compared to the life time of PV plants such as individual supermarkets, which are more often modernised at an early age. Ground based PV plants are problematic due to progressive land sealing and should be avoided, even if they would yield a good energy return.

6.5. End–of–Life Management of PV Modules

The first European expanding upturn for PV plants was in 2010 (see chapter 5). As of 2030, that is 20 years later, the first generation of large scale PV plants begins to reach its end–of–life. There are substantial efforts in the European Union to develop sustainable and economic recycling technologies.

It must be understood that there is a difference between the recycling of manufacturing waste and company owned end–of–life modules, repectively, and modules with unknown material composition. The situation is similar to all waste electric and electronic products, where large material fractions, such as glass, metals, and plastics, can be separated and (down)cycled. But the energy and precious material intensive core components are difficult and in many cases impossible to separate and recycle.

6.5.1. Material Fractions of Silicon Based and Thin Film PV Modules

The material composition of Si–based and thin film PV modules are shown in the tables 6.4. and 6.5. as they are used in the ecoinvent database for life cycle assessment (Frischknecht et al., 2015; Ecoinvent, 2017). The largest material shares of silicon based PV modules are glass and aluminium with 85 % and thin film modules have a share of 97 % glass (highlighted in blue). The EU e-waste Directive (WEEE, 2012) requires a recovery rate of 85 % and a recycling rate of 80 %. The remaining share of 15 % can be disposed in landfilles that means that this part is not anymore included in the circular economy. A comparison of these values with the mass values in the tables shows that the valuable and hazardous materials of PV modules are not required to be covered.

Table 6.4. Material composition of silicon based PV modules.

Material composition	**Mass per m^2 / Area**
Photovoltaic cell	0.935 m^2
Silicon product	0.122 kg
Aluminium alloy, $AlMg_3$	2.13 kg
Tin	0.013 kg
Lead	0.0007 kg
HDPE	0.0024 kg
Solar glass, low-iron	8.81 kg
Copper	0.1 kg
Glass fibre reinforced plastic, PA	0.3 kg
EVA foil	0.88 kg
PVF film	0.11 kg
PET granulate	0.35 kg
Total	**12.8 kg**

Table 6.5. Material composition of cadmium telluride PV modules.

Material composition	**Mass per m^2**
Silicon product	0.003 kg
Solar glass, low-iron	8.34 kg
Flat glass, uncoated	8.16 kg
Copper	0.011 kg
Glass fibre reinforced plastic, PA	0.11 kg
EVA foil	0.48 kg
CdTe, semiconductor grade	0.0233 kg
CdS, semiconductor grade	0.00352 kg
Total	**17.1 kg**

6.5.2. End–of–Life Treatment of PV Modules

PV CYCLE is a network of photovoltaic producers to install a pan–European take–back and recycling programme for PV modules (PV CYCLE, 2016). The recycling procedure of ***silicon based modules*** includes following steps (PV CYCLE, 2017):

- Removal of the aluminium frame and connectors
- Shredding
- Treatment in a flat glass recycling plant with following outputs:
  – Ferrous metals
  – Non–ferrous metals
  – Glass
  – Silicon breakage
  – Plastics
- Separation by optical sensors or chemical baths
- Glass is mixed with standards glass breakage and partly recycled for
  – Glass fibre products
  – Insulating products
  – Glass packaging

Cadmium telluride thin film modules are treated in a different way described by Martens and Goldmann (Martens, 2016):

- Drying process → Destruction of the laminate compound
- Semiconductor material is exposed to an oxidizing leach → the solution contains semiconductor metals, solid glass residues, and plastic residues
- Filtration: the dissolved semiconductor is precipitated by NaOH (sodium hydroxide) and filtered → filter cake including 96 % of the valuable semiconductor materials
- Glass and plastics recycling: solution residues are sorted in glass and laminate by means of a vibrating table → glass for recycling glass fibre

The upper process descriptions are general procedures which do not address specific problems and particular challenges during the waste treatment and recycling process. The e–waste directive includes a recovery target for PV modules of 80 % (WEEE, 2012). A possible higher recycling rate of e.g. more than 90 % or 95 % addresses all parts but the PV cell itself, as can be seen in tables 6.4. and 6.5. Valuable materials such as the silicon layer and copper as well as hazardous materials such as lead, cadmium telluride and cadmium sulfide do not necessarily have to be recovered to reach the required recycling rate, because their mass share is below 2 %. Therefore, there is no information, if the heavy metals which are dispersed on the surface, are recycled, incinerated or landfilled. As an example, the economically delamination and the residue–free separation of the adhesive layer from the PV cells is still an unsolved problem.

The project CABRISS, for example, which was funded in the frame of the EU Horizon 2020 programme, had the ambitious target to offer cradle–to–cradle solutions for PV modules on three levels: firstly, the economically high value recovery of secondary materials from broken cells and silicon kerf losses; secondly, innovative approaches for cost–efficient high–quality reuse of the recovered materials, such as silicon powder, conductive inks, and thirdly, enabling a sustainable cost–efficient production of new PV cell from recovered materials. The project recognised the problem of large transport distances from the widely spread PV plant locations to the recycling facilities and from there to the glass industry. Therefore it is recommended to install PV module recycling facilities located near the flat glass production industry to decrease end–of–life transport efforts and pollution, respectively. (CABRISS, 2017; Hoffmann et al., 2017)

6.5.3. Future Environmental Pressure of End–of–Life Modules

There are logistically large differences between roof–top PV plants and free–field plants. PV modules on roofs are included in already built–up areas and do not require additional land or other infrasturcture. Their end–of–life treatment can be included in construction waste disposal and fall under the concept of "urban mining". In general, roof tiles have a longer lifetime than PV systems. It should be considered that due to ownership and lack of financial resources it must be expected that an unpredictable number of PV systems will not be exchanged at the end of their lifetime. Electrical waste including valuable and toxic substances will remain of the roof and the roof will be "occupied" reducing the local energy generation at the same time. Other financial problems can occur, if roof tiles get damaged under the PV system, the PV system is newly mounted on an old roof, or the life time of the entire building is less than the PV plant. These circumstances should be evaluated and considered in the planning process of strategic regional roof–top PV expansion, the related expected electrical energy generation, and waste management.

6.6. Conclusion

The study shows that in 2030, the Austrian climate targets concerning electrical energy generation by photovoltaic plants may not be achieved exclusively with small and medium size roof–top systems which is shown by a representative model of a small town under the current technical and political funding framework. The forcasted electrical energy generation is 40 % below the target value. Policy for photovoltaic expansion should firstly, evaluate the reasons why commercial buildings in Austria are not equipped with PV plants, secondly, delevop strategies to change this situation, and thirdly, after successful exploitation of roofs the expansion of free–field PV plants should be newly evaluated and assessed.

The chosen system model provides for a linear increase of PV plant expansion. This recommended market development considers firstly, that there is enough free roof area for technologically new developed PV modules in the next decades and secondly, the future annual waste streams will be stable.

The continuous increase of PV plants leads to mass flows, which will have considerable environmental effects in the future in terms of end–of–life management on local level. Although roof–top PV systems logistically fall under construction waste management and urban mining, a new regional waste disposal logistics and sustainable waste treatment technology will be necessary from the time of the first module failures. As of 2030, in case of the calculation model of Eggenburg with 3500 inhabitants, 170 tonnes PV modules will be installed and it is expected that an annual waste material flow of 8.4 tonnes, including 1.4 kg cadmium compound and 12 kg lead according to the average share of module technologies, is generated. As stated in Chapter 5, this values are estimated on basis of literature data and there is no clear information about the actual distribution of heavy metals due to the large number of different manufacturers.

There is need of accurate legal and funding strategies to significantly increase the share on roof–top plants. The building codes should also be checked to faciliate the PV expansion on buildings. There is further need of research on future PV module material flows, in particular toxic substances, and local logistics for end–of–life roof–top PV systems.

6.7. References

Bayerisches Landesamt für Umwelt (2011). Berechnung von Immissionen beim Brand einer Photovoltaik-Anlage aus Cadmiumtellurid-Modulen. Augsburg, 2011 [German].

BMVIT (2016). Technologie–Roadmap für Photovoltaik in Österreich. Nachhaltig wirtschaften, Berichte aus Energie- und Umweltforschung 15/2016; Bundesministerium für Verkehr, Innovation und Technologie (Austria) [German].

BMWFJ (2010) Federal Ministry of Economy, Family and Youth Austria. National Renewable Energy Action Plan 2010 for Austria (NREAP-AT) under Directive 2009/28/EC of the European Parliament and of the Council.

CABRISS (2017). Implementation of a CirculAr economy Based on Recycled, reused and recovered Indium, Silicon and Silver materials for photovoltaic and other applications (EU Project Horizon2020) https://www.spire2030.eu/cabriss [30.10.2017].

Cordis (2017). European Commission. Community Research and Development Information Service. Project Solargis: http://cordis.europa.eu/project/rcn/26977_en.html [29.10.2017]

Diermann, Ralph (2011). Blei bleibt beliebt. Photovoltaik – Solartechnik für Installateure, Planer, Architekten. Ausgabe 04-2011 [German].

Ecoinvent (2017). http://www.ecoinvent.org/ [31.10.2017].

EU Directive (2009). Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC.

EU Directive (2016). Proposal for a Directive of the European Parliament and of the Council on the promotion of the use of energy from renewable sources (recast). COM(2016) 767 final; 2016/0382 (COD).

Fraunhofer ISE (2016). Photovoltaics Report, updated: 17 November 2016. https://www.ise. fraunhofer.de / de / downloads / pdf-files / aktuelles / photovoltaics-report-in-englischer-sprache.pdf [7.1.2017].

Fraunhofer ISE (2017): Photovoltaics Report, updated: 12 July 2017. https://www.ise.fraunhofer.de/content/dam/ise/de/documents/publications/studies/Photovoltaics-Report.pdf [19.9.2017].

Frischknecht, R.; Itten, R.; Sinha, P.; De Wild-Scholten, M; Zhang, J.; Fthenakis, V.; Kim, H. C.; Raugei, M.; Stucki, M., (2015), Life Cycle Inventories and Life Cycle Assessment of Photovoltaic Systems, International Energy Agency (IEA) PVPS Task 12, Report T12-04:2015.

Hoffmann, Marie-Charlotte; Suitner, Harald; Thomas, Robert (2017). H2020 CABRISS Public Business Plan. CABRISS – Grant Agreement 641972, European Union. www.spire2030.eu/cabriss [04/2019].

Jungbluth, N.; Stucki, M.; Frischknecht, R. (2009). Photovoltaics. In Dones, R. (Ed.) et al., Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. ecoinvent report No. 6-XII, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, 2009 [German].

KLIEN (2017). Austrian Climate and Energy Fund. https://www.klimafonds.gv.at/home-en-US/ [20.9.2017].

Martens, H.; Goldmann, D.;(2016), Recyclingtechnik. Springer-Verlag [German].

Narodoslawsky M.; Krotscheck C.; (1995). The sustainable process index (SPI): Evaluating processes according to environmental compatibility; J. of Hazardous Materials, 1995, 41 (2+3), 383-397.

Ökostromgesetz (2012). Federal Act on Supporting Electricity Produced from Renewable Energy Sources (Green Electricity Act 2012), Austria. Upcoming amendment 2017.

PV CYCLE (2016). Annual Report 2016. http://www.pvcycle.org/wp-content/uploads/2017/08/2016-Annual-Report-PV-CYCLE-AISBL.pdf [30.10.2017].

PV CYCLE (2017). Recycling of silicon based PV modules: https://www.youtube.com/watch?v=81-MEpcA-Rc [20.9.2017].

PV Datasheets [14.9.2017]:
SunPower X-Series:
https://us.sunpower.com/sites/sunpower/files/media-library/data-sheets/ds-x22-series-360-residential-solar-panels.pdf
SunPower E-Series:
https://www.helvetic-energy.ch/wp-content/uploads/2015/06/9_Modul_Sun Power_E20_327_WHT_DE.pdf
Yingli Solar Panda Series:
http://www.yinglisolar.com/de/products/monocrystalline/
Yingli Solar YGE Series:
http://d9no22y7yqre8.cloudfront.net/assets/uploads/products/downloads/DS_YGE60Cell-29b_35mm_EU_EN_20160121_V04.pdf
First Solar Series 4:
http://www.firstsolar.com/en-EMEA/-/media/First-Solar/Technical-Documents/Series-4-Datasheets/ Series-4V2-Datasheet.ashx
Solar Frontier Avancis Power Max:
http://pvtrade.eu/img/sol/615-024.pdf

RoHS 2 (2011). Directive 2011/65/EU of the European Parliament and of the Council of 8 June 2011 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (recast) (actual version 2015).

Solar-Log (2016). Solare Datensysteme GmbH. http://home.solarlog-web.at/ [7.1.2017].

Solargis (2016). GHI Solar Map © 2016 Solargis. http://solargis.com [7.1.2017].

Sonnenstrom limited ZNL Waidhofen (2016). Realisierte Photovoltaikanlagen, Einige ausgewählte Projektbeispiele unserer Kunden [German]
http://www.waldviertler.at/solarzelle/content/referenzen.html [7.1.2017].

Statistik Austria (2017). Statistisches Jahrbuch 2017, Kapitel 22 Energie.

Statistik Austria (2018). Einwohnerzahl und Komponenten der Bevölkerungsentwicklung http://www.statistik.at/blickgem/pr1/g31105.pdf [03/2019].

United Nations (2015). Framework Convention on Climate Change. FCCC/CP/2015/L.9/Rev.1. Paris, France, 2015. http://unfccc.int/resource/docs/2015/cop21/eng/l09r01.pdf [14.9.2017].

Veigl, Andreas (2015). Energiezukunft Österreich. Szenario für 2030 und 2050; im Auftrag von Global 2000, Greenpeace und WWF. https://www.global2000.at/sites/global/files/Energiezukunft%20%C3%96sterreich_.pdf (German)[31.10.2017].

WEEE (2012). European Commission: Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE) (recast). 2012.

7. Environmental Considerations on Market Expansion, Implications on Users, and End–of–Life Treatment of LED Lighting

Abstract: *LED–lamps are a very inhomogeneous group of illuminants made of various materials with different physical and electrical properties in different packaging designs. Types and materials of LED technologies are presented, followed by health effects and issues on glare and colour temperature. Subsequently, the global lighting market and the energy consumption of lighting in Austria are investigated in detail. Furthermore, the end–of–life management of LED–lamps is critically discussed. The study addresses material and technical data uncertainties and material flows of valuable substances for LED–lamps. Further, market development, local energy balances, and future environmental pressures are investigated. Holistic considerations are made concerning the well–being of the end–user and the rebound effect associated to the consumer behaviour. LED–lamps have a wide dispersion of designs, and the current market high–end luminous efficacy is in the range of 90 lm/W to 115 lm/W at a lifetime of 15,000 h. LED–lamps have a high share of blue light in the mixed colour spectrum, particularly "cold white" lamps with a colour temperature > 3000 K. Serious concerns have occurred regarding outdoor light pollution, health effects of blue light exposure, and the problem of glaring. The analysis of the Austrian energy statistics shows, that within ten years a significant rebound effect of + 29 % in terms of electrical energy consumption in household lighting can be observed although energy efficient lamp technologies were introduced to the market at the same period. The investigation of the e–waste management shows that LED–lamps do not meet the EU recycling targets for lamps as they have to be treated in the category of small electrical equipment with a low recycling rate due to their material composition.*

7.1. Introduction

LED–lighting systems have entered the European market on a large scale since the sale of incandescent lamps was restricted by the European Union in 2005 (EU Directive, 2005). The global market share of LED–lamps and luminaires is expected to be in the range between 67 % and 80 % of the total lighting revenue in 2022 (Zissis & Bertoldi, 2018). The global economic market share of residential LED–lamps increased from 13 % in 2012 to estimated 49 % in 2016 (McKinsey & Company, 2014).

The long term white light stability of LED–lamps is still not technically mature. The major problems are thermally induced failures (Fulmek et al., 2013) and an undesired colour shift (Schweitzer et al., 2014) over the lifetime and other degradation mechanisms. These effects lead to a significantly shorter lifetime of the LED–lamp as anticipated for the first generations of LED–lamps.

The application fields of white light LED–lamps are general lighting which includes household lighting, commercial and street lighting, lighting for automotive application, and backlighting for displays and TV with different technical requirements, respectively (McKinsey & Company, 2014). In this study the focus is on general lighting.

Besides ongoing research efforts to solve the problems of cooling and colour shift of the removable or "retrofit" LED–bulbs with standard threads, a parallel design line of LED–lighting with different technical and design requirements has entered the market: a fixed connected unchangeable LED–luminaire system, so called "dedicated" LED–lamp, which enables an attractive variety of designs and provides better cooling opportunities for higher efficacy. However, it is probable that a large share of low cost LED–lamps and luminaires have a significant shorter lifetime as it would be theoretically possible, because the thermal optimisation of lamps with high efficacy requires sophisticated and cost intensive design and manufacturing processes.

This work aims to give an overview on technological, environmental, and economic aspects of LED–lighting. The study presents technologies and types of LED–lamps and investigates published health effects of LED–lighting. Further, the global market of lighting and local Austrian market of LED–lamps as well as the relations of local energy consumption of household lighting in Austria are investigated. Concerning e–waste management, considerations about end–of–life treatment of LED–lamps in the frame of EU regulations are discussed.

In the frame of the thesis, the study addresses material flows of valuable substances, market development and future environmental pressure of electric devices on local level. The questions of local energy balance, uncertainty of technology and energy predictions, and rebound effect are investigated and discussed. Further, holistic considerations concerning the well–being of the end–user are investigated.

7.2. Materials and Methods

The considered topics of technologies as well as environmental, energy, and market aspects and their interactions are investigated. All data and information is based on scientific literature, reports of consulting companies, studies for governmental institutions, European legal documents, Statistic Austria, data sheets, studies of lamp manufacturers, and patents. Own calculations based on these data to link different data were conducted whenever necessary.

The study is structured into following topics:

- Technology of white light LED–lamps and luminous efficacy
- Types of white light LED–lamp components
- Flows of rare earths and strategic metals into LED production
- Health effects on humans and animals
- Global market of lighting and EU market of LEDs with different luminous efficacies
- Austrian energy balance and energy consumption of residential lighting in the light of larger LED market shares
- Aspects of end–of–life management of LED–lamps

7.3. Technology of LED–Lamps

LED–dice are based on semiconductor technology. Semiconductors are solid state materials characterised by a small band gap (the energy difference between isolating and conducting states). With the help of so-called dopants, the activation energy and the charge carrier density can be controlled over a wide range. (Dupuis et al., 2008, Nakamura, 2015)

Light emitting diodes are made from two adjacent semiconductor layers. One layer has a surplus of electrons (n–layer) and the other layer has a shortage of electrons (p–layer). If a voltage is applied, electrons of the n–layer can diffuse to the p–layer, where they recombine with the holes. The related energy is released in form of visible light. The wavelength of the emitted light depends on the material composition of the semiconductor layers.

The LED–package fabrication includes three main steps, the wafer substrate production for the semiconductor layers, followed by the LED–die fabrication, and finally, the electrical bonding and assembly of the die, the colour converter, and the optics.

Today the highest efficiencies are achieved with blue–light and red–light emitting LEDs. To generate white light, either several LEDs with different emission wavelengths can be combined or a blue-light emitting LED can be coated with a colour converter, typically a phosphor compound in a transparent matrix. The colour temperature, that is the warm or cold white impression, can be tailored by the material composition of the colour converter. The manufacturing of LED–dice is subject to large quality variations that are expressed in the price, representing different efficiencies, colour temperatures, and lifetime.

A LED–lamp consists of (a) the LED–array, (b) the electronic ballast for driving the LED–array, (c) the heat sink for heat dissipation of the LED–array and electronics, (d) an optical element, and (e) the housing. Materials and masses vary in a wide range, depending on the lamp design. Exemplarily, the total mass of retrofit LED–lamps varies between 83 g and 290 g (Navigant, 2012, Part 1). Dedicated LED–lamps, where the LED–array and the electronics are integrated in the luminaire such as ceiling lamps, have a mass between 520 g and 1.75 kg (Navigant, 2009; Principi & Fioretti, 2014; Tähkämö et al., 2013). Desk and floor lamps can weigh several kilograms. Streetlamps have a mass in the range of 12 kg and 15 kg (Abdul Hadi et al., 2013; Tähkämö & Halonen, 2015).

One of the key parameters of a LED–lamp is the *luminous efficacy* that is the luminous flux per consumed power expressed in lumen per watt (lm/W). There are different values of the luminous efficacy of LED–lamps. The initial value is the luminous efficacy of the LED–die. The assembly of the entire lamp consists of several components, which contribute to efficacy and energy losses in the operating state (Kölper et al., 2011):

LED–lamp efficacy = LED–chip efficacy × optical efficiency × driver efficiency × thermal efficiency

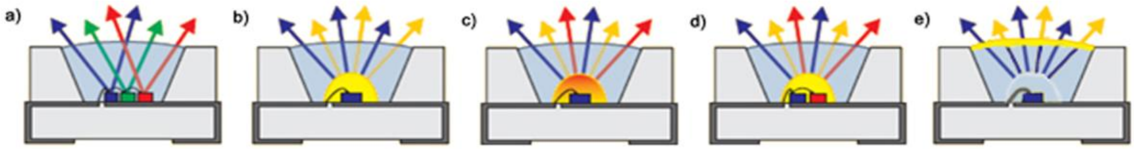
Strictly speaking, the LED–chip is again divided into the semiconductor, e.g. emitting blue light, and the colour converter. The driver efficiency includes the losses that occur when converting from 230 V AC to 2 V DC for the excitation of the semiconductor. All factors include thermal effects. In this case, the thermal efficiency depends on the operation temperature: as the junction temperature of the LED–chip increases, fewer photons are emitted and the LED becomes less efficient.

In summary, the LED–lamp / luminaire system efficacy is reduced by more than 50 % of the semiconductor. Retrofit LED–lamps have a lower efficacy due to their limited space and design options for thermal management. Dedicated LED–lamps have larger design opportunities for thermal management, because parts of the luminaire can be included in the cooling design. The more the technical features of drivers, e.g. dimming or colour adjustment, the higher are the energy losses.

7.4. Types of White LED–Lamps

LED–lamps can be classified into three application types: general lighting with 78 % economic market share, automotive lighting with 19 %, and backlighting such as monitors with 3 % market share in 2014 (Zissis & Bertoldi, 2018). The focus of this study is general lighting, whereby the technology principle is similar for all applications.

There are five main technologies to produce white light LEDs. Figure 7.1 shows typical examples of LED–technologies (Kölper et al., 2011). White light can be generated in following principle ways: combination of monochromatic LEDs with different colours (colour mixing, see figure 7.1.a) or activation of the so called *phosphor* or *colour conversion layer* by blue or UV emitted light (phosphor–conversion). Both technologies also can be combined (hybrid method). All principles base on optical colour addition. The colour converter layer can be directly applied to the LED–die (globe–top phosphor, see figure 7.1.b–d), or applied to the mixing chamber (remote phosphor, see figure 7.1.e).



a) Combination of a number of monochromatic LEDs
b) Excitation of a phosphor layer by a blue or UV LED
c) Additionally use of a number of different phosphor layers
d) Additionally use of monochromatic LEDs instead of a phosphor multilayer
e) Excitation of a remote–phosphor layer by a blue or UV LED

Figure 7.1. LED–component technologies. (Kölper et al., 2011; modified)

For general lighting, blue/UV light emitting semiconductors with a variety of sophisticated phosphor layer solutions became widespread. The world market leaders use semiconductors based on garnet wafers or orthosilicates (Tasch et al., 2010). The environmental impact of LED–lighting is dependent on various LED materials and technologies, but also on the thermal and colour quality of the lamp as well as on the life time of the mechanically stressed parts of the luminaire. "Warm white" lamps with a colour temperature of e.g. 2700 K have a lower luminous efficacy than "cold white" lamps up to 4000 K due to the lower share of blue light transmitting the colour converter and therefore require more energy to achieve the same lighting service.

7.5. Rare Earths and Strategic Metals in LEDs

The semiconductor and phosphor layer of the LED–component can be made of various materials using different technologies. Therefore, there are LED–lamps with different properties, efficacies, and reliabilities on the market which is not easily distinguishable for the average consumer. Table 7.1. shows rare earths and strategic materials used in different LED–components.

Table 7.1. Rare earths and strategic materials in LEDs.

Component part	Element	Rating
Semiconductor	Ga, In	Strategic metals
Color converter carrier crystal	Y, Gd, Lu	Rare earths
	Ge, Mg, Si	Strategic metals
Color converter doping element	Ce, Eu	Rare earths

The rare earths used in phosphor compounds must be 99.999 % pure, which means that LED–phosphor materials require more processing and purification steps than for other applications. (Wilburn, 2012)

The following tables give an overview of the market and global deposits of rare earth elements. The estimated world market demand of rare earth oxides used in LEDs for the year 2014 ± 15 % is shown in table 7.2. (Von Nauckhoff, 2011).

Table 7.2. Demand and Supply of rare earth oxides for 2014.

Rare earth oxide (REO)	Demand		Supply / Production	
	REO Tonnes	% of all REO	REO Tonnes	% of all REO
Cerium	65750	36.5 %	81750	40.2 %
Europium	840	0.5 %	850	0.4 %
Gadolinium	2300	1.3 %	3000	1.5 %
Yttrium	12100	6.7 %	11750	5.7 %
Ho-Tm-Yb-Lu	200	0.1 %	1300	0.5 %

Table 7.3. Deposit of rare earths in the earth´s crust (Gunn, 2014).

Element	Ce	Eu	Gd	Lu	Y
Crustal abundance [ppm]	64	0.88	3.8	0.32	22

There is no information on how much of the rare earths is contained in LEDs and how much it makes up of the total amount available. According to table 7.2., the availability of europium and yttrium could be a problem. The European Commission published in 2017 a list of critical raw materials for the EU. Concerning LED–lamps, all listed rare earths and strategic metals of table 7.1. are listed as critical raw materials (EC Communication, 2017). In general, it is reported that LED–lighting consumes very small amounts of rare earths compared to fluorescent lamps as they generate twenty times more lumens per gramme of rare earth (e.g. Guyonnet, 2018). This value should be verified as in many studies there is no clear indication of the rare earth quantity and furthermore, the lifetime of LED–lamps is assumed far too high. However, the extraction and recycling of rare earths from fluorescent lamps is possibly easier.

There is a need to perform a detailed material flow analysis of rare earths and strategic metals of the lamp sector as there are inaccurate and incomprehensive data circulated in the literature. There is no information about recycling technologies for rare earths of LED–lamps. The environmental pressure is less expected from the rare earth metal itself contained in the lamp than from the extraction of raw materials, as large amounts of acids contaminate the mining sites (Lohmann & Podbregar, 2012).

7.6. Health Effects on Humans and Animals

To evaluate the environmental compatibility of a product, the standardised method of life cycle assessment (ISO 14040, 2006; ISO 14044, 2006) is commonly used. LCA methods also may include assessment of human- and ecotoxicities, which, however, relate to metabolic effects. The following presentations show research results on the health impact of LED–lighting beyond metabolic impacts.

The life cycle assessment methods do not consider damage effects to the eyes and the circadian rhythm, due to their predefined boundary conditions. A study by the International Energy Agency states that LED–lamps have no worse health effects than other lamp types (IEA, 2014). In contrast, a study conducted by Jaadane et al. found retinal damage induced by commercial white and blue LED–lamps with different wave lengths on rats. The lamp distance to the animals was 25 cm and the exposure time was 6 hours to 72 hours (Jaadane et al., 2015). The investigated LED–lamp distance is comparable to desk lamps or bedside lamps.

An article published in Neuroscience compared in a very controlled manner the effects of different light sources. It was found that white, blue, and green LEDs provoke retinal damage while compact fluorescent lamps (CFL) and long fluorescent lamps (FL) do not. The authors highlight once more the danger of blue light and particularly of blue–LEDs (Krigel et al., 2016).

There is evidence that the increasing blue content of light can worsen the biological impact on humans and wildlife. The eye has nonvisual receptors for blue light which controls the diurnal cycles, that is the sleep and wake rhythm. The impact of LED–street lamps on the circadian rhythms is five times higher than of conventional street lamps. The negative impacts on bugs, bats, birds, and sea turtles are emphasised. (Hecht, 2016)

A study by Zissis investigated a standardised light exposure test and found that cold white LEDs with a luminous flux of 200 lm have a moderate eye damage risk by a maximum exposure time of 40 to 100 seconds. Warm white LEDs have no risk at 100 lm and 200 lm, and a low eye damage risk at a luminous flux of 500 lm at an exposure time of 100 seconds. (Zissis, 2016)

Villa et al. investigated pedestrian discomfort glare from urban LED–lighting. The findings confirmed previous studies, which recommend frosted or opalescent lamp covers to limit the maximum luminance and therefore the discomfort glare. (Villa et al, 2016)

The study by Dale et al. states that the city of San Diego has limited the maximum colour temperature of street lamps to 3500 K (Dale et al., 2011). The latest recommendation concerning the maximum colour temperature of outdoor LED–lamps is 3000 K (Hecht, 2016).

The above examples show that there is scientific evidence that lighting with LED–lamps can have a problematic health impact on humans and animals. This fact is neither recognized by streamlined LCA studies, nor are there any regulations on the colour spectrum and how to handle LED–lamps. There is a need for further research on health and environmental issues of LED–lamps for residential, public, and outdoor application. It can be observed that it is still common to newly install LED–lamps with higher colour temperature than 2700 K particularly outside private households.

7.7. Global Market of Lighting

7.7.1. Global Market of All Lamp Technologies and Sectors

The global lighting market is divided into three main sectors: general lighting (79 %), automotive lighting (20 %), and backlighting (1 %), with an estimated total producer price volume of 91 billion Euro in 2016. Figure 7.2. shows the annual producer price market share in 2016, depending on the application category and the share of LED–lamps and other lamp technologies in each category. The lamp technologies include incandescent lamps, halogen lamps, high–intensity discharge lamps (HID), long fluorescent lamps (LFL), compact fluorescent lamps (CFL), and light emitting diode lamps (LED). The market of lighting control systems is included with an average share of 5.6 % of the general lighting market and a growth rate anticipated at almost 20 % per year. (McKinsey, 2012)

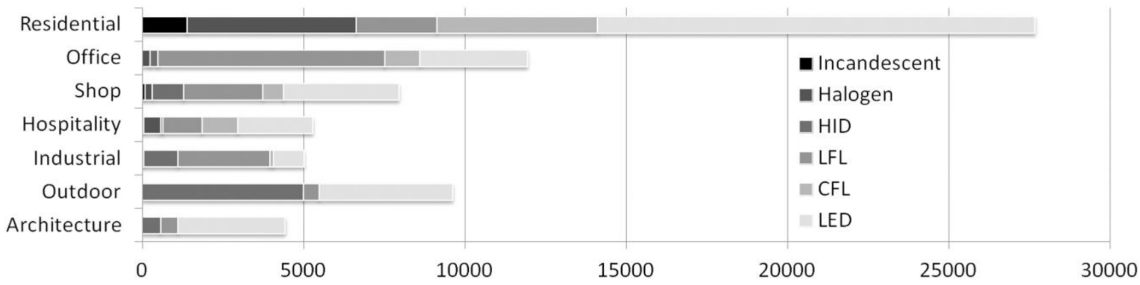


Figure 7.2. Economic general lighting market: annual new installations and replacements in 2016 [million EUR]. (Own work; Data source: McKinsey, 2012)

Figure 7.2. would possibly lead to the assumption that the residential sector has the highest number of total installed fixtures, but these values mainly seem to represent the higher replacement rate in this sector. This might be caused by the fact that the residential sector has the highest share of incandescent or CFL lamps with relatively short lifetime compared to the other sectors.

Figure 7.3. compares the global lighting market for different regions of the world. From a territorial point of view, Asia holds 42 % of the global lighting market share, followed by Europe and North America. Russia has a market share of 8 % of the European lighting market in 2016.

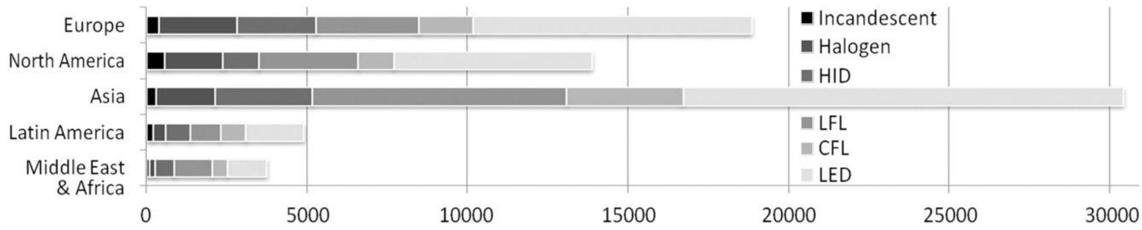


Figure 7.3. Global annual economic share for different application segments of the general lighting market for different regions of the world, 2016 [million EUR]. (Own work; Data source: McKinsey, 2012)

In total, the number of globally installed light sources for general lighting amounts to 32 billion units. The McKinsey lighting study further estimates that 4.3 billion of these were to be newly installed or replaced in 2016. 58 % are LED–light sources and thereof, 47 % are retrofit LED–lamps and 53 % are full or dedicated LED lamps. For comparison how statistics change between unit–% and Euro–%, figure 7.4. shows two diagrams with the same market situation in 2016.

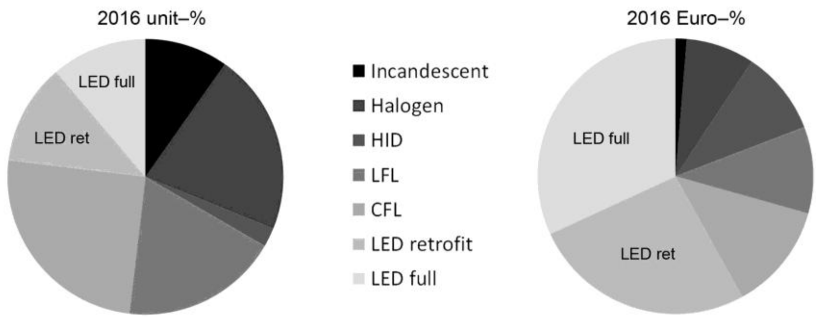
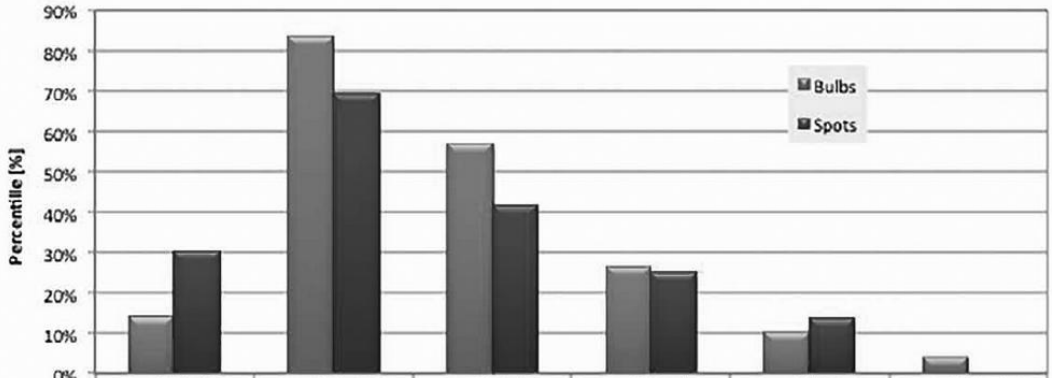


Figure 7.4. Global newly installed or replaced light sources by technology in 2016 per unit share (left) and per economic share (right). (Own work; Data source: McKinsey, 2012)

The different representations of the diagrams show that LED–lamps have a 23 % market share from the point of view of units and 58 % from the economic point of view.

7.7.2. European Market of LED–Lamps and Their Luminous Efficacy

A study conducted by Zissis measured the luminous efficacy of 370 samples of warm white high quality omnidirectional and spot LED–lamps for residential lighting from store shelves of the European Union. The study also measured the deviations from claimed values. Figure 7.5. shows the respective results from this study. (Zissis, 2016)



	η < 60 lm/W	η > 60 lm/W	η > 70 lm/W	η > 80 lm/W	η > 90 lm/W	η > 100 lm/W
Bulbs	15%	85%	57%	27%	10%	4%
Spots	31%	69%	42%	25%	14%	

Figure 7.5. Distribution of measured luminous efficacies (lm/W) of LED–lamps. Light bars: LED–bulbs, dark bars: LED–spots. (Zissis, 2016; modified)

The figure shows that 27 % of the retrofit LED–bulbs have a luminous efficacy > 80 lm/W and only 10 % > 90 lm/W in the period of the study.

Table 7.4. shows the results of an evaluation of commercial LED–lamps offered by mainstream and brand companies in 2015. The table is divided in street lamps, retrofit warm white lamps, and dedicated lamps for household application. The quoted properties are the luminous efficacy and the predicted lifetime of the respective lamp. The respective datasheets are added in blue brackets and referenced in Section 7.12.

The luminous efficacies of street lamps are in the range from 68 lm/W to 120 lm/W and the lifetime is specified up to 100,000 h. Warm white retrofit lamps have a specified luminous efficacy up to 90 lm/W and a lifetime up to 25,000 h. Dedicated LED–lamps tend to have a lower luminous efficacy with longer lifetime. The market review of 2015 in table 7.4. shows that the market share of retrofit lamps with a luminous efficacy of about 80 lm/W significantly increased within a few years compared to the previous findings of Zissis in figure 7.5.

Table 7.4. Luminous efficacy and declared lifetime of commercial LED–lamps in 2015.

Luminous efficacy of commercial LED–lamps [lm/W]							
Street lamps [lm/W]							
130-140[1)] 110[2)] (1)	120 (2)	116 (3)	105 (4)	100 (5)	90 (6)	68 (7)	87 (8)
80000 h	100000 h			50000 h		30000 h	
[1)] Bridgelux Led–chip [2)] Light yield							
Retrofit lamps, "warm white" [lm/W]							
[3)] 60 W	40 W	75 W	60 W	40 W	75 W	75 W	40 W
90 (9)	78 (10)	78 (11)	81 (12)	78 (13)	88 (14)	117 [4)] (15)	63 (16)
15000 h			25000 h	15000 h	20000 h	15000 h	15000 h
[3)] ≙ nominal power of incandescent lamps [4)] 4000 K "cold white"							
Dedicated LED–lamps, household [lm/W]							
desk lamp			built-in lamp		ceiling lamp		floor lamp
50 (17)	50 (18)	50 (19)	80 (20)		67-84 (21)		101[5)] (22)
40000 h	15000 h	20000 h	20000 h		50000 h		50000 h
[5)] 4000 K "cold white"							

A short market review in 2019 shows that the luminous efficacy of retrofit LED–lamps with a colour temperature of 2,700 K is in the range of 90 lm/W to 115 lm/W with a lifetime of 15,000 hours in the high quality sector. It is clearly recognisable that in recent years the declared lifetime of LED–lamps has been significantly reduced to the range of 15,000 h. This reduction has a large influence on LCA results, because the environmental impact of the LED manufacturing has a considerable share which is shown in Chapter 10.

7.8. Austrian Energy Balance and Energy Consumption of Residential Lighting

The following section investigates the ratio between the total final energy consumption in Austria and the electrical energy consumption of residential lighting. In particular, the changes of the electrical energy consumption of households in the course of the introduction of LED–lamps over the years are investigated.

The energy statistics in table 7.5. shows the final energy consumption of residential / household lighting and total lighting & IT (pre-defined category) in Austria from 1999 and 2014. The two sectors, private and public/commercial, use different lamp types regarding power, colour temperature, ambient conditions or lifetime. The respective data sheet is noted in blue brackets and referenced in Section 7.12.

Table 7.5. Final energy / electrical energy consumption and sectoral values between 1999 and 2014 in Austria, and calculation of the shares (as far as data were available).

Sector	1999	2003	2008	2012	2013	2014
Final Energy Consumption / TJ (Statistik Austria, 2015a)						
Total	934394	1060574	1109303	1099791	1119241 (23)	1073502 (24)
Households	268260	268153	262957	275815	278171 (23)	n/a
Share in total final energy	28.7 %	25.3 %	23.7 %	25.1 %	24.9 %	n/a
Lighting & IT (total)	28535	28324	32533	32333	32828 (23)	n/a
Share in total final energy	3.1 %	2.7 %	2.9 %	2.9 %	2.9 %	n/a
Final Electrical Energy Consumption / TJ						
Total		197927 (25)	217740 (25)	221560 (25)	223731 (25)	220573 (24)
Households (26)	49227	58523 59893 (27)	58028 60127 (27)	60696 (27)	60820 (25)	59993 (24)
Lighting (27)	n/a	5562	5605	7153	n/a	n/a
Share in el. household		9.5 %	9.7 %	11.8 %		
Share in el. total		2.8 %	2.6 %	3.2 %		
Share in household final energy		2.1 %	2.1 %	2.6 %		
Share in total final energy		0.5 %	0.5 %	0.65 %		

Private lighting consumes about 12 % of the household electrical energy demand, 2.6 % of the household final energy consumption, and < 1 % of the total energy consumption in 2012.

The electrical energy consumption of households increased from 2003 to 2012 in the following way:

- Total household final energy: + 1 %
- Total household lighting: + 29 %
- Lighting per household: + 16 %

The difference between 29 % and 16 % results from the fact that the population has grown by 4 % during that period, the number of households increased and the number of persons per household decreased from 2.42 to 2.27 (Bittermann et al., 2014).

The electrical energy consumption per capita depends on the number of persons per household. A single household consumes 325 kWh/a, a five-person household consumes 230 kWh per person and year for lighting in 2012. The study by Bittermann et al. provides a comprehensive and detailed analysis of electrical energy consumption for lighting and other household sectors dependent on various parameters like number, age, education, or profession of the household members and should be representative for central Europe (Bittermann et al., 2014). In comparison, an average U.S. household consumes 1700 kWh/a for lighting with about 70 lamps per household (U.S. Dep. of Energy, 2012), which is more than the double compared to Austria, which is estimated by own evaluations. Figure 7.6. shows a graphical overview of the Austrian energy shares with respect to household lighting.

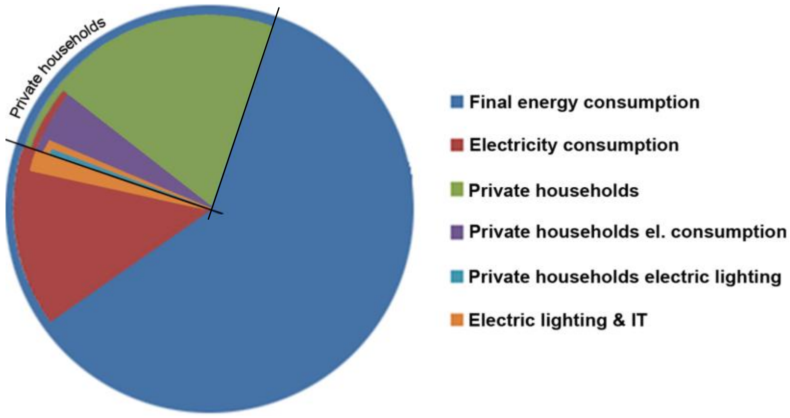


Figure 7.6. Schematic energy balance of Austria with the focus on private households.

The figure shows the annual final energy consumption of all sectors in Austria (blue circle) and thereof the share of the final energy consumption of the households (green circle segment). The further drawn sectors are the total electrical energy consumption (red) and electric lighting & IT (orange) for both households and all other sectors. The analysis shows that in private households the energy consumption of IT is higher than the lighting. Finally, the electrical energy consumption for household lighting is coloured in light blue. The coloured segments are prepared in that way that all overlapping data are visible.

Household lighting amounts to 0.65 % of the total energy consumption and about 3 % of the electrical energy consumption of Austria in 2012. There are no specific data of commercial and public lighting, but a total value of the combination "lighting & IT", which amounts to 3 % of the total final energy consumption and about 15 % of the total final electrical energy consumption, respectively.

The results of table 7.5. together with figure 7.6. show that the household electrical energy consumption for lighting is of minor significance compared to the other energy related sectors. The results further show, that in the same decade as energy saving lamps, that is compact fluorescent lamps followed by LED–lamps, were introduced to households, the residential lighting energy consumption has increased. That means, the energy saving potential has been overcompensated by the rebound effect whose detailed cause should be investigated more closely.

Besides the increase of the number of households in Austria, which might be one explanation for the rebound effect, the following considerations can explain the high value of the total increase of the energy consumption of lighting: LED–lamps were intensively promoted as exceptional low energy consumers that on the one hand led to behaviour not to turn off the light if not needed and on the other hand to increase the number of lighting points as in parallel a large number of new lighting designs entered the market. It can be further observed that in the course of building renovations the illumination intensity increases for example concerning elevators, corridors and staircases. The reason for that is unclear but it might correspond to new standards which are commonly adhered by industry. Further, sensor controlled lighting in facilities could lead to longer operation time of lamps particularly in areas where a certain degree of natural light source is available. In larger cities (e.g. in Vienna) it can be observed that street lighting based on long fluorescent lamps have a night lowering of the illumination in some areas of secondary streets, which worked well for decades. This is not the case for exchanged modern LED–lighting systems. There are no studies so far that have scientifically investigated these empirical observations about possible reasons for the rebound effect seen in the consumption data.

7.9. End–of–Life Management of LED–Lamps

The European WEEE–Directive (waste of electrical and electronic equipment) obliges the collection and proper treatment of all waste lamps (EU Directive, 2012). The category "Lamps" of Annex IV explicitly also includes LED–lamps. Since August 2018, the minimum treatment and recycling target for lamps requires that 80 % shall be recycled.

Gas discharge lamps and LED–lamps are very different in their technical structure. Gas discharge lamps, including long fluorescent lamps and compact fluorescent lamps, consist of a glass tube filled with phosphor, gas, a small amount of mercury, and in the case of CFL lamps an integrated electronic part. The recycling procedure is done by a broken glass washer or the "Kapp-Trenn" (trim and separate) process where in both cases glass, metals, phosphor and mercury are separated (Martens, 2011). Long fluorescent lamps consist 80 % to 90 % of glass.

Retrofit LED–lamps consist of the electronic part (17 %), the LED–chip (1 %), the heat sink made of aluminium (41 %), the bulb made of plastics and synthetic resin (38 %), and other non-ferrous metal (3 %) (Osram, 2011). There are also LED–bulbs made of glass on the market, however, their market share is not clear, as the material is not declared. There is no mercury in LED–lamps compared to fluorescent lamps but there can be other hazardous materials included such as lead and arsenic (Drachenberg et al., 2016), particularly outside the European market. There is still no successful recirculation technology available for precious components and materials of LED–lamps.

LED–lamps are structurally similar to small electronic equipment. According to personal information from the e-waste collection sector in Austria, LED–lamps are collected in the category of "Lamps" and further redistributed to the category "Small equipment" according to WEEE Annex III (EU Directive, 2012), at least in some areas. Although this decision is reasonable in terms of content, particularly also for dedicated LED–lamp systems, it contradicts the WEEE regulation, which explicitly excludes lamps from "Small equipment". The required recycling targets of small equipment are far lower than lamps, that is 75 % recovered and 55 % recycled. There is no information what further happens with the electronic part and particularly the LED–chip, especially since they are relatively small and therefore, not necessarily in the scope of the separation and recycling target. There is also no further information, if the components including rare earths and strategic metals are in any way collected and stored until there is a commercial recycling technology available. From the current point of view it seems that the valuable materials of waste LED–lamps are lost due to incineration or landfilling.

7.10. Conclusion

LED–lamps are a very inhomogeneous group of illuminants made of different semiconductor and phosphor materials with different properties, designs, lifetimes, luminous efficacies, and reliabilities. This circumstance makes it difficult to make general statements on the environmental performance of LED–lighting, particularly in comparison with other efficient lighting technologies. In scientific and public communication about the luminous efficacy and energy saving potential of LED–lamps, it must be questioned exactly which technology step, which specific product and application, and which color temperature these data refer to.

LED–lamps contain a number of different rare earths and strategic metals in very small doses in the semiconductor and colour converter of the LED–chip. Nevertheless, the total quantity is relevant due to the large number of devices on the market. There is no clear information about types and representative quantities and it is recommended to perform a comparative material flow analysis from cradle–to–grave of rare earths and strategic metals through the lighting sector, including also the rare earth mining sector.

White LED–lamps include a significant share of blue light in the mixed colour spectrum. A higher energy efficiency is combined with a higher share of blue light, controlled by the applied colour converter, at the expense of uncomfortable lighting that causes damaging light pollution in indoor as well as outdoor application. In the last years serious concerns have occurred regarding outdoor light pollution, health effects of blue light exposure, and the problem of glaring, which have been confirmed in several studies. These aspects are not in the scope of a streamlined life cycle assessment. An approach to solve the problems of glaring and blue light exposure could be a combination of blue and red emitting LEDs combined with a new colour converter technology and frosted or opalescent lamp covers. However, these measures commonly lead to a reduction of the luminous efficacy of the lamp. Nevertheless, it is recommended that the practice and standards regarding the applied colour temperature should be checked and limited to a maximum of 3000 K, for example in pedestrian priority zones such as corridors, staircases and paths, and possibly also for street lighting.

A global market review shows that in 2016, the estimated share of LED–lamps of newly installed and/or replaced light sources is 23 % on unit basis and 58 % on economic basis. A further European market review shows that the luminous efficacy of retrofit LED–lamps with a colour temperature of 2,700 K is advertised in the range of 90 lm/W to 115 lm/W in the high quality sector. The average luminous efficacy of LED–lamps however is still below 90 lm/W. It is recognisable that in the recent decade the declared lifetime of LED–lamps has been significantly reduced from the range of 50,000 h to currently 15,000 h.

The energy balance on local level in Austria shows that household lighting has a share of about 0.65 % of the total national energy consumption and 11.8 % of the household electrical energy demand. The lighting sector and particularly LED–lamp systems have low impact on the overall energy demand, but it is obviously interesting in the commercial sector, which is also shown in the market review. The energy statistics show further, that in the same decade as energy efficient lamps were introduced to the market the lighting energy consumption increased by 16 % per household. That means, the energy saving potential of CFL and LED–lamps has been overcompensated by the rebound effect. It needs to be studied more closely, by what exactly the rebound effect is caused, whether by advertising pressure for additional and brighter lamps and/or extensive longer operation time at presumed very low energy consumption. The question of the energy consumption of sensor controlled lighting should be also taken into consideration. In general, energy saving strategies urges to consider both the technical improvement of electric devices and potential changes of the consumer behavior and/or technical standards, such as brightness, which are driven by the industry.

LED–lamps are structurally similar to small electronic equipment. It can be assumed that the e–waste treatment of LED–lamps does not reach the recycling targets of the European WEEE regulations for lamps due to the material composition, which should be considered in a recast of the Directive. A possible full market penetration of LED–lamps instead of fluorescent and other efficient lamp types has a negative influence on the lamp waste management and increases the future environmental pressure of lighting as a whole. That is notably because it turned out that the lifetime of LED–lamps is not in the expected range but similar to long fluorescent lamps. Fluorescent lamps are still an energy efficient competitor to LED–lamps (see also Chapter 10) and the considerations about waste management could point to further parallel research in technology and more attractive design of long fluorescent luminaires. There is a need to evaluate the recycling potential and real recycling rates of rare earths and strategic metals of the lamp sector to reduce the high environmental pressure of mining of these materials.

7.11. Acknowledgement

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7.12. References

Abdul Hadi, S.; Al Kaabi, M.R.; Al Ali, M.O.; Arafat, H.A. (2013). Comparative Life Cycle Assessment (LCA) of streetlight technologies for minor roads in United Arab Emirates, Energy for Sustainable Development 17(5), Pages 438–450, 2013.

Bittermann, W.; Gollner, M.; Strasser, M. (2014). Modellierung des Stromverbrauchs in den privaten Haushalten Österreichs nach unterschiedlichen Verwendungszwecken; Statistik Austria, Direktion Raumwirtschaft, Energie, 2014. (Modeling of the electrical energy consumption in private households, German).

Dale, A. T.; Bilec, M. M.; Marriott, J.; Hartley, D.; Jurgens, C.; Zatcoff, E. (2011). Preliminary Comparative Life–Cycle Impacts of Streetlight Technology. Journal of Infrastructure Systems, 2011, 17(4), 193–199.

Drachenberg, Elizabeth; Mihailova, Darja; Nandamuri, Sai Siddhartha (2016). Raw Materials Use in LEDs. EU INTERREG: Lighting Metropolis.

Dupuis R. D.; Krames, M. R., (2008). History, development, and applications of high-brightness visible light emitting diodes, Journal of Lightwave Technology 26, 1154 – 1171.

EC Communication (2017). Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of Regions on the 2017 list of Critical Raw Materials for the EU. Brussels, 13.9.2017 COM(2017) 490 final.

EU Directive (2005). Directive 2005/32/EC: Ecodesign requirements for non-directional household lamps. 2005.

EU Directive (2012). Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE) (recast). 2012.

Fulmek, P.; Sommer, C.; Hartmann, P.; Pachler, P.; Hoschopf, H.; Langer, G.; Nicolics, J.; Wenzl, F. P. (2013). On the Thermal Load of the Color-Conversion Elements in Phosphor-Based White Light-Emitting Diodes; Advanced Optical Materials, Vol. 1, 10, pp. 753–762, 2013.

Gunn, G. (Ed.) (2014). Critical Metals Handbook, British Geological Survey, Wiley, 2014.

Guyonnet, Dominique; Lefebvre, Gaétan; Menad, Nourredine (2018). Rare earth elements and high-tech products. Prepared for CEC4EUROPE (Circular Economy Coalition for Europe), August 2018.

Hecht, Jeff (2016). The Early–Adopter Blues. Spectrum IEEE, October 2016.

IEA International Energy Agency (2014). Life Cycle Assessment of Solid State Lighting, Solid State Lighting Annex. Final Report, 2014.

ISO 14040 (2006). Environmental management – Life cycle assessment – Principles and framework.

ISO 14044 (2006). Environmental management – Life cycle assessment – Requirements and guidelines.

Jaadane, I.; Boulenguez, P.; Chahory, S.; Carre, S.; Savoldelli, M.; Jonet, L.; Behar–Cohen, F. (2015). Retinal damage induced by commercial light emitting diodes (LEDs). Free Radical Biology and Medicine 84(2015) 373–384.

Kölper, Christopher; Bergbauer, Werner; Strassburg, Martin; Linder, Norbert (2011). Die Licht(r)evolution. Weiße LEDs für die Allgemeinbeleuchtung. Phys. Unserer Zeit. Viley-VCH, 2/2011 (42), p. 92–98.

Krigel, A.; Berdugo, M.; Picard, E.; Levy-Boukris, R.; Jaadane, I.; Jonet, L.; Dernigoghossian, M.; Andrieu-Soler, C.; Torriglia, A.; Behar-Cohen, F. (2016). Light-Induced Retinal Damage Using Different Light Sources, Protocols and Rat Strains Reveals LED Phototoxicity. Neuroscience 339, 296-307, 2016.

Lohmann, Dieter; Podbregar, Nadja (2012). Im Fokus: Bodenschätze. Die Jagd nach Seltenen Erden und anderen Rohstoffen. Springer-Verlag Berlin Heidelberg 2012.

Martens, Hans (2011). Recyclingtechnik. Spektrum Akademischer Verlag Heidelberg.

McKinsey & Company (2012): Baumgartner, T.; Wunderlich, F.; Jaunich, A.; Sato, T.; Bundy, G.; Grießmann, N.; Kowalski, J.; Burghardt, S.; Hanebrink, J. (Eds.) (2012). Lighting the way: Perspectives on global lighting market. Second edition.

McKinsey & Company (2014). Lighting the way: Perspectives on the global lighting market, second edition, 2014.

Nakamura, S. (2015). Background story of the invention of efficient blue InGaN light emitting diodes (Nobel Lecture), Annalen der Physik 527, 335–349, 2015.

Navigant Consulting (2009). Life Cycle Assessment of Ultra-Efficient Lamps; Department for Environment, Food and Rural Affairs, UK, 2009.

Navigant (2012). Life–cycle assessment of energy and environmen- tal impacts of LED lighting products. Part I–II. U.S. Department of Energy, 2012.

Osram (2009). Life Cycle Assessment of Illuminants. A Comparison of Light Bulbs, Compact Fluorescent Lamps and LED Lamps. Executive Summary, 2009.

Osram (2011). PARATHOM CL A 40 Warm White DTC DIM BES. Material Declaration Sheet and RoHS Compliance Declaration.

Principi, P.; Fioretti, R. (2014). A comparative life cycle assessment of luminaires for general lighting for the office – compact fluorescent (CFL) vs Light Emitting Diode (LED) – a case study. Journal of Cleaner Production, Volume 83, 15 November 2014, Pages 96–107.

Schweitzer, S.; Sommer, C.; Hartmann, P.; Fulmek, P.; Nicolics, J.; Pachler, P.; Hoschopf, H.; Schrank, F.; Langer, G.; Wenzl, F. P. (2014). A comprehensive discussion on colour conversion element design of phosphor converted LEDs; Journal of Solid State Lighting, 1:18, 2014.

Statistik Austria (2015a). Gesamtenergiebilanz; Energiestatistik: Energiebilanzen Österreich 1970 bis 2013. Erstellt am 12.01.2015. (Energy statistics Austria; German).

Tähkämö, L.; Martinsons, C.; Bazzana, M.; Zissis, G. (2013). Life cycle assessment of light-emitting diode downlight luminaire - A case study, Int. Journ. of Life Cycle Assessment, June 2013.

Tähkämö, L.; Martinsons, C.; Ravel, P.; Grannec, F.; Zissis, G. (2014). Solid State Lighting Annex: Life Cycle Assessment of Solid State Lighting. Final Report; Energy Efficient End-Use Equipment, Int. Energy Agency, 2014.

Tähkämö, L.; Halonen, L. (2015). Life cycle assessment of road lighting luminaires – Comparison of light–emitting diode and high–pressure sodium technologies. Journal of Cleaner Production 93 (2015) 234–242.

Tasch, Stefan; Pachler, Peter; Roth, Gundula; Tews, Walter; Kempfert, Wolfgang; Starick, Detlef (Inventors); Tridonic Optoelectronics GmbH 8380 Jennersdorf (AT); Litec GBR 17489 Greifswald (DE); Leuchtstoffwerk Breitungen GmbH 98597 Breitungen (DE); TOYODA GOSEI CO., LTD. Nishikasugai-gun, Aichi-ken 452-8564 (JP) (Assignee) (2010). Light Source Comprising a Light-Emitting Element. Patent EP1352431 B1, 21.04.2010.

U.S. Department of Energy (2012). Residential Lighting End-Use Consumption Study: Estimation Framework and Initial Estimates, 2012.

Villa, C.; Bremond, R.; Saint-Jacques, E. (2016). Assessment of pedestrian discomfort glare from urban LED lighting. Lighting Res. Technol. 2016, 0: 1–26.

Von Nauckhoff, M. H. (2011). Strategische Metalle und Seltene Erden. Investieren in Technologiemetalle und Hightech-Metalle: Indium, Wismut, Terbium & Co.; Finanzbuch Verlag, 2011.

Wilburn, D.R. (2012). Byproduct metals and rare-earth elements used in the production of light–emitting diodes – Overview of principal sources of supply and material requirements for selected markets: U.S. Geological Survey Scientific Investigations Report 2012–5215, 15 p.; http://pubs.usgs.gov/sir/2012/5215/ [04/2019].

Zissis, Georges (2016). Quality and Health Aspects of LED Lamps for the European Residential Lighting Market. LED professional Review. Trends & Technologies for Future Lighting Solutions, July/August 2016, Issue 56, pp. 40–56.

Zissis, Georges; Bertoldi, Paolo (2018). Status of LED-Lighting world market in 2017, Ispra, European Commission, 2018, JRCxxxxx.

(1) Datasheet: Shenzhen Solar-LED Lighting, SoonLiTte Lighting Systems, SNF284-50W, 2015.

(2) Datasheet: Trilux LIQ 90-AB7L/20000-740 24G1S ETDD, 2015.

(3) Datasheet: Trilux LIQ 90-AB7L/15000-740 20G1S ETDD, 2015.

(4) Datasheet: Trilux LIQ 90-AB7L/13500-730 16G1S ETDD, 2015.

(5) Datasheet: ledplanet.at: 98W LED street lighting Panel SL9104, 2015.
(6) Datasheet: ledplanet.at: LED street lamp 5700K day light, 2015.
(7) Amazon: Horoz LED COB aluminum street light HL194L, 2015.
(8) Datasheet: Amazon: Horoz LED aluminum street light HL195L, 2015.
(9) Datasheet: Philips LED-Lampe E27 2700 Kelvin, 9 Watt, 806 Lumen, 2015.
(10) Datasheet: Philips LED-Lampe E27 2700 Kelvin, 13Watt, 1055 Lumen, 2015.
(11) Datasheet: Philips LED Lampe E27 2700 Kelvin, 6 Watt, 470 Lumen, 2015.
(12) Datasheet: OSRAM LED Classic A, E27, 10 Watt warmweiß matt, dimmbar, 2015.
(13) Datasheet: OSRAM LED Star Classic A40, E27, 6 Watt (40 Watt Ersatz) warmweiß matt, 2015.
(14) Datasheet: OSRAM LED Classic A 11W (75W-Ersatz) warmweiß matt E27 dimmbar, 2015.
(15) Datasheet: OSRAM LED Star Classic A / 75 Watt Ersatz / kaltweiß / matt / E27, 2015.
(16) Datasheet: IKEA LEDARE LED-Lampe E27, dimmbar, 400 lm, 6.3 W, 2015.
(17) Datasheet: TaoTronics desk lamp 9 W, 450 lm, 40000 h, 2015.
(18) Datasheet: Philips EyeCare desk lamp, 6 W, 300 lm, 15000 h, 2015.
(19) Datasheet: IKEA Stockholm floor/reading lamp, 350 lm, 9 W, 20000 h, 2015.
(20) Datasheet: Obi Einbauleuchte EEK, 15 W, 1200 lm, 20000 h, 2015.
(21) Datasheet: Zumtobel, ceiling lamp:Helissa LED D360/D450, 2015.
(22) Datasheet: Zumtobel, floor lamp, SFERA-S 33/33W LED840 SC WH SR2, 4000K, 2015.
(23) Statistik Austria (2015b). Gesamtenergiebilanz. Energiestatistik Österreich, 2015.
(24) Statistik Austria (2015c). Vorläufige Energiebilanz Österreich 2014. Erstellt am 05.05.2015. (Preliminary Energy statistics, German; Austria; 2015).
(25) Bittermann, W. (2014). Statistik Austria: Energiebilanzen Österreich 1970–2013, Detail-information, 2014. (Total energy balances, details; Austria 2014).
(26) Statistik Austria (2010): Energiestatistik: Energiegesamtrechnung Österreich 1999 bis 2008. Erstellt am: 29.03.2010. (Energy statistics Austria, German).
(27) Bittermann, W.; Gollner, M.; Strasser, M. (2014). Modellierung des Stromverbrauchs in den privaten Haushalten Österreichs nach unterschiedlichen Verwendungszwecken; Statistik Austria, Direktion Raumwirtschaft, Energie, 2014. (Modeling of the electricity consumption in private households, German).

Part II

8. Environmental Assessment of Electric and Electronic Products

Electronic products are characterised by containing of a large number of components, made of various valuable and/or hazardous materials in the range from macro– to nano–scale, which are manufactured by means of sophisticated technologies and ancillaries. Therefore, to evaluate the environmental impact of electronic devices, a detailed knowledge of the complex processes is necessary, particularly, if similar products or product groups should be compared with each other.

8.1. Life Cycle of a Product

To evaluate the environmental impact of a product, the model of the product life cycle offers an ordered structure to systematically acquire all environmental relevant data for further assessments. The ideal life cycle of an electronic product is shown in figure 8.1.

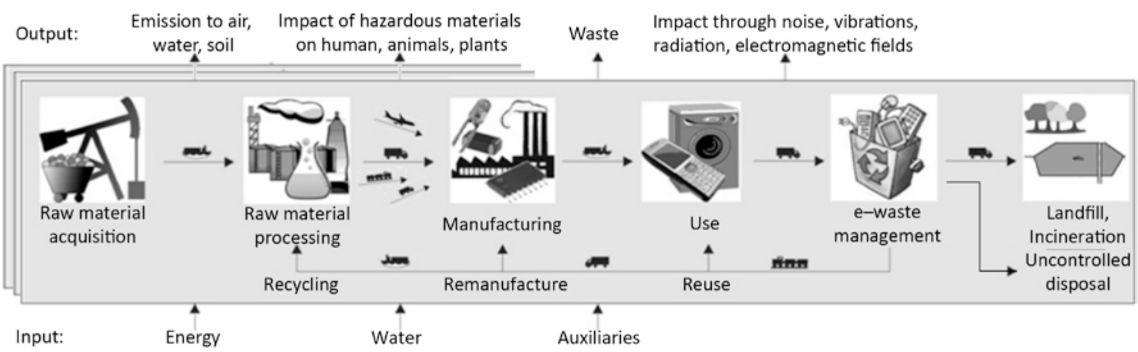


Figure 8.1. Schematic life cycle of an electric or electronic product. (Own illustration)

The cradle–to–grave life cycle of a device consists of five stages: raw material acquisition, raw material processing, fabrication of the product, use phase, and end–of–life management. In the case of electronic products, which are assembled from a large number of components, each part has its own life cycle from raw material acquisition to components manufacturing, which are put together in the assembly state of the electronic device. These parallel life cycle phases are indicated by the multiple shadows of the upstream part of figure 8.1. and represent the supply chain of a large number of independent manufacturers, which can be located in different continents.

The manufacturing phase is followed by the use phase, which is characterised by electrical energy consumption and maintenance in terms e.g. of battery change. Some household equipment additionally consumes water, detergents, and filters in the use phase. The repairing of electric and electronic devices, which is part of the use phase, seems to recede into the background due to the shortening of the service life of the products and uneconomical spare parts, respectively.

The end–of–life management of electric and electronic products can be divided into the following procedures, dependent on country specific standards and regulations:

- Formal recycling
- Informal recycling
- Hazardous waste landfill
- Non–hazardous waste landfill
- Waste incineration
- Hazardous waste incineration
- Legal and illegal export
- Uncontrolled disposal

Electronic waste (e–waste) management in terms of circular economy aims to recover valuable parts and materials, and to separate hazardous substances. The most sustainable e–waste management follows the hierarchy of re–use, remanufacture, and recycling of the full device, the components and material fractions. Formal recycling means the e–waste treatment based on the best available techniques, including the protection of people and the environment. Informal recycling relates to so called backyard workshops, which are widely spread in Asia. At this, valuable components and materials are manually extracted under unhealthy conditions. Figure 8.2. shows emissions to air, emission from wastewater, and some examples of solid waste of formal recycling compared to informal recycling in China (Hong et al., 2015).

		Unit	ET-D	ET-ND
			Amount	Amount
Direct air emissions	Particulates	g	2.38	5.29×10^{4}
	Nitrogen oxides	g	3.48	96.99
	Ammonia	g	8.66×10^{-2}	2.04
	Hydrogen chloride	g	1.53	39.24
	Sulfuric acid mist	g	9.75×10^{-2}	2.04
Direct emissions from wastewater	Nickel	g	7.98×10^{-3}	1.11×10^{-2}
	Petroleum	g	1.06×10^{-2}	3.31×10^{-2}
	Zinc	g	3.19×10^{-3}	2.26×10^{-2}
	Cyanogen	g	6.4×10^{-4}	6.4×10^{-4}
	Suspended solids	g	6.39×10^{-2}	3.19
	Chemical oxygen demand (COD)	g	1.72	5.43
	Chromium	g	4.79×10^{-3}	4.79×10^{-3}
	Copper	g	7.98×10^{-3}	19.2
	Cadmium	g	7.98×10^{-3}	7.98×10^{-3}
	Lead	g	3.19×10^{-2}	3.19×10^{-2}
	Ammonia–nitrogen	g	2.63×10^{-2}	5.28×10^{-2}
Solid waste	Antimony	g		85.17
	Arsenic	g		4.26
	Barium	g		207.14
	Beryllium	g		2.0×10^{-4}

Figure 8.2. Examples of hazardous substance flows of formal (ET-D) and informal (ET-ND) disassembling processes in China per tonne e–waste (Hong et al., 2015).

Between the life cycle stages there are manifold transport routes, which are covered by all available means of transport, such as airplanes, container ships, trucks, trains, and small transporters. In many cases, high–tech components are pre–manufactured in Europe or USA, then sent to Asia for further processing and transferred forward to other countries for sale on their market.

The system model of the full life cycle including all its processes is surrounded by the system boundary, which delimits it from input and output flows. Typical input flows are energy, water, and ancillary materials such as packing materials as well as process and cleaning chemicals. The typical output flows are waste, emissions to air, water, and soil, as well as immaterial emissions, such as electromagnetic fields, noise, vibration, etc. These inputs and outputs refer to each life cycle step and process within the system boundary.

8.2. Material Flow Analysis

The method of material flow analysis (MFA) is a tool to describe complex material systems to locate and quantify material and substance flows along the life cycle of e.g. an electronic product. MFA provides useful information regarding the patterns of resource use and the losses of materials entering the environment. The term "material" includes goods which consist of different material compounds and substances, such as a personal computer or a plastic compound with flame retardants or phthalates, and substances which consist of one chemical element or a pure chemical compound, such as copper or carbon dioxide. Materials can be raw materials of geogenic origin or anthropogenic materials which are physically or chemically processed by human hands. Within a defined spatial and temporal system boundary, each flow begins and ends in a "process" (or so called "activity") or it crosses the system boundary, respectively. Each process describes the transformation, the transport and / or the stock of materials. The key characteristic of a process is the law of the conservation of mass:

$$\Sigma \text{ input flows} = \Sigma \text{ output flows} + \text{stock change}$$

Material flows serve as a basis for environmental impact assessment. (Brunner & Rechberger, 2004)

Material flow analysis is a suitable tool to evaluate and visualise e.g. hidden hazardous and toxic substance flows, and valuable metal and rare earth flows in electronic devices. One of the challenges of MFA is to choose the right system boundary including applicable spatial and timely limits. MFA is a different tool for environmental decision support than life cycle assessment (LCA) which is presented in the following section. The two methods are different with respect to the actual subject of investigation and the definition of the system boundary. LCA and MFA complement each other and they increase the study quality in both domains. (Laner & Rechberger, 2016)

8.3. Life Cycle Assessment – LCA

A meaningful environmental assessment assumes that the assessment procedure is transparent, comparable, and generally applicable. The aim is to open a discussion, to improve products, and to avoid environmental damages. The most commonly applied methods are attributional LCA and consequential LCA. The attributional LCA is characterised by using normative cut–off rules and allocations and isolates the investigated product from other influences such as physical and socio–economic conditions. It is a useful tool for the product and process design phase. The consequential LCA is more comprehensive and links processes of a system to the change in the environment that occurs as a consequence of this activity. The functional unit also changes and allocation problems are circumvented.

The framework to conduct a life cycle assessment is standardised by the ISO 14040 and ISO 14044 (ISO, 2006a; ISO, 2006b), which consists of four mandatory phases:

- Goal and scope definition
- Inventory analysis
- Impact assessment
- Interpretation

Figure 8.3. shows the stages and direct application opportunities of a life cycle assessment study and its results.

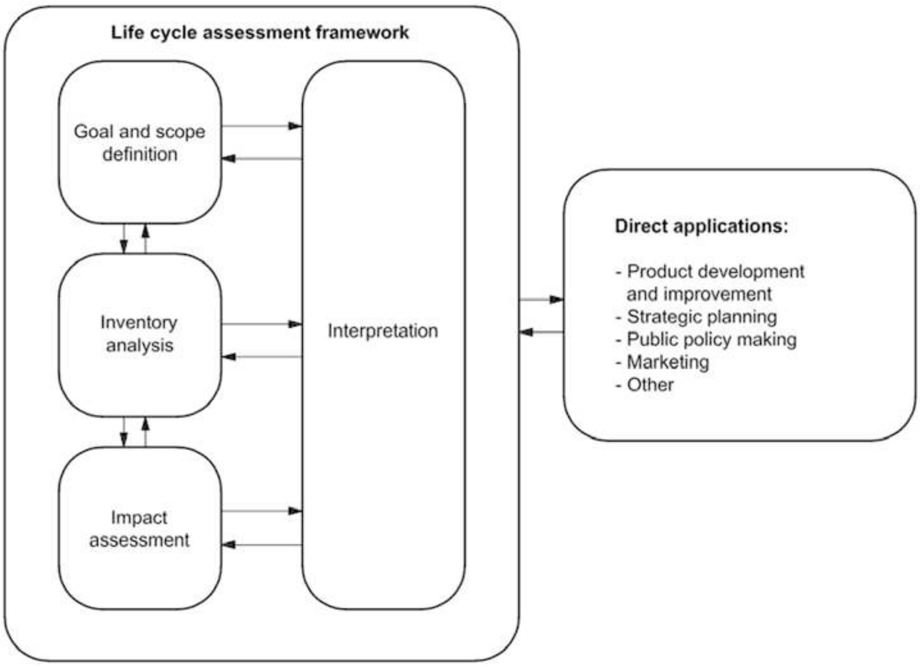


Figure 8.3. Stages of a LCA (ISO, 2006a).

It is also possible to conduct only a life cycle inventory analysis (LCI), which undergoes the same procedure except for the life cycle impact assessment (LCIA). An LCI analysis is useful to describe material flows along the value–added chain.

8.3.1. Goal and Scope Definition

This stage includes the definition and explanation of the background, the structure, the functional unit, the system boundary, the origin and depth of data, including the time frame of the assessed sample. In the goal and scope definition it is also defined, if a single case study of one product system is investigated or a comparative life cycle assessment of products based on different technologies with the same application is performed. Further, the assessment method and the considered impact categories have to be defined.

The functional unit is a reference, to which inputs and outputs are related. It's definition must ensure that comparisons between different products covered in the LCA or results of sensitivity analysis remains fair. In case of electrical equipment, the functional unit can for example be the entire device over its expected lifetime, the waste treatment of one tonne of e–waste, or the manufacturing of a specific electronic component. Another approach in case of the comparison of lamp technologies, for example, is a functional unit defined with one lumen–hour.

The system boundary is the most critical definition in a life cycle assessment. It defines the product systems and their considered processes as well as related material and substance flows. The system boundary indicates, whether important environmental impacts are inside or outside the boundary and hence the assessment. On the global level, for example, if the e–waste management is assumed to be treated according to the best available technology, the results of the life cycle impact assessment will not identify the contamination of air, water, and soil and the poisoning of people in the course of informal waste treatment, which is the case in many places of the world. The following processes and flows should be considered according to the LCA standard:

- Acquisition of raw materials
- Inputs and outputs in the main manufacturing/processing sequence
- Distribution/transportation
- Production and use of fuels, electrical energy and heat
- Use and maintenance of products
- Disposal of process wastes and products
- Recovery of used products (including reuse, recycling and energy recovery)
- Manufacture of ancillary materials
- Manufacture, maintenance and decommissioning of capital equipment
- Additional operations, such as lighting and heating

8.3.2. Life Cycle Inventory Analysis – LCI

The inventory analysis is the most quantitative and scientific part of any LCA study. It defines the system by the material and energy flows exchanged with the environment, including product trees and reference flows. Based on that, input and output data have to be collected considering cut–off and allocation rules.

Cut–off criteria are the specification of the amount of material or energy associated with the respective process to be excluded from the study, due to expected insignificant environmental impacts. Allocation means the partitioning of input and output flows of a process between the study sample and other processes outside the system boundary of the study.

Data collecting is the most difficult part, particularly for electronic products, due to the large uncertainties, the complex supply chain, and proprietary process data of manufacturers. Furthermore, electronic components and fabrication processes differ in a wide range and they are rapidly developing, so that there are often no actual data available in the databases.

8.3.3. Life Cycle Impact Assessment – LCIA

The LCIA assigns the flows exchanged with the environment identified in the life cycle inventory analysis to the selected impact categories, to provide information on the environmental issues associated with the inputs and outputs of the production of the considered product. Impact categories can be, for example, human toxicity potential, water toxicity potential, or land use. The selection of the categories has a great influence on the result of the overall evaluation of the electrical product, which will be discussed in more detail in Part II. The life cycle assessment standard does not prescribe a specific method or which impact categories have to be considered. It is left to the author to choose the appropriate method. The results of a LCIA are a relative expression of potential environmental impacts to the reference unit. The mandatory elements of a life cycle impact assessment are the following:

- *Selection* of impact categories, category indicators, and characterisation models
- *Assignment* of LCI results (classification)
- *Calculation* of category indicator results (characterisation)

After conducting these three stages, the category indicator results (LCIA results or LCIA profile) are obtained. Optionally, the results can be normalised to a reference product system or as an example shown in figure 8.4., to life cycle phases.

Figure 8.4. shows exemplarily the environmental impact results of the single case LCA study of a Fairphone with an assumed use phase of three year operation time (Güvendik, 2014).

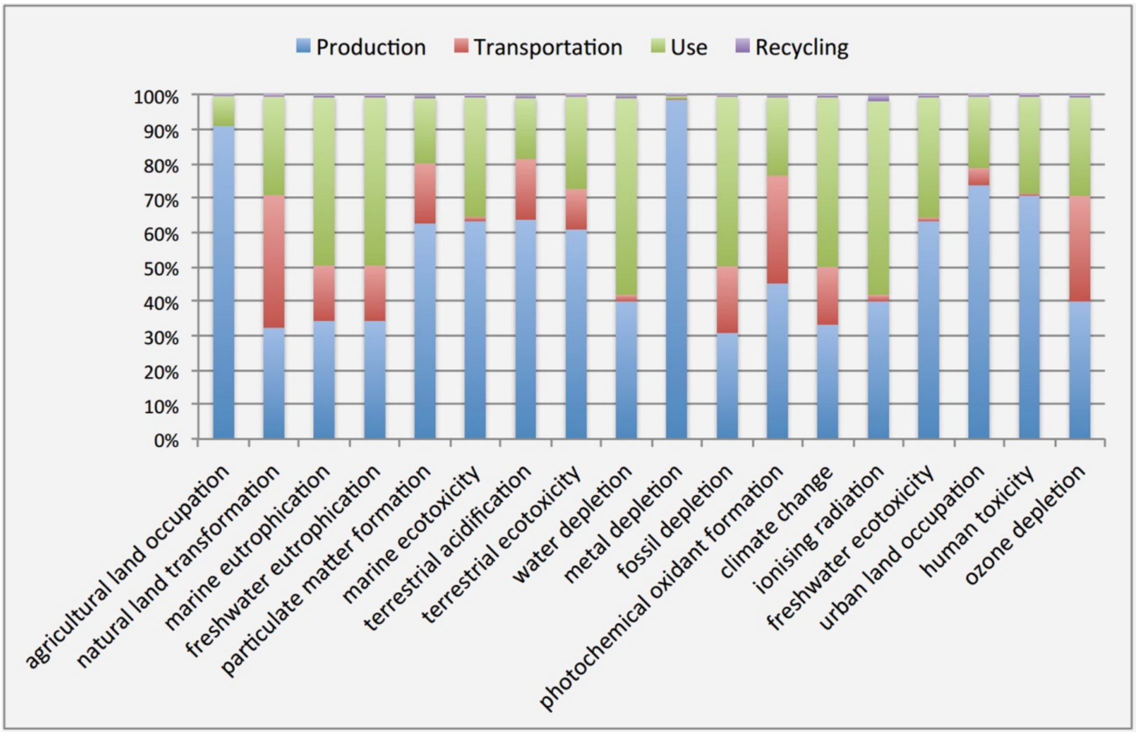


Figure 8.4. Contribution of each life cycle phase on each of the impact categories. (Güvendik, 2014)

It is important to note that the results of a LCIA address only the metabolic problems within the chosen system boundaries and only the categories included in the assessment, which is specified in the goal and scope definition. An uncritical generalisation of study results can lead to a significant underestimation or overlooking of other potential impacts. An example is the current global focus on the "carbon footprint", which favours nuclear power plants in electrical energy generation, compared to nearly all other energy sources without taking other emissions than Green House gases or other risks into consideration (Schlömer et al., 2014).

8.3.4. Life Cycle Interpretation

The findings of the LCA are systematically analysed to identify and quantify the environmental impact conclusions according to the goal and scope definitions. It is an iterative procedure through all stages of the LCA. A sensitivity study should be performed in that way, that environmental key parameters are varied, due to uncertain data or unexpected events, to evaluate the robustness of the study results.

The last step of a life cycle assessment, according to the standard, is a critical review by an internal or external scientific expert, who verifies, whether the LCA study has met the requirements for the methodology, the data and interpretation according to the goal and scope definitions.

The annex of the standard remarks, that LCA does not include sustainability assessment and economic or social aspects.

It is important to distinguish between a LCA study, which consists of all stages according to figure 8.1., and a life cycle inventory study (LCI), which leaves off the stage of the Life Cycle Impact Assessment. LCI studies describe and quantify the material flows exchanged with the environment. Both variants are covered by the standard.

8.4. Types of Life Cycle Assessments

There are several approaches to conduct a life cycle assessment. The most frequently used are attributional and consequential life cycle assessments, which are both environmental life cycle assessment (E–LCA) methods. In the following, commonly used and extended new approaches are presented.

8.4.1. Attributional Life Cycle Assessment (A–LCA)

The most commonly used life cycle assessment method for electronic devices is the attributional LCA. The A–LCA describes the environmental properties of the life cycle of a product system. The related life cycle inventory data consider the flows from and to the environment within a chosen temporal window. The aim is to describe environmentally relevant physical flows to and from a life cycle and its subsystems. Inputs and outputs are attributed to the functional unit (Ekvall et al., 2016). All material and energy flows, that is all environmental burdens, are allocated to the actual first source. Consequently, reused or recycled materials have a low share of the input and output flows.

8.4.2. Consequential Life Cycle Assessment (C–LCA)

The aim of a consequential LCA is to describe how environmentally relevant physical flows will *change* in response to possible decisions. In the life cycle there are activities included in the product system, which are expected to change as a consequence of a change in demand for the functional unit (Ekvall et al., 2016).

The difference between the A–LCA und C–LCA is illustrated by the following example: Advanced electronic technologies include a small amount of rare earths and/or strategic metals. Commonly, an A–LCA is conducted to evaluate the environmental impact of the new product technology based on established databases. A future global large scale production of the new electronic device leads to extremely high demand of the limited resources and therefore the necessity to find new recycling technologies or to substitute materials, which will change the environmental impact of the product life cycle. These scenarios are not visible in an A–LCA, but can be considered and estimated in a C–LCA.

8.4.3. Streamlined Life Cycle Assessment (SLCA)

A "full" life cycle assessment from cradle–to–grave needs a large amount of data to evaluate the environmental impact. The idea of a streamlined LCA is to simplify and shorten the LCA procedure by targeting and limiting the goal of the study and therefore, adjusting the goal and scope definitions. A typical approach to streamline the life cycle inventory is omitting life cycle stages, e.g. cradle–to–gate studies, which exclude the use phase and the end–of–life phase. A further example is to remove upstream processes and only include the manufacture of the final material, the use phase and waste management (Todd & Curran, 1999).

8.4.4. Social and Socio–Economic Life Cycle Assessment S–LCA

S–LCA complements the environmental life cycle assessment by social and socio–economic aspects of all life cycle phases. They are linked, for example, to the behaviour of enterprises or impacts on social capital. The life cycle inventory includes data such as number of working hours, allowed or not allowed labour unions, child labour, educational level, etc. Figure 8.5. schematically shows the S–LCA framework. The life cycle impact assessment encounters positive as well as negative impacts. (UNEP, 2009)

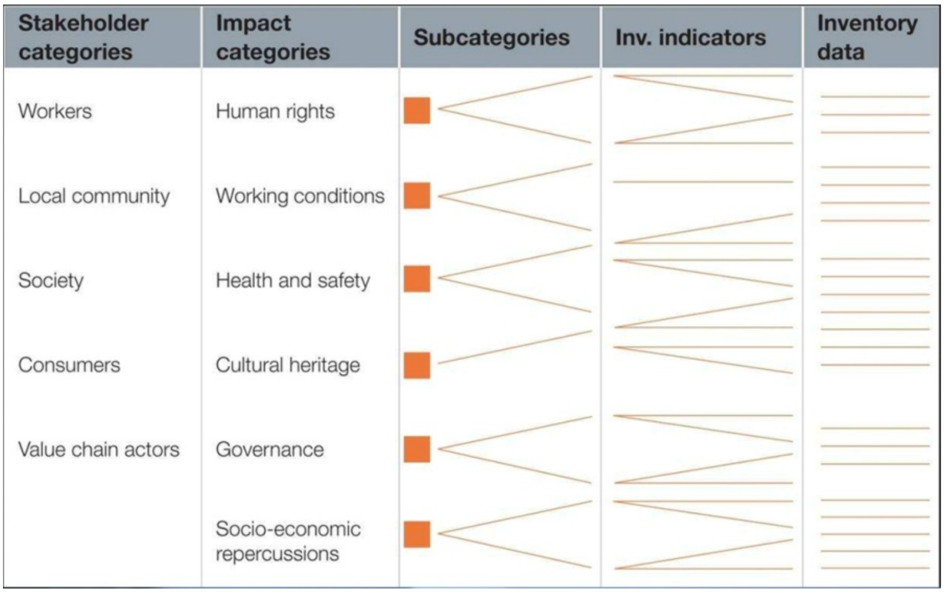


Figure 8.5. Social LCA Framework. (UNEP, 2013)

The framework in figure 8.5. is systematically processed and documented. As an example, figure 8.6. shows an excerpt of the social impact documentation of cobalt mining workers in the Democratic Republic of Congo.

Stakeholder Category: Workers	
Subcategory	Impact description
Child labour	28% of the total workers are children under the age of 15 (legal limit in DRC), some of them are as young as 6.
Fair salary	A miner usually earns between US$3 and US$5 for a day's work, although sometimes a concentrate deposit can yield up to US$30 a day. (Average daily expense in Katanga for a five-person household is US$2.5.) The revenue is comparatively higher for digging (a task forbidden
Working hours	Compared to working hours in the formal private sector (39 hours/week), a full-time artisanal miner performs between 52 and 59 hours/week (additional 35% to 52% workload) and thus exceeds the threshold of 48 hours/week for mine work and even the maximum workload of 56
Health and safety	Significantly higher urinary concentrations of As, Cd, Co, Cu, Pb and U were observed among communities living in a radius of 10 km from mines or smelting plants, especially children

Figure 8.6. Excerpt of social impacts on cobalt mining workers in the DR Congo (Valdivia et al., 2011).

Figure 8.7. shows a different representation of the impact results of a S–LCA, in this case an excerpt of an eco–labelled notebook along the entire life cycle.

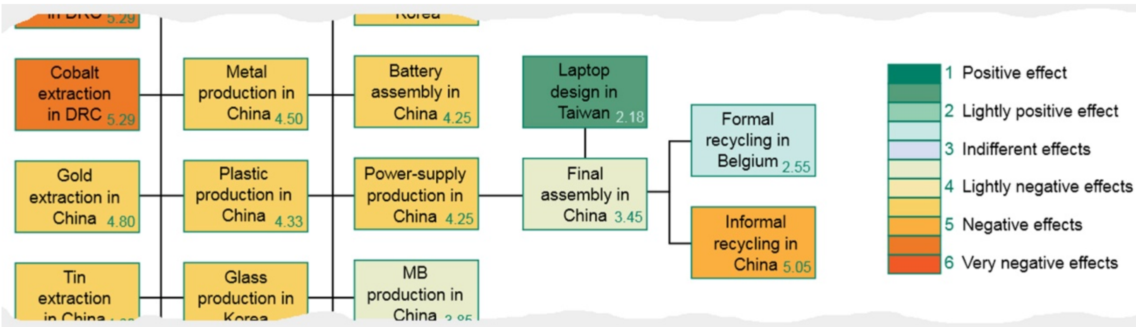


Figure 8.7. Excerpt of the S–LCA for the stakeholder group "value chain actors" and the stakeholder group "workers" of an eco–labelled notebook (Valdivia et al., 2011).

8.4.5. Working Environment Life Cycle Assessment WE–LCA

The conventional environmental LCA is focused on the potential impacts on the external environment. The Working Environment LCA evaluates the impacts on humans by the product system along the life cycle. It examines whether environmental product improvements are implemented at the expense of a deteriorated working environment. The WE–LCA evaluates the emission impacts in the working environment, from raw material acquisition to waste treatment and is aggregated to the functional unit of the product. (UNEP, 2009)

8.4.6. Life Cycle Costing LCC

Life cycle costing compiles and assesses all costs related to a product over its life cycle. It includes not only the purchase costs, but all other costs which are caused during the entire lifetime and disposal. It has been developed for long living products, such as railways, buildings, and investment goods. Mainly industry and governments apply this method, but it can be also applied for small scale products (UNEP, 2009). End–of–life treatment is not included in the life cycle costing (Hunkeler et al, 2008). The life cycle costs differ by the profit rates, dependent on the perspective of a producer or consumer.

8.4.7. Environmental Life Cycle Costing E–LCC

Environmental life cycle costing uses equally functional units and system boundaries as life cycle assessments for cradle to grave calculation and therefore also include waste treatment. All costs covered by any actor, which concern the life cycle of the product, including the costs of current and future external effects. (Hunkeler et al., 2008; UNEP, 2009)

8.4.8. Life Cycle Sustainability Assessment LCSA

The method of life cycle sustainability assessment encompasses all three pillars of sustainability: LCA, S–LCA, and LCC. The environmental, economic, and social negative impacts and benefits of the same product system are evaluated separately. The system boundaries of the assessments can be different, according to their impact relevance. For example, the product design phase is not relevant for the environmental LCA, but for the social S–LCA. The use phase is not relevant for the S–LCA, but it has relevance for the environmental LCA. Figure 8.8. schematically shows the different system boundaries of a LCSA in the life cycle of a product system. (Valdivia et al., 2011)

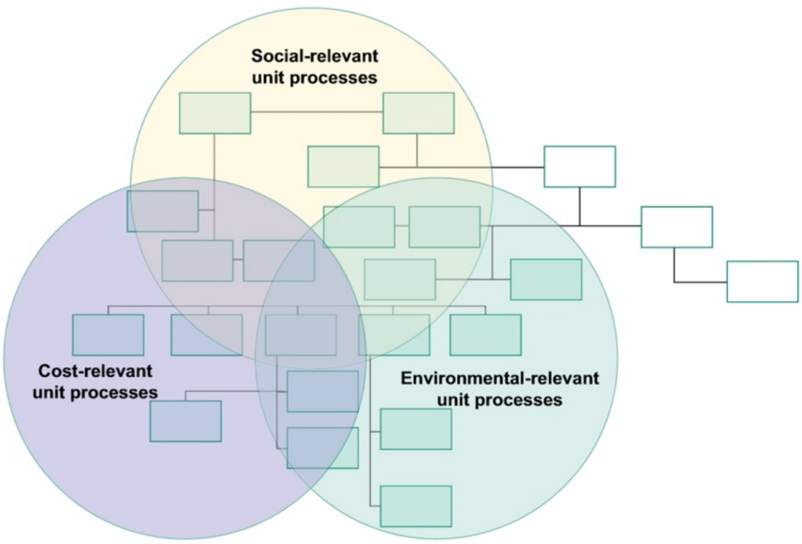


Figure 8.8. System boundaries of a LCSA. (Valdivia et al., 2011)

The LCSA method, which is supported by the United Nations, focuses on the working conditions of the Global South, therefore the use phase is not in the scope of the S–LCA. Concerning electric and electronic products, the situation is different in the Global North, where the use phase of consumer electronics in combination with social media becomes a large social problem for certain age groups. Therefore, a S–LCA for these product groups is necessary and interesting.

8.5. Methods and Impact Categories for Life Cycle Assessments

Two basic assessment methodologies can be distinguished, the midpoint and endpoint methods such as CML and ReCiPe (Hauschild & Huijbregts, 2015) as well as highly aggregated metabolic indicators such as Ecoindicator 99 (VROM, 2000), Sustainable Process Index–SPI (Narodoslawsky & Krotscheck, 1996), and Material Input per Unit of Service–MIPS (Schmidt–Bleek, 1998).

The idea is to follow the path of a hazardous substance or material in the cause–effect chain as shown in figure 8.9. The indicator for the midpoint impact category is, in this case, an early stage where the chemical flows into the lake and increases the concentration which might lead to a dangerous level. Midpoint indicators are metabolic indicators and therefore, they are easier to reconcile with the entire LCA method. They "translate" environmental objectives into metabolic value categories. Midpoint impact categories are for example water eutrophication, ozone depletion, acidification, etc. The endpoint method indicates the impact of the chemical only at the end of the cause–effect chain, in this case the extinction of fishes. Endpoint indicators are environment–oriented and therefore, they are not directly accessible by metabolic analysis.

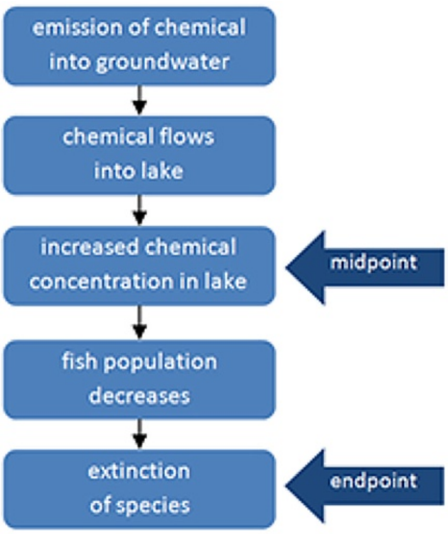


Figure 8.9. Example of a cause–effect chain (PRe, 2016).

Midpoint methods are CML–IA (CML, 2016), EDIP 2003 (PRE, 2018, p. 8), EPS 2015 (PRE, 2018, p. 11), IMPACT 2002+ (Humbert et al., 2014), LIME2 (Itsubo & Inaba, 2012), LUCAS (Toffoletto et al., 2007), ReCiPe (Goedkoop, 2013), and TRACI 2.1 (Bare, 2012). The methods have a different number of impact categories between 9 and 25 out of the 91 possible (partly overlapping) impact categories (Hauschild & Huijbregts, 2015).

Endpoint methods are Swiss Ecoscarcity 2013 (Frischknecht & Büsser Knöpfel, 2013), Ecoindicator 99 (VROM, 2000) and EPS (PRE, 2018, p. 11), which include a small number of categories such as human health, ecosystem quality, or resources. A detailed listing of the included categories can be found in (Hauschild & Huijbregts, 2015). In literature, the distinction between endpoint and highly aggregated methods are not consistently defined.

A third variant of assessment approach is a mixture of the midpoint and endpoint method, which promises an improved reflection of the environmental performance. It is applied to the methodologies of LIME2 (Itsubo & Inaba, 2012), ReCiPe (Goedkoop, 2013), IMPACT2002+ (Humbert et al., 2014).

Further LCIA methods are carbon footprint, eco–efficiency assessment, resource efficiency assessment, input–output and hybrid LCA, material flow analysis, (Finkbeiner, 2016), MEEuP2005 (VHK, 2005), and LUCAS (Toffoletto et al., 2007). The methods are described in the ILCD handbook by the European Union (ILCD, 2010). Highly aggregated life cycle assessment methods are MIPS–Material Input per Service Unit (Schmidt–Bleek, 1998) and the method SPI–Sustainable Process Index (Narodoslawsky & Krotscheck, 1996).

The life cycle inventory material and energy flows in the investigated product system are, with the help of databases, assigned to elementary output flows to air, water, and soil, which are further assigned and summarised to the respective impact categories, see figure 8.10. The ILCD reference elementary flow table includes 43000 datasets of different elementary flows as of August 2016 (JRC, 2016).

The aggregation of the material flows into one endpoint or a highly aggregated indicator facilitates the comparison of environmental impact results of different products or processes and provides vivid results. It is applied in the product design phase as well as a tool to support rethinking processes in the course of public relation work and education. On the other hand, a single index insinuates easy answers for complex questions. Numbers represent objectivity in our society without revealing which subjective assumptions were made in their calculation in order to get this result. Aggregation models differ with regard to period under consideration, assumptions, and system boundaries, but in particular regarding the normative framework of the aggregation. Therefore, the discussion about material flows and impact categories to be included should be at interdisciplinary level (Stahl, 1998).

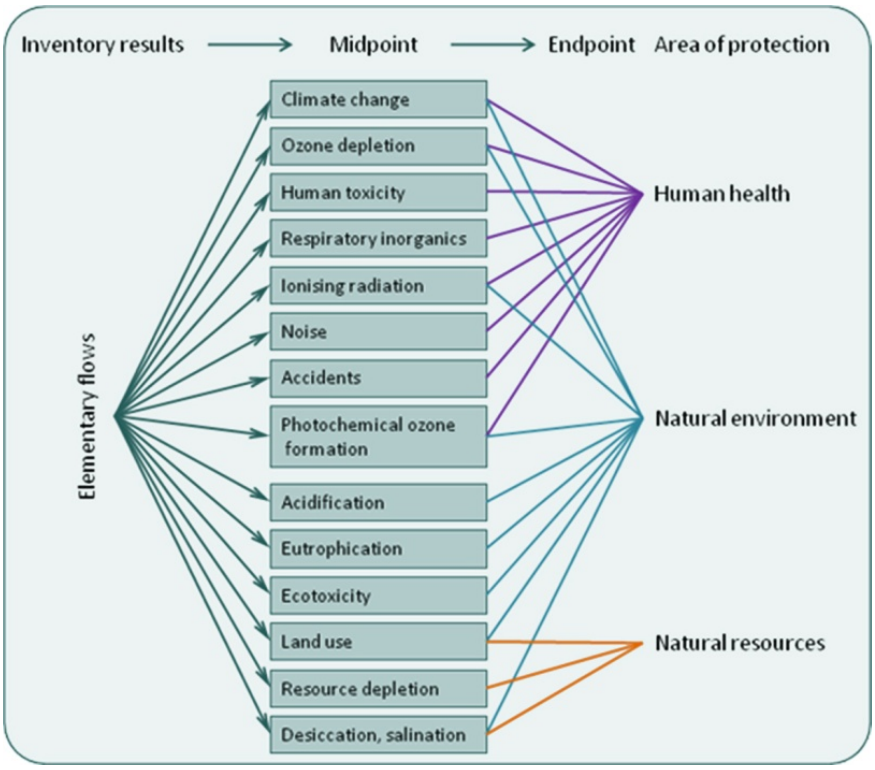


Figure 8.10. Schematic assignment of elementary flows to impact categories and aggregation to endpoint indicators (JRC, 2016).

From the point of view of electrical engineering, a comparison between different LCIA methods is conducted in the study by (Steiner et al., 2005), where the question concerning the point of time to replace a washing machine, due to environmental reasons is investigated. Figure 8.11. shows the different weighting results of the life cycle impact assessments, dependent on the considered impact categories and LCIA methods.

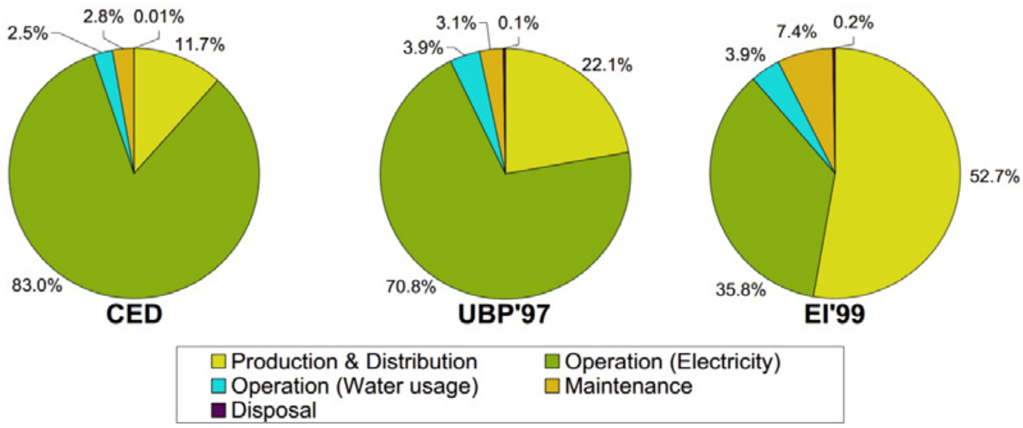


Figure 8.11. The relative shares of the different life cycle stages resulting from the calculation of the cumulative energy demand (CED), the ecological scarcity points 97 (UBP'97) and the Eco-indicator points 99 (EI'99) for a modern washing machine. (Steiner et al., 2005)

Calculated with the CED method, the use phase dominates the life cycle impact with 83 %. In contrast, the calculation with the EI´99 method shows, that the manufacturing phase has a share of 53 % and the use phase has only 36 %. The replacement of the machine with one of higher energy efficiency in the use phase is much earlier recommended in the case of the CED results than in the case of calculating with the Ecoindicator `99 method. The different weighting between the respective impact categories of the three assessment methods leads to a different weighting between manufacturing and use phase between the applied methods.

8.6. Critical Aspects of Standardised Life Cycle Assessments for Electronic Products

Electronic devices are characterised by a large number of different components from thousands of varying suppliers all over the world. These components are partially made of highly purified raw materials, with the help of highly purified chemicals and sophisticated energy intensive processes, under clean room conditions. The main life cycle inventory data source for the electronics sector is the ecoinvent database (Ecoinvent, 2019). The datasets relating electric and electronic equipment are collected and aggregated by a relative small number of central European companies and the associated documentations give a precise overview on assumptions, simplifications, uncertainties, and data gaps. The problem which derives from this fact is that all over the world this European driven database is used to conduct life cycle assessments of electronic products, due to the lack of other data sources.

It must be assumed that there are large life cycle inventory data gaps in the manufacturing phase. Many studies complain about the insufficient data situation, due to corporate secrets, and it seems that it is easier to obtain energy consumption data, than material and ancillary process chemical data. Besides the process data, particular attention should be paid on the selected raw materials. Particularly advanced materials, made of highly purified pre–manufactured materials, that means, for example 99.99 % purity, are not sufficiently represented in the database. This concerns ceramic powders, process gases, process chemicals, as well as rare earths and other doping materials. The degree of additional manufacturing efforts is not easy to estimate, but could be illustrated with a model of miniaturisation presented in section 8.7. With this background information the question arises, whether a comparative life cycle assessment of similar electronic products is appropriate.

The statements so far only concern the data selection for the life cycle inventory analysis. A further critical aspect is the functional unit, which for electric and electronic products in most cases is commonly one device with its nominal lifetime. In the case of comparative studies, the inventory data are normalised to a reference unit to ensure equal and comparable conditions. That means, for example, if one device has the double lifetime, the other one is inventoried as two devices with double manufacturing data. This general approach does not consider, firstly, the individual operation time per year, and secondly, how many devices are used at the same time. In other words, the consumer behaviour is not considered and therefore not a parameter in the life cycle inventory. This aspect can be the cause for an unexpected rebound effect following a technology change towards more energy efficient devices, which can induce to longer daily operation time per device and the parallel operation of several devices.

Rebound effect in this case means, that energy efficiency actions do not reduce the global energy consumption, according to the calculated values and new technical parameters, but may even increase it, due to changing consumer behaviour. This experience has been made for lighting equipment, which is described in detail in Chapter 7.

The effect of an unconsidered rebound effect is shown in figure 8.12. The attributional life cycle approach is independent of the number of pieces and the environmental impact is linearly related to it. In the case of equal numbers of pieces, the global environmental impact for product A is larger than for product B. In the case of a rebound effect, the product B is multiple parallel in operating usage. Beyond a certain number of pieces, the product group B has a larger total environmental impact than the product group A, with an individually higher environmental impact.

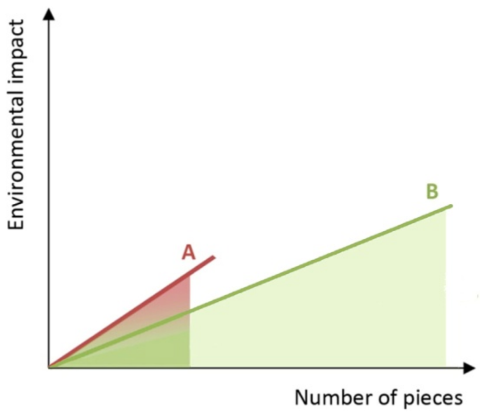


Figure 8.12. Total Environmental impact of two product systems dependent on the respective number of pieces.

Theoretical considerations show that besides the different number of pieces, another phenomenon can turn the global impact for the products A and B. It can be assumed that the dependence of the global environmental impact on the number of pieces is not linear, but shows a polynomial curve. Figure 8.13. shows an example of two products, which have different impact curves and the point of intersection changes the global environmental preference of the product groups. If the polynomial functions have more inflection points, more points of intersection occur.

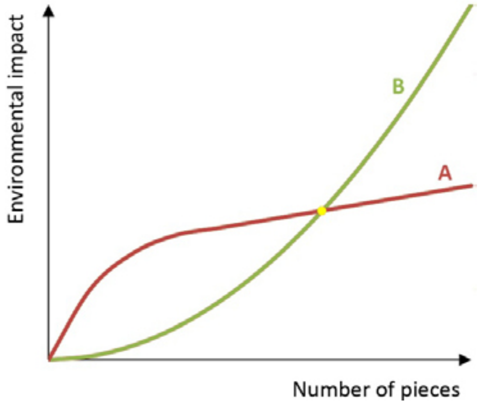


Figure 8.13. Polynominal curves of the environmental impact of two product systems dependent on the number of pieces.

The change of the curves can be affected for example by enlarging the production site and equipment, improving processing technologies, change from renewable to fossil energy generation due to higher consumption, change of the manufacturing location, other supply chains, change of the material quality, and many more.

So far, the considerations are all independent of the location. Life cycle assessments commonly assign the production and the use phase of one product to one location. Electronic devices can have very different locations concerning their production and usage location. Asia holds the largest part of manufacturing. From the Asian point of view, efficient products, which generally have a higher manufacturing impact, increase their country environmental burden for export products. The same products, which are used in another country than its production country, reduce the environmental burden of the consumer countries. Similar considerations can be made concerning the electronic waste. The European Union is the only region, where e–waste treatment is strictly regulated. All other regions, particularly Africa, India, and China have a high level of pollution caused by electronic waste treatment. These facts are not considered in the databases.

The so called "planned obsolescence", which means a calculated or deliberately accepted premature failure of a component, should be considered in an additional environmental devaluation of the respective component by a defined factor. The referring components are typically plastic parts instead of metals with a limited lifetime, or thermally undersized materials. The correction factor should be in the range of a new manufactured full device for the remaining theoretical lifetime. One possible way is to reduce the calculated lifetime of the entire device to the limited lifetime of the respective early breaking material. Concurrently, the choice of the functional unit should be extended to a defined total operation time, e.g. 10 years, to ensure the consideration of resulting higher manufacturing efforts.

8.7. Miniaturisation Model of Components and Processes

From the electrotechnical point of view, the data acquisition systems and assessment methods for performing a life cycle assessment have been developed for "macroscopic" and "simple" product systems, which consist of relatively small amounts of different materials, processes, transport routes and without variations in the product design itself. To include the product sector of electronic equipment in the life cycle assessment databases, life cycle inventory data are generated by a relatively small number of Central Europe companies. Due to the complexity of the individual products and product groups, far reaching cross–sectorial assumptions and simplifications are made, which are documented in detail in the dataset documentations.

Electronic devices have a long and partly intransparent supply chain. It can be assumed that a not quantifiable number of subprocess steps, ancillary process materials, energy consumption, and transport routes are not considered in the databases. This especially applies to advanced new materials and nanomaterials, which are included in a large number of modern electronic devices. Particularly highly purified raw materials, rare earths, process chemicals, etc., do not seem to be sufficiently represented in the databases.

One possible approach to compensate this mismatch could be a miniaturisation factor, which could be applied to an overseen supply chain, to a grade of miniaturisation with simultaneous increase in the number of units, or to a grade of purification of materials. The following theoretical model should illustrate how the material turnover changes in the case of a diversification and miniaturisation of a process.

The reference model is a work piece with the dimensions of (10 x 10) cm with 1 cm in height, of which a circular part is cut out with a laser for further processing, see figure 8.14. In case of a miniaturisation by the factor 100, the total area and total mass remains equal. In the ideal case, the raw material demand and cutting waste initially do not change.

Considering real conditions, the separation cut consumes 1 mm material, which results in an additional material consumption of +10 %. In microelectronics the cutting width is in the range of 50 µm or less, depending on the cutting technology. Nevertheless, a specific percentage of additional raw materials is necessary.

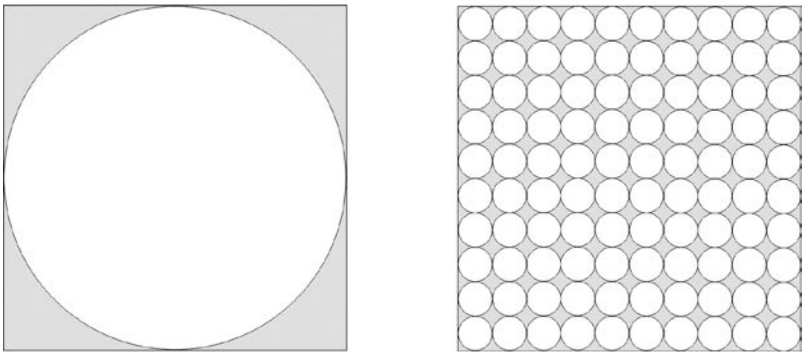


Figure 8.14. Schematical miniaturisation model.

These separation cuts provide an initial indication, that the process efforts noticeable increase. The miniaturised model has a 10–fold cutting length, that means, the laser consumes 10 times as much energy, plus 10 times as much polishing agents for the edges, and 10 times as much for all other ancillary materials, including their respective raw materials.

Similar considerations concern the packing materials. In the ideal case, the packing material efforts increase by the factor 2.5. If each packaged sample is protected with 5 mm wadding, the cardboard consumption increases by the factor 7.5 in the miniaturised case. The volume of the wadding increases by the factor 5. Consequently, the total mass and therefore the transport efforts also increase.

With this approach different scenarios can be simulated. Single components of an electronic device, or parts of it, are not manufactured at the location where the product is assembled. The parts are fabricated in a supply chain of tens or hundreds of companies in regional and international distances. It is not uncommon, that small high–tech components are pre–manufactured in Europe or in the USA, then packaged and sent to a second country for finishing, and finally again packaged and sent to a third country for the product assembly. Microscale electronic components are packed in tape–and–reel packages for automatic assembly lines and the mass of which is many times as much as the component itself. The transportation is probably carried out by airplane. Subsequently, the final product assembly is transported to the end consumer back to the Northern Countries. The transport efforts in standardised life cycle assessments for electronic products commonly results in the range of 1 % of the total impact and they are often neglected. The described complex transport chain is with high probability not accurately calculated in the standard datasets.

In summary, the LCA databases for electronic devices are only applicable to a certain extent. Due to the macroscopic origin and nature of life cycle assessment, the large extent of miniaturisation particularly of the electronics sector in terms of nanoparticles or fine surface structures is not sufficiently considered. Some examples are discussed for LED lamps in Chapter 10, for wafer manufacturing and nanoparticles in Chapter 13, and for embedded components in Chapter 14.

8.8. References

Bare, Jane (2012). Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI). User´s manual. EPA-United States Environmental Protection Agency. https://nepis.epa.gov/Adobe/PDF/P100HN53.pdf [02/2019].

CML-Department of Industrial Ecology (2016). CML-IA Characterisation Factors. Institute of Environmental Sciences (CML), Department of Industrial Ecology. Universiteit Leiden, NL. https://www.universiteitleiden.nl/en/research/research-output/science/cml-ia-characterisation-factors [02/2019].

Ekvall, Tomas; Azapagic, Adisa; Finnveden, Göran; Rydberg, Tomas; Weidema, Bo P.; Zamagni, Alessandra (2016). Attributional and consequential LCA in the ILCD handbook. International Journal of Life Cycle Assessment (2016) 21:293–296.

Ecoinvent (2019). http://www.ecoinvent.org/ [04/2019].

Finkbeiner, Matthias (2016). Special Types of Life Cycle Assessment (LCA–Compendium – The Complete World of Life Cycle Assessment), Springer, 2016, 399 pages.

Frischknecht, Rolf; Büsser Knöpfel, Sybille (2013). Swiss Eco-Factors 2013 according to the Ecological Scarcity Method. Methodological fundamentals and their application in Switzerland. FOEN-Federal Office for the Environment, öbu – works for sustainability, Bern, 2013, 256 pages.

Goedkoop, Mark; Heijungs, Reinout; Huijbregts, Mark; De Schryver, An; Struijs, Jaap; van Zelm, Rosalie (2013): ReCiPe 2008 – A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First Edition (revised). Ruimte en Milieu. Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer, 2013.

Güvendik, Merve (2014). From Smartphone to Futurephone. Assessing the Environmental Impacts of Different Circular Economy Scenarios of a Smartphone Using LCA. Master thesis, Delft University of Technology and Leiden University.

Hauschild, Michael Z.; Huijbregts, Mark A.J., Eds. (2015). Life Cycle Impact Assessment (LCA–Compendium – The Complete World of Life Cycle Assessment). Springer, 2015, 339 pages.

Hong, Jinglan; Shi, Wenxiao; Wang, Yutao; Chen, Wei; Li, Xiangzhi (2015). Life cycle assessment of electronic waste treatment. Waste Management 38 (2015) 357–365.

Humbert, Sébastien; De Schryver, An; Bengoa, Xavier; Margni, Manuele; Jolliet, Olivier (2014). IMPACT 2002+: User Guide, Draftfor version Q2.21. Quantis Sustainability counts. 47 pages.

Hunkeler, David; Lichtenvort, Kerstin; Rebitzer, Gerald, Eds. (2008). Environmental Life Cycle Costing. CRC Press Inc., 191 pages.

ILCD International Reference Life Cycle Data System Handbook (2010). Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment. European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy.

ISO 14040 (2006a). Environmental management – Life cycle assessment – Principles and framework.

ISO 14044 (2006b). Environmental management – Life cycle assessment – Requirements and guidelines.

Itsubo, N.; Inaba, A. (2012). LIME2: Life-cycle impact assessment method based on endpoint modeling. JLCA News Lett. Life-Cycle Assess. Soc. Jpn. Vol 12, 2012.

JRC Joint Research Centre (2016). Set of elementary flows (August 2016); http://eplca.jrc.ec.europa.eu/?page_id=1159 [10/2016].

Narodoslawsky M.; Krotscheck C.; (1995). The sustainable process index (SPI): Evaluating processes according to environmental compatibility; J. of Hazardous Materials, 1995, 41 (2+3), 383-397.

PRé Consultants (2016). https://www.pre-sustainability.com/consider-your-audience-when-doing-lca [10/2016].

PRé Consultants (2018). SimaPro Database Manual-Methods Library. https://www.pre-sustainability.com/download/manuals/DatabaseManualMethods.pdf [02/2019].

PRé Consultants (2018). SimaPro Database Manual-Methods Library. EPS 2015d and EPS 2015dx-Environmental Priority Strategies in product design. https://www.pre-sustainability.com/download/manuals/DatabaseManualMethods.pdf [02/2019].

Schlömer S., T. Bruckner, L. Fulton, E. Hertwich, A. McKinnon, D. Perczyk, J. Roy, R. Schaeffer, R. Sims, P. Smith, and R. Wiser, 2014: Annex III: Technology-specific cost and performance parameters. In: Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Edenhofer, O., R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, T. Zwickel and J.C. Minx (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Schmidt–Bleek, Friedrich (1998). Das MIPS Konzept: weniger Naturverbrauch mehr Lebensqualität durch Faktor 10, Droemer Knaur, München 1998.

Stahl, Beate (1998). Methodenvergleich und Methodenentwicklung zur Lösung der Bewertungsproblematik in produktbezogenen Ökobilanzen. Dissertation, University of Bremen, Germany.

Steiner, Roland; Faist Emmenegger, Mireille; Jungbluth, Niels; Frischknecht, Rolf (2005). Timely Replacement of White Goods – Investigation of Modern Appliances in a LCA. ESU–sevices fair consulting in sustainability, Uster, Switzerland. http://esu-services.ch/ [10/2016].

Todd, Joel Ann; Curran, Mary Ann (1999). Streamlined Life–Cycle Assessment: A Final Report from the SETAC North America Streamlined LCA Workshop. Society of Environmental Toxicology and Chemistry (SETAC) and SETAC Foundation for Environmental Education, July 1999.

Toffoletto, Laurence; Bulle, Cécile; Godin, Julie; Reid, Catherine; Deschênes, Louise (2007). LUCAS - A New LCIA Method Used for a Canadian-Specific Context. The International Journal of Life Cycle Assessment, March 2007, Volume 12, Issue 2, pp 93–102.

UNEP (2009). Guidelines for Social Life Cycle Assessment of Poducts. United Nations Environment Programme and SETAC. 104 pages. ISBN: 978-92-807-3021-0; http://www.lifecycleinitiative.org/ [10/2016].

UNEP (2013). The Methodological Sheets for Sub-categories in Social Life Cycle Assessment (S-LCA). Pre–Publication version. United Nations Environment Programme and SETAC. 152 pages. http://www.lifecycleinitiative.org/ [10/2016].

Valdivia, Sonia; Ugaya, Cássia Maria Lie; Sonnemann, Guido; Hildenbrand, Jutta; Eds. (2011). Towards a Life Cycle Sustainability Assessment. United Nations Environment Programme UNEP / SETAC Life Cycle Initiative, 2011, 86 pages. ISBN: 978-92-807-3175-0.

VHK-Van Holsteijn en Kemna BV (2005). Methodology Study Eco-design of Energy-using Products, Final Report, MEEUP Methodology Report. European Commission DG ENTR. Unit ENTR/G/3, Brussels, Belgium, 188 pages.

VROM-Ministry of Housing, Spatial Planning and the Environment (2000). Eco-indicator 99, Manual for Designers, A damage oriented method for Life Cycle Impact Assessment. The Hague, The Netherlands, October 2000, 49 pages. https://www.pre-sustainability.com/download/EI99_Manual.pdf [02/2019].

9. Life Cycle Assessment of Low Temperature Co–Fired Ceramics (LTCC) Substrates

Abstract: *This study evaluates the energy consumption, global warming potential (GWP), and material turnover of three different commercial low–temperature co–fired ceramic (LTCC) substrate materials under laboratory small–batch manufacturing conditions. The system boundary is a process chain from the green tape delivery to the co–fired substrate. Further, the ceramic powder production and a life cycle transportation scenario is considered. The study addresses four key questions, the selection of an appropriate functional unit, the comprehensiveness of environmental relevant statements based on selected midpoint indicators, the availability of life cycle inventory data and the consequences of excluded process steps, and the transferability of laboratory data to industrial processes. The GWP results of the LTCC substrates are in the range of 75 kg CO_2–e to 234 kg CO_2–e per kg within the system boundary and dependent on the tape material and the used co–firing furnace. Environmental hot spots are the furnaces, the compressor, and the laser equipment. The specific energy demand is both mass and geometry dependent. It is recommended to further investigate upstream manufacturing chains and chemical auxiliaries which are likewise expected to generate significant environmental pressures.*

9.1. Introduction

Electronic circuits consist of basic substrates which can be made of different materials dependent on costs and application of the electronic device. Besides the conventional and cheap FR4 material, which is made of glass fibre reinforced epoxy resin, ceramic substrates are suitable for specific physical and thermal requirements.

Low–temperature co–fired ceramics (LTCC) are applicable for high frequency circuits, the integration of electronic components used in harsh environment as well as for micro–fluidic devices, which consist of small channel structures surrounded by an electronic circuit to characterise smallest quantities of fluids. The used substrate glass/ceramic composites have lower dielectric loss than organic materials. The principal manufacturing procedure of a multilayer LTCC component is the following: The so–called "green tapes", which are unfired substrate layers, are laser cut and structured into the desired shape. Then the respective metallisation paste for the electric circuit is printed on the substrate by screen printing. Depending on the used technology, each metallised green substrate layer is separately co–fired and/or the layers are stacked together, pressed and laminated, and then the co–fired. The conductive metallisations consist of copper, gold or silver and their alloys, which require a sinter temperature below their melting points in the range of about 900°C to 1000°C dependent on the material. (Imanaka, 2010)

There is little information about the environmental performance of LTCC materials. A comparative midpoint LCA study by Herrmann et al. investigated the environmental impact of FR4 and ceramic based populated printed circuit boards (PCBs). The substrate material of the ceramics was aluminium oxide Al_2O_3 burned at 1500°C and the metallisation pastes of the electrical circuit were burned at 800°C. Each paste system (different conductive and dielectric pastes) was printed and burned separately. The study found i.a. that in case of the unpopulated ceramic PCB the shares of the global warming potential 100 (GWP 100) are the following: manufacturing of the ceramic substrate: 10 %, manufacturing of the matallisation pastes: 65 %, manufacturing of the metallisation layers (printing, burning, etc.): 25 %. The study only provides normalised values to 100 % and no absolute values. (Herrmann et al., 2001)

The study by Chen et al. compared four different substrate materials for system–in–a–package applications by assessing their mechanical reliability, electrical performance, and environmental influence, including i.a. LTCC substrates. The Monte Carlo simulation for GWP showed that the glass/ceramic material had an arithmetic mean of 1,099 g CO_2/kg with a standard variance of 87. The calculations were based on the assumption that the LTCC glass/ceramic material consists of 23.5 % ceramics, 55 % glass, 1 % plasticisers, 13 % solvents, and 6.5 % plastics by mass. (Chen et al., 2004)

In this work, life cycle assessment case studies of three different commercial LTCC substrate materials with similar manufacturing procedures in a laboratory gate–to–gate process chain are performed including two different co–firing technologies. Further, an international transport scenario is determined. The focus lies on the energy consumption, CO_2 emissions and ancillary material turnover.

In the frame of the thesis the study includes midpoint life cycle assessment methods and the material input per unit of service (MIPS). The question of the appropriate functional unit and the consequences of an inappropriate selection are addressed. Further, the availability of life cycle inventory data and the consequences of excluded process steps are investigated. The utilisation of laboratory data and their transferability to industrial processes, the related differences, and data uncertainties are discussed.

9.2. Materials and Methods

Three commercial LTCC substrate materials with different material properties, masses, and thicknesses are investigated: Heraeus CT 765 (green tape thickness: 50 μm), Heraeus CT 700 (green tape thickness: 200 μm), and Ceramtec GC (green tape thickness: 300 μm).

The consideration for choosing a functional unit is the following: It is assumed, that the environmental impact of a number of process steps for manufacturing of a co–fired LTCC device is not only dependent on the mass but also on the geometry of the sample.

Therefore, a functional unit is defined under consideration of the optimal utilisation of the existing laboratory equipment in terms of energy consumption whilst ensuring the widest possible application of the process parameters. The chosen functional unit is one sample of a laser structured 6–layer co–fired LTCC device with the dimensions of 6 x 8 cm^2 consisting of the pure substrate material without metallisation. All determined input flows are assigned to the sample unit as well as converted into a functional unit on kg–basis according to the standard unit of the ecoinvent life cycle inventory database (Ecoinvent, 2019). 1 kg of each LTCC material is processed in a small–batch production.

The study is divided into the following parts:

- Design of a flowchart for the full life cycle of a LTCC device
- Energy demand and global warming potential for the gate–to–gate manufacturing of a LTCC sample from the green tape delivery to the co–fired sample, extended by substrate powder manufacturing
- International transportation scenario
- Material turnover of the gate–to–gate process chain and transportation

9.2.1. Design of a Flowchart for the Full Life Cycle of a LTCC Device

An extended detailed generally valid flowchart of the life cycle of a functional co–fired LTCC substrate is developed including ecoinvent LCI data for raw material acquisition and ceramic powder preparation (Classen et al., 2009), possible process chain, measured gate–to–gate process data, and variations of material composition for substrates, metallisation pastes, and ancillary chemicals (Imanaka, 2010).

9.2.2. Energy Demand and GWP for the Gate–to–Gate Manufacturing

To evaluate the energy consumption and global warming potential of the gate–to–gate process the following process steps are considered: laser structuring, lamination, and co–firing of the green tapes with a belt furnace and alternatively with a batch furnace. All assumptions are based on empirical data of the laboratory equipment at the Institute of Sensor and Actuator Systems, TU Wien, Austria. The related equipment is accessible for energy measurements and the evaluation of ancillary materials.

A representative sample unit and process parameters are determined based on laboratory experience and load capacity of the belt furnace (BTU Systems, max. 1050°C). The samples are composed of six identical layers. Each layer is laser–cut (ROFIN–SINAR Laser RSM 100D) into a rectangle of 6 x 8 cm^2 including via holes. Subsequently, each layer is printed twice: first via hole printing and drying at 75°C, then printing of 20 % of the sample area and again drying at 75°C. The screen printing process itself is outside the system boundary and not further considered, because the mechanical operation is manually done without energy consumption. Further, the mass of the metallisation material is very small and there is no information about the material composition and the paste manufacturing. The average operating time of the drying furnace (HERAEUS, max. 250°C) is 2 minutes per printing step.

Six layers of the LTCC substrate each are pressed and laminated in an isostatic press for 10 minutes. A stack of 25 samples of HERAEUS CT 765, 8 samples of HERAEUS CT 700, and 5 samples of CERAMTEC GC, respectively, are jointly pressed including separation layers during each press procedure. The water in the press is electrically heated by a laboratory immersion heater (ELHO–Wärmetechnik). The average operation time of the heater is a warm–up time of 1 hour and a 10 minutes reheating time per press procedure. Before the press procedure, the samples are dried in the drying furnace. The average operation time of the drying furnace is 20 minutes per press procedure.

The subsequent co–firing process is carried out using two different types of furnaces. One charge of the 6–layer samples is co–fired in the belt furnace with six temperature zones adjusted to 350/580/859/900/900/851°C. Each sample is placed between two CERAMTEC A substrates for mechanical stabilisation reasons and put on a base substrate. The samples are co–fired for 2 hours with a mounting time of 5 minutes per sample. The warm–up time of the belt furnace is approximately 5.5 hours, this value is aliquotely added to each small batch production. It is presumed that the machine is mounted on a 24 hours basis.

The other charge of samples is co–fired in a batch furnace (LINN HIGH THERM HT 1600 M Vac Spezial) with a comparable core sintering profile to the belt furnace. The loading capacity of the batch furnace is 24 laminated ceramic samples per co–firing procedure. Parallel to the operation of both furnaces, a compressor has to be operated. The compressor provides pressed air with a pressure between 6 to 9 bar to generate a controlled gas flow to flush out the evaporated organic chemicals, that are binders, plasticiser, solvent, etc., during the firing process. The firing process is performed under oxygen atmosphere or other gases dependent on the fired materials. The energy consumption of the compressor is fully allocated to the operating time of the respective furnace.

The two furnace types are not always exchangeable. The belt furnace is characterised by previously adjusted temperature zones, which are passed through with a previously fixed constant speed of the conveyor belt. The total firing time is relatively short in the range of some hours and the temperature transitions between the zones are limited. For LTCC materials which need longer temperature transitions and holding times to get specific material properties, the belt furnace is less suitable. They have to be fired in a batch furnace with more variable temperature profile settings at the expense of higher total energy demand. The load capacity of the batch furnace is limited and for each firing cycle the total heating up and cooling down time to ambient temperature has to be run through including the respective energy demand.

The power consumptions of the laser, the drying furnace, the immersion heater, the belt furnace, the batch furnace, and the compressor are measured with a 3–phase power analyser (PCE–PA 8000). The warming–up and cooling–off times of the furnaces and heaters are aliquotely added to the operating times.

In this study, aluminium oxide is used as representative substrate powder material for LTCC, because it is the main component of the substrate CERAMTEC GC (Franz et al., 2012). There is no information about the material composition of the substrates HERAEUS CT 765 and HERAEUS CT 700. LCI data for the raw material acquisition of bauxite and the powder production of aluminium oxide as well as respective transport data are taken from literature (Classen et al., 2009).

The CO_2 emission of the electrical energy consumption was determined on basis of the UCTE low voltage electricity mix of 562 g CO_2 / kWh (Frischknecht et al., 2007). The results are compared with available literature data.

Cutting losses of the tapes and carrier foils, sample rejects, pastes and paste production, chemicals production, screen printing, usage of sacrificial material, as well as building infrastructure, using phase and end of life phase are not considered due to unavailability of data.

9.2.3. Material Turnover of the Gate–to–Gate Process Chain and Transportation

The method of MIPS–Material Input per Unit of Service evaluates the mass of the total material turnover along the life cycle of a product including the final product itself and all associated ancillary materials, fuel, waste, etc. (Schmidt-Bleek, 1993). In this study, the material turnover of the gate–to–gate process chain from the delivery of the green tape to the co–fired LTCC substrate and life cycle transportation is determined. The green tapes and the co–fired tapes are weighted (DENVER INSTRUMENT SI–234 A) to calculate the amount of organic chemicals which burn out during the co–firing process according to the weight loss, as well as the carrier foils and the packing materials of the green tape rolls, containing 1 roll of CERAMTEC GC material and 2 rolls of HERAEUS CT 765 material. To calculate the total cradle–to–gate mass of the packing for transportation it is assumed, that it is at least half of the sample net product weight.

9.2.4. International Transportation Scenario

In lack of available published precise data on international freight routes for electronic products, the following scenario for calculations is assumed: Since the majority of the mass production of electronic devices is located in East Asia (Gontermann, 2012), a transport route between two major international seaports, Shanghai and Hamburg, is assumed. Additionally, a realistic estimation of overland transport distances from/to LTCC manufacturing sites, user site and disposal site is added.

The transport service from the site of raw material mining to the site of powder production is 7.5 tkm per kg, based on estimations of average transoceanic import service to Germany (Classen et al., 2009). The total overland transport route between the sites of powder production and disposal is assumed with 7,000 km. Port to port distance from Shanghai to Hamburg is 19,700 km (Sea Rates, 2011).

It is assumed that the overland transport is covered by two types of means of transport, a 40 t truck for long distance transcontinental routes and a 3.5 t transporter for all other kinds of deliveries for shorter distances along the life cycle. Realistic values of fuel consumption are considered according to two technical reports. The 40 t truck has a diesel consumption of 35 l/100 km and an average load of 18 t (Zeitzen, 2007). 3,000 km of the land transport is assumed to be undertaken by a 3.5 t transporter with a diesel consumption of 13.2 l/100 km and a load of 1,290 kg (Unruh, 2008).

Further data are provided by the ecoinvent database (Ecoinvent, 2019): The transoceanic water transport service requires a heavy fuel consumption of 1.3 g/tkm (Spielmann et al., 2007). The heating value of diesel is 9.99 kWh/l, of heavy fuel oil 11.44 kWh/l. The density of diesel is 0.84 kg/l and of heavy fuel oil 1 kg/l (Jungbluth, 2007). The CO_2 emission of diesel fuel is assumed to be 3.172 kg CO_2/kg and the emission of transoceanic water transport is given as 4 g CO_2/tkm (Spielmann et al., 2007).

9.3. Flowchart for the Full Life Cycle of a LTCC Device

Figure 9.1. shows the detailed flowchart of a typical life cycle of a LTCC sample. There are optional various materials for the base substrate, paste systems and ancillary chemicals which are quoted in the literature. The most important materials and possible process steps including some process data are compiled in the figure according to the literature and empirical data of the laboratory (Chartier et al., 1997; Hosokawa & Yokoyama, 2007; Ibanez–Garcia et al., 2008; Imanaka, 2010; Jantunen et al., 2004; Khoonga et al., 2010; US Patent, 2005; Yuping et al., 2000). Tape embossing, laser punching, via filling, pattern printing, resistance trimming, or usage of sacrificial material for micro–fluidic application are optional process steps depending on the product application.

Ceramics is based on different powder materials depending on recommended material properties. A selection of potentially used powder materials is listed in figure 9.1. on the top left (Balluch et al., 2008; Bienert et al., 2010; Imanaka, 2010; Jantunen et al., 2004; Sebastian, 2008; Tietz, 1994; US Patent, 2005; Yuping et al., 2000).

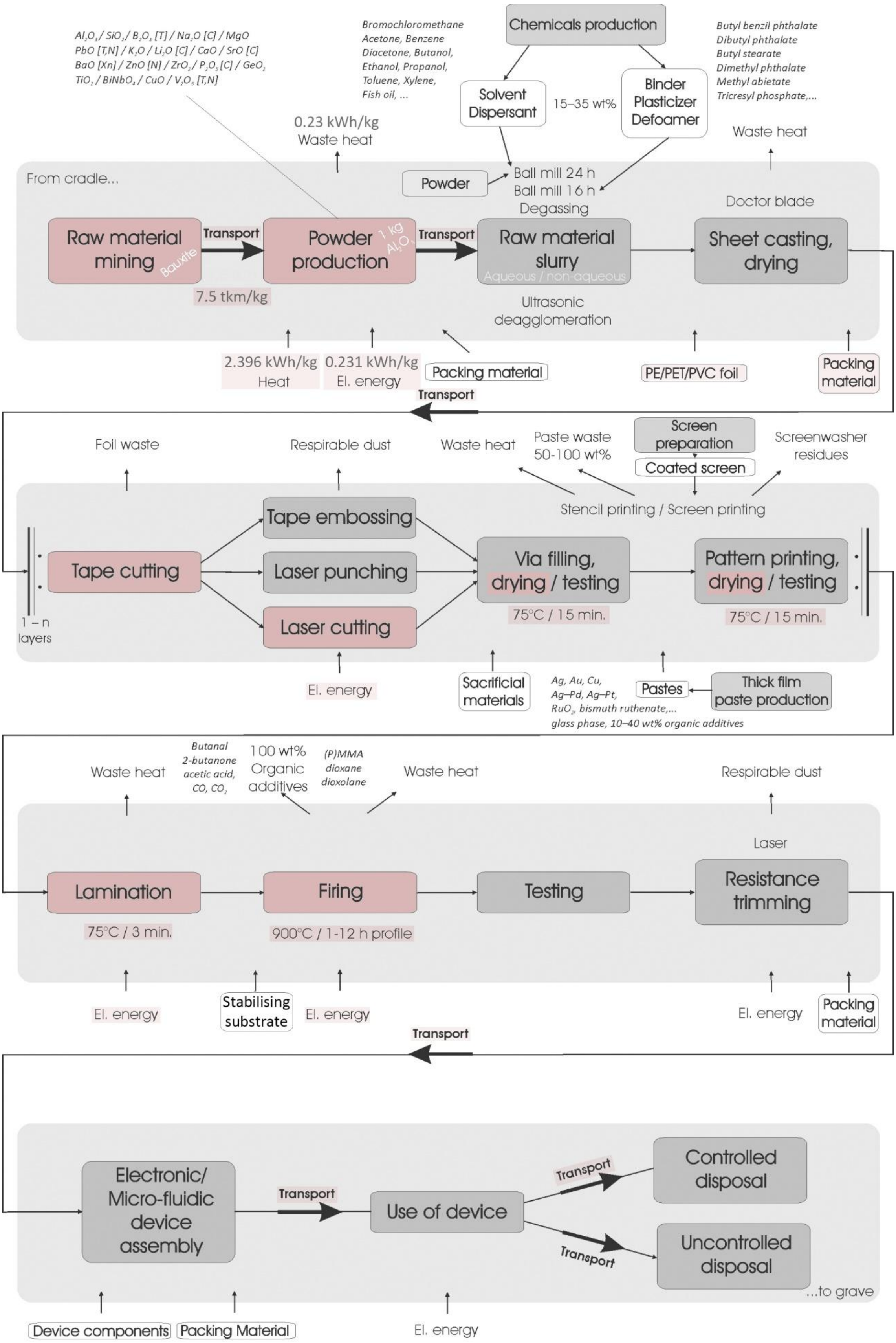


Figure 9.1. Schematic life cycle of a LTCC sample including relevant process steps, material variants, energy and process data for 1 kg fired LTCC substrate material. The rose coloured steps are included in the LCA study. (Own work)

For powder production, sheet casting, and sheet processing various solvents, binders, plasticisers, and dispersing agents are used to increase the formability, to obtain rheological properties, plasticity, and to control the pH value of the unfired material. The raw material slurry is manufactured by using a ball mill for several hours. The production of the glass/ceramic slurry is a high–precision process chain including highly purified materials and additives with small particle size down to nanoparticles. (Imanaka, 2010)

During the co–firing process all organic materials are completely decomposed and debound at about 500°C (Imanaka, 2010; Jantunen et al., 2004). Exemplarily, selected organic additives are listed in figure 9.1 on the top. The initial share of ancillary chemicals is between 15 % and 35 % and a part of the chemicals evaporates during each drying process step. Data are taken from literature (Heunisch et al., 2010; Hosokawa & Yokoyama, 2007; Ibanez–Garcia et al., 2008; Imanaka, 2010; Jantunen et al., 2004; Salam et al., 2000a; Salam et al., 2000b; US Patent, 2005). The percentage of organic materials of the green tape is between 5 wt% and 15 wt% (Ibanez–Garcia et al., 2008; Yuping et al., 2000).

The weight loss after co–firing correlates to the percentage of organic materials of the green tape rolls. The measured weight losses of two samples in this study are 10.31 wt% (HERAEUS CT 765) and 16.58 wt% (CERAMTEC GC). A binder burnout study by means of thermal analysis showed in the mass spectra of the flue gas during the co–firing process, amongst others, butanal 2-butanone and acetic acid (CERAMTEC A tape), and PMMA (polymethyl methacrylate) and dioxane dioxolane (CERAMTEC GC tape) (Balluch et al., 2008).

The production of LTCC devices for micro–fluidic application requires sacrificial material to fill the cavities before laminating and co–firing to prevent the green ceramic layers from collapsing. Sacrificial materials are PMMA films, PVA–S (polyvinyl acetate), or HEC (hydroxyethylcellulose) as powder and 2 % solution dissolved in de–ionized water (Wang et al., 2008).

This detailed description of the manufacturing process of an LTCC device is intended to show the complexity and variety of materials and processes. Many of these process steps, materials, and chemicals could not be evaluated because there are no detailed LCI data available. In figure 9.1., the LCA steps considered in this study are coloured rose.

9.4. Energy Demand of LTCC Manufacturing

As a first step, the electrical energy consumption of all energy driven equipment, including furnaces with the predefined respective temperature profile, is measured. The energy consumption of the ***belt furnace*** is measured during the warm–up and the co–firing time. Figure 9.2. shows the graphs of the real power and the energy consumption recorded for ten hours. The energy consumption of the belt furnace is 42.5 kWh for preheating to 900 °C and then 4.14 kWh per operating hour.

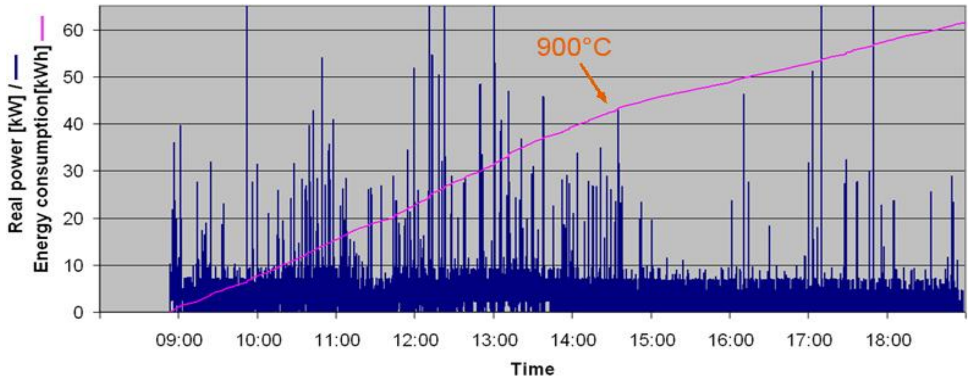


Figure 9.2. Measurement report of the real power and cumulative energy consumption of the belt furnace. The scale applies to both.

The ***batch furnace*** operates 6 hours (including the long cooling-off time) per firing profile with an energy consumption of 6.66 kWh. The energy consumption of the ***compressor*** is 3.57 kWh per operating hour. The energy consumption of the ***drying furnace*** is 0.36 kWh per operating hour.

To determine the energy consumption of the ***laser***, five different typical cutting profiles are adjusted and measured. Figure 9.3. shows the measurement report of the real power and the energy consumption of the laser. The measured average energy consumption for 60 laser structuring process steps is 1.88 kWh. The average energy consumption per sample is 0.22 kWh, including the cutting of auxiliary co–firing tapes. It was attempted to choose a most average laser structure profile possible. However, the actual structure can be very different dependent on the application.

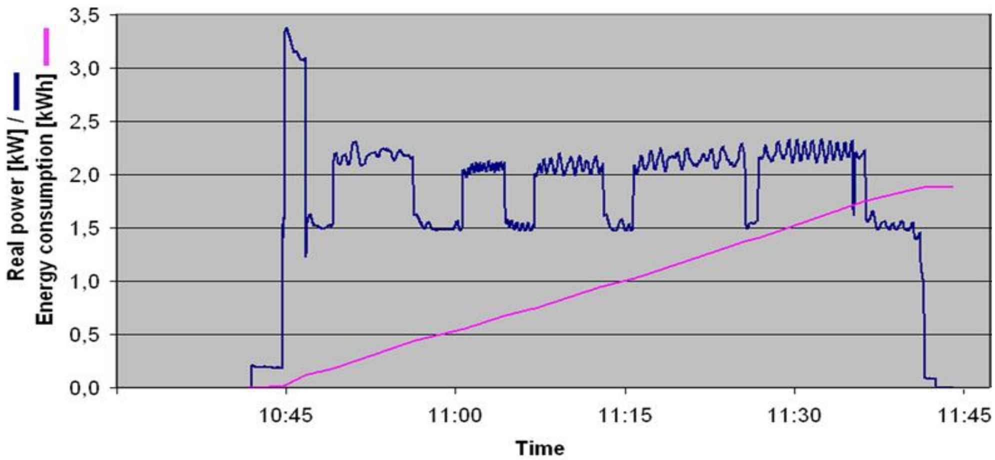


Figure 9.3. Measurement report of the real power and cumulative energy consumption of the laser. The scale applies to both.

All energy measurements and calculations of the life cycle within the system boundary, that is heat, transportation, and electrical energy, are normalised to one 6–layer sample and also 1 kg, respectively, for each LTCC material type. Differing thicknesses and densities of the LTCC tape materials result in different numbers of samples per 1 kg unit. Therefore, the number of 6–layer samples of 1 kg of co–fired material corresponds to 108 samples of HERAEUS CT 700, 265 samples of HERAEUS CT 765, and 81 samples of CERAMTEC GC. The results of the total energy consumption in the life cycle within the defined system boundaries for the three sample materials are shown in table 9.1. All values are standardised to a 6–layer sample unit and to a 1 kg unit.

Table 9.1. Cradle–to–gate energy consumption of LTCC substrate manufacturing within the system boundaries dependent on the furnace type.

Sample	Total energy consumption	
	belt furnace [kWh / sample]	batch furnace [kWh / sample]
Ceramtec GC	2.15	1.71
Heraeus CT 700	1.85	1.66
Heraeus CT 765	1.35	1.58
	belt furnace [kWh / kg]	batch furnace [kWh / kg]
Ceramtec GC	173	138
Heraeus CT 700	200	179
Heraeus CT 765	358	420

On a sample unit basis, the HERAEUS CT 765 material co–fired in the belt furnace has the lowest environmental impact. On a kg–basis, the CERAMTEC GC material co–fired in the batch furnace has the lowest environmental impact. The deviations of the total energy consumption range from 160 % to 250 % and the order of the degree of the energy consumption is reversed, depending on the method of normalisation to kg or area based sample unit.

The detailed energy consumption of the different processes and life cycle steps are presented in figure 9.4., normalised to a one sample unit, and in figure 9.5., normalised to a 1 kg unit.

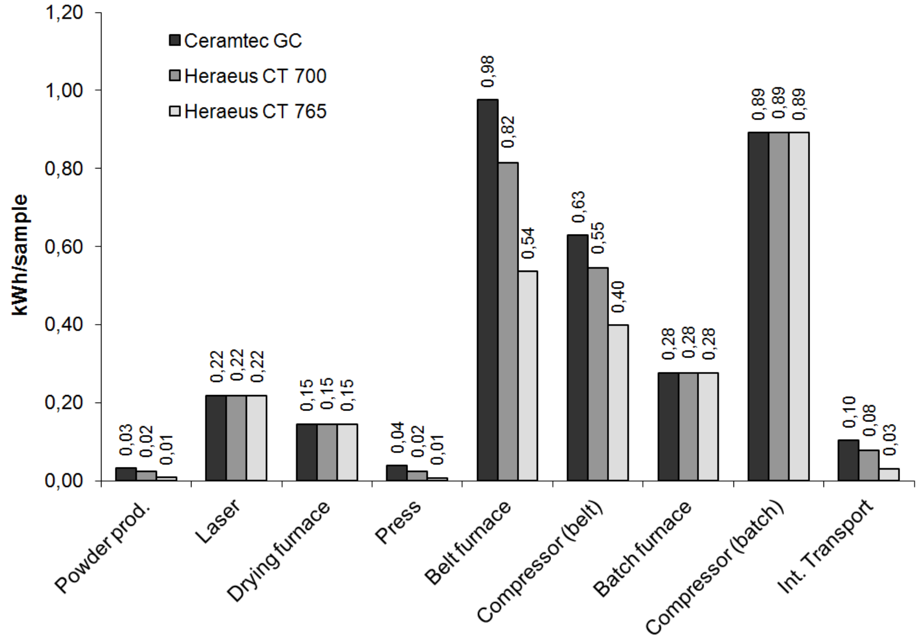


Figure 9.4. Energy consumption of three different LTCC substrates within the system boundary per sample unit.

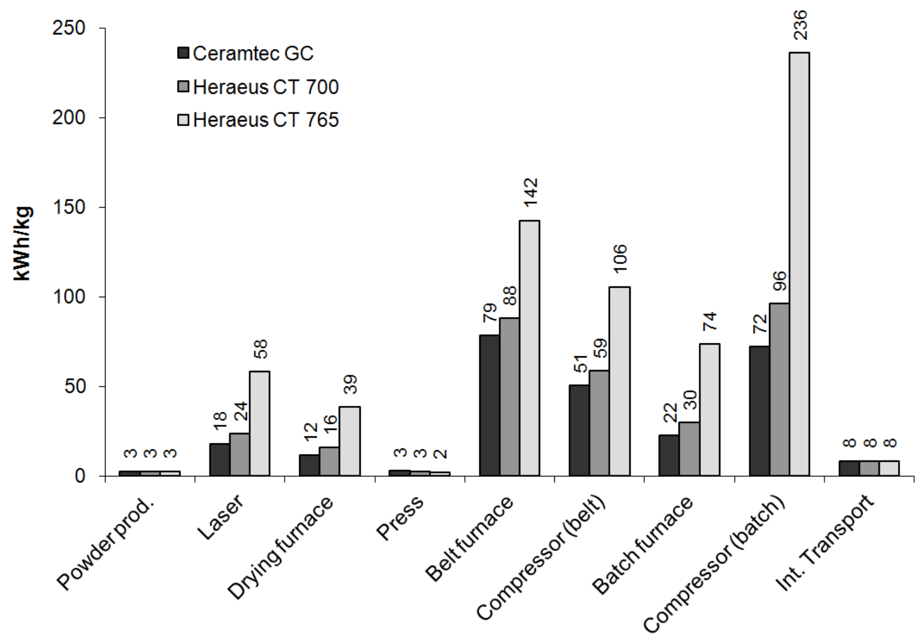


Figure 9.5. Energy consumption of three different LTCC substrates within the system boundary per kg.

Both figures show that powder production, lamination, and transportation have a low energy demand. The next higher energy consumption results from the drying furnace and the laser operation. By far, the largest energy consumption results from co–firing with both the belt furnace and the batch furnace in combination with the compressor.

Whereby, the batch furnace itself has a lower energy demand than the belt furnace, but a longer operation time which is reflected in the energy demand of the associated compressor.

There is one significant difference between small–batch and industrial production visible in figure 9.4.: The results for the belt furnace are expected to be on the one hand equal for all three materials on sample basis and on the other hand, generally, the belt furnace should have less energy demand than the batch furnace. In this case there are different numbers of pieces per 1 kg LTCC material each and due to the small sample number the allocated preheating time of the belt furnace has a large influence on the results. If the sample numbers for all materials are increased for example in the range of 10,000 pieces, still under laboratory conditions, the energy consumption of the belt furnace decreases to 0.36 kWh/sample for all three materials and analogue the compressor value.

The high energy consumption of the compressor was not expected. The compressor is part of the building infrastructure which is several decades old. Since the Ecodesign Directive of the European Commission included compressors into the working plan, it can be assumed that compressors have been recognised as highly inefficient and that energy efficiency measures have been taken across the EU (EU Directive, 2009). That should be an indication to check the energy consumption of old but still operating equipment besides the core machines.

The transfer of results from the laboratory experiment to industrial processes leads to some relevant shifts of the relations of the energy consumption. Besides the reduction of the energy consumption of the belt furnace discussed above, it is expected that the value will decrease even more, because large industrial devices have a wider conveyor belt and the stack height can be probably slightly increased. A state–of–the art compressor will also reduce the energy consumption of both furnace types. Industrial batch furnaces could have a larger firing chamber, but a reduction of the energy consumption in the range of the belt furnace is not expected due to the limited mounting space and the necessary preheating and cooling–off phase of each firing cycle.

The advantage of a particular type of furnace depends on the number of samples, the firing time of the LTCC material, the thickness of the tape, and the number of layers. A principal limitation of the choice are the required chemical and physical properties of the co–fired materials which could be considered in the product design phase. The belt furnace is preferred for a large number of samples and short co–firing profiles due to the long pre–heating time, whereas the batch furnace has a long cooling time and is suitable for a small number of pieces and long co–firing profiles.

The laser energy consumption is not expected to be significantly reduced on industrial scale and therefore it will become a second hot–spot besides the co–firing process. The drying furnace is expected to be more efficiently used. Nevertheless, its total energy demand depends on the number of process steps which need interim drying activities according to figure 9.1. The same applies to the laser operation, which is highly dependent on the product design and required process steps.

9.5. Global Warming Potential of LTCC Manufacturing

The CO_2 emissions are calculated on the basis of literature data. The results for the life cycle steps are proportional to the energy consumption according to section 9.4. within the life cycle steps, hence, the co–firing process has the highest environmental impact. Table 9.2. shows the values of the total CO_2 emissions within the system boundaries dependent on the co–firing method.

Table 9.2. Cradle–to–gate CO_2 emission of LTCC substrate manufacturing within the system boundaries dependent on the furnace type.

Sample	Total CO_2 emission	
	belt furnace [kg CO_2e / sample]	batch furnace [kg CO_2e / sample]
Ceramtec GC	1.170	0.930
Heraeus CT 700	1.019	0.912
Heraeus CT 765	0.750	0.881
	belt furnace [kg CO_2e / kg]	batch furnace [kg CO_2e / kg]
Ceramtec GC	95	75
Heraeus CT 700	110	98
Heraeus CT 765	199	234

The comparison with literature data show that the results for the CO_2 emissions of this study are far higher. The CO_2 emissions of an unspecified LTCC material is reported with 1.099 kg CO_2/kg (Chen et al., 2004). If only the lowest values of the laser, the drying furnace, and the lowest energy consumption of the co–firing furnace without all other life cycle steps are considered, the CO_2e emissions are 30 times higher.

The detailed CO_2 emissions of all considered processes and life cycle steps are presented in figure 9.6., normalised to a one sample unit, and in figure 9.7., normalised to a 1 kg unit.

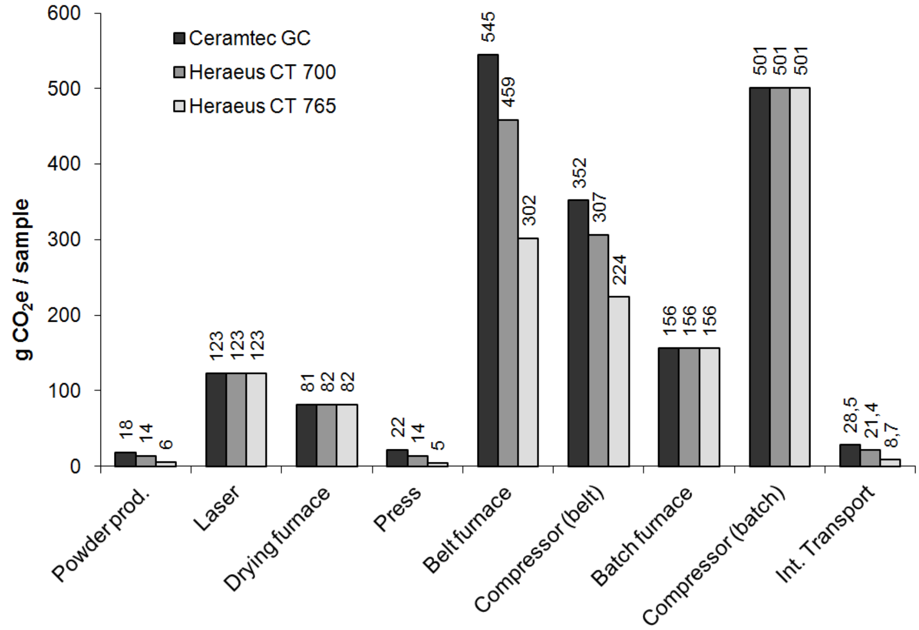


Figure 9.6. CO_2 emission of three different LTCC substrates within the system boundaries per sample unit.

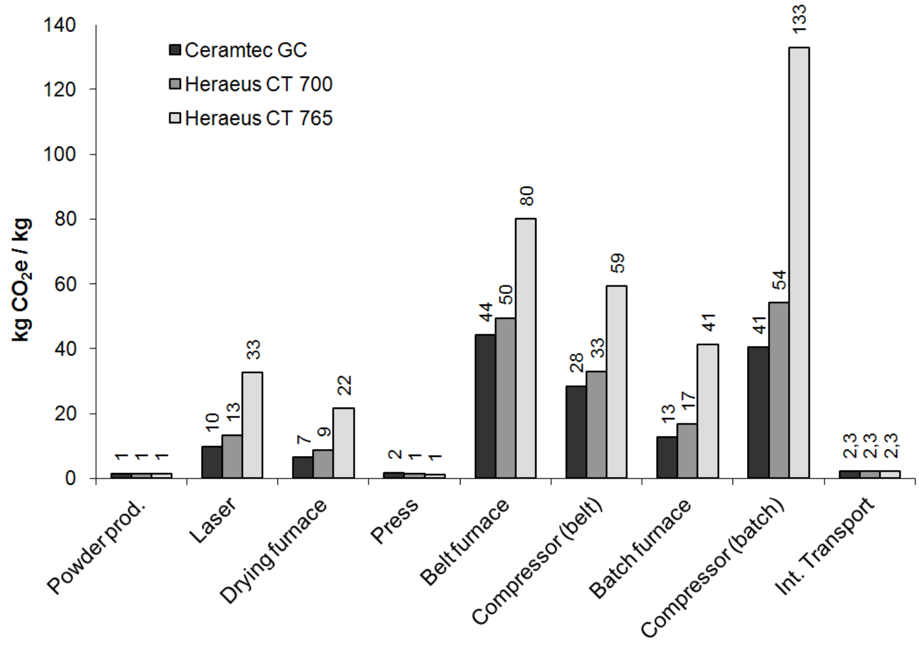


Figure 9.7. CO_2 emission of three different LTCC substrates within the system boundaries per kg unit.

However, the CO_2 emissions of individual process steps can be much higher or lower, depending on which energy source is used for energy production. From the environmental point of view, the proper choice of the production country with its respective energy mix is a key factor for the environmental performance of the LTCC device or component.

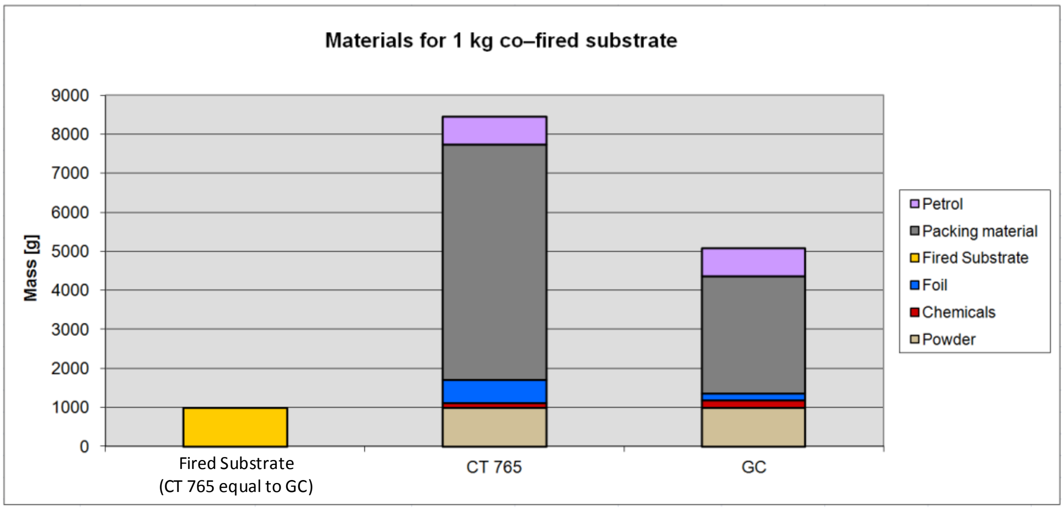
9.6. MIPS – Material Input per kg Co–fired LTCC Substrate

The evaluated material input per kg LTCC substrate includes ceramic powder, ancillary organic chemicals of the green tape, carrier foil, packing material of the sheet rolls, and transport fuel. Table 9.3. shows the results of the mass measurements and calculations of the green and co–fired LTCC samples, as well as standardised to 1 kg co–fired ceramic material. The considered tape materials are HERAEUS CT 765 and CERAMTEC GC.

Table 9.3. Measured and calculated MIPS per kg LTCC substrate within the system boundary.

Material	**HERAEUS CT 765**	**CERAMTEC GC**
	Mass [g]	
Green 6–layer sample unit	4.21	14.89
Co–fired 6–layer sample unit	3.78	12.42
Ceramic powder	1000	1000
Carrier foil	600	162
Organic chemicals	115	199
Fuel	700	700
Packing Material:		
Bubble wrap	2 x 60	60
Plastic wrap	2 x 40	40
Plastic foil for sheet roll protection	2 x 36	36
Carrier roll, carton	2 x 2328	2328
Packaging carton	2 x 550	550
Total	8443	5075

The results of material inputs show that the packing material has the highest mass share, even though only one of at least four packing processes in the life cycle is considered. For both materials the same packing materials are calculated, but the thinner green tape of HERAEUS CT 765 has a longer run length. An alternative carrier roll material made of plastics used by another LTCC substrate manufacturer such as ESL has a mass of 830 g. Figure 9.8. shows the material turnover for both substrate materials for 1 kg and one sample unit.

a) 

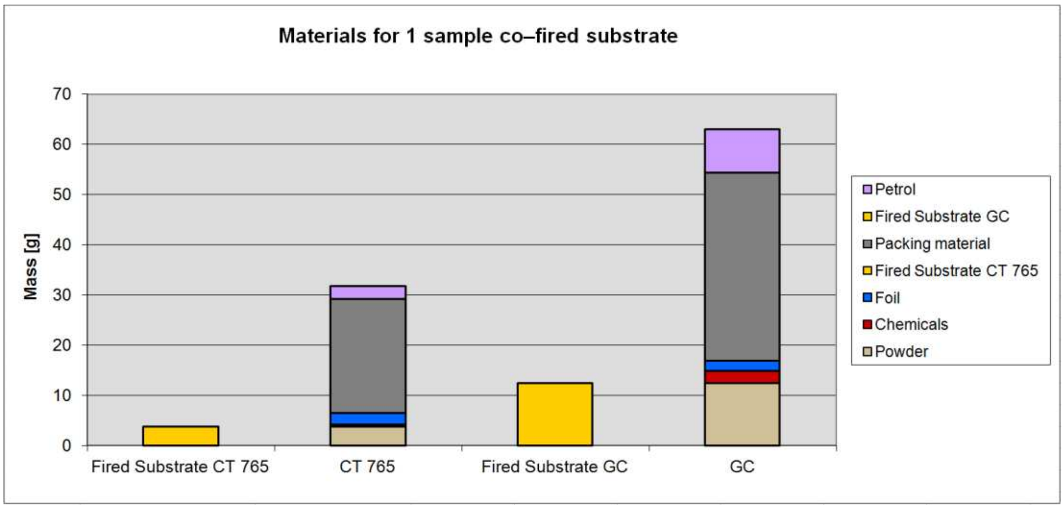
b) 

Figure 9.8. Comparison of the MIPS a) per 1 kg and b) one 6-layer sample unit for two LTCC material types.

These two notations of the results are chosen to visualise the difference between a 1 kg unit and a sample unit. The two sample materials show different relations of the MIPS, depending on whether they are compared as 1 kg unit or as sample unit. The tape with the lower thickness HERAEUS CT 765 needs more packing material, because the green tape carrier foil is relatively thicker and the number of windings on the carrier roll is limited not to destroy the tape. Strictly speaking, the transportation mass should be therefore also higher, which is not considered, because the mass difference of the carrier roll is just one of many other life cycle steps. The result of the MIPS indicates, that smaller and thinner materials or components as well as highly sensitive materials tend to require more packing material.

9.7. Conclusions

The study determined the energy consumption and the global warming potential of three different commercial LTCC substrate materials, 1 kg each, under laboratory small–batch manufacturing conditions. Besides the determination of specific midpoint and aggregated assessment results, one of the key questions was to investigate the influence of the chosen functional unit on general valid statements on potentially environmental critical process parameters within the group of LTCC glass/ceramic components. Therefore, a representative gate–to–gate process chain from green tape delivery to the final co–fired LTCC sample was evaluated and compared in detail on kg–unit and sample–unit basis.

The results of the electrical energy consumption for the gate–to–gate process chain without powder production and transportation are between 127 kWh and 409 kWh per kg LTCC substrate material within the system boundaries. The difference is caused on the one hand by the utilisation of different common co–firing furnace technologies and on the other hand by the different numbers of pieces which can be made from one kg substrate powder due to different tape thicknesses. Therefore, the number of geometry dependent processes vary proportional to the number of pieces. While the energy demand of the laser processing and the co–firing process is dependent on the sample *area*, which is equal for all materials, particularly for a large number of pieces, the normalisation on kg–basis shows different results for the three materials. This raises the question how applicable are life cycle inventory data of databases, which are mainly provided on a kg–basis, on the specific case to be evaluated. Therefore, for any application, it is recommended to calculate different normalisation methods for basic material comparisons, in order to avoid misleading results. For a preliminary assessment of energy hot spots of the process chain, both functional units seem to be sufficient, at least in this case study.

The aggregated assessment method MIPS shows that on a kg–basis the material with smaller layer thickness HERAEUS CT 765 generates 66 % more material turnover than the thicker material CERAMTEC GC. This corresponds to the experience that each kind of miniaturisation, in this case the reduction of the layer thickness, causes a higher mass–specific environmental pressure. The trends of the aggregated method of MIPS correspond to the midpoint indicator global warming potential between the different materials and layer thicknesses, respectively.

The question of the transferability of laboratory processes to industrial manufacturing is answered by the following considerations: The process chain by itself will be similar, but the drying and co–firing processes are expected to be more efficient at least by the factor four and likewise the associated compressor should be more energy efficient. It is further expected that the belt furnace on industrial scale has a lower energy demand than the batch furnace. Therefore, the design of a LTCC device should be carried out for co–firing profiles of the belt furnace. By reducing the energy consumption of the furnaces, the laser, whose energy demand is not expected to significantly decrease, will be brought to the fore.

The calculation of an average value of the electrical energy consumption for the group of LTCC substrate materials for LCI databases is to be considered with the greatest caution. Besides the varieties between different material compositions and tape thicknesses, the individual design of the respective LTCC device can additionally require a number of intermediate co–firing processes and more complex laser structuring.

The gate–to–gate study considered only LTCC substrate materials as of the green tape. According to figure 9.1., the manufacturing process chain for the green tape and in particular the use of ancillary chemicals are not considered due to unavailable data. Further, a functional LTCC device also include a number of metallisations, which are less significant in terms of the mass, but they are made of sophisticated material compositions with high purity and partly including nanoparticles. In the literature there is one reference which indicates that there is a further high environmental impact to be expected (Herrmann et al., 2001). The study did not consider other midpoint environmental indicators such as human- and ecotoxicity which is recommended to work out due to the high share of different toxic ancillary chemicals particularly in the upstream process chain.

9.8. References

Balluch, B.; Lüftl, S.; Seidler, S.; Smetana, W. (2008). "Binder Burnout Studies of LTCC Tapes by Means of Thermal Analyses," Proceedings of the XXXII International Conference of IMAPS Poland Chapter, 2008.

Bienert, C.; Roosen, A. (2010). "Characterization and improvement of LTCC composite materials for application at elevated temperatures," Journal of the European Ceramic Society, Vol. 30, 2010, pp. 369–374.

Chartier, T.; Penarroya, R.; Pagnoux, C.; Baumard, J. F. (1997). "Tape Casting Using UV Curable Binders," Journal of European Ceramic Society, Vol. 17, 1997, pp. 765–771.

Chen, L., Andræ, A. S. G., Zou, G., Liu, J. (2004): Characterization of Substrate Materials for System-in–a–Package Applications. Journal of Electronic Packaging 126 (2): 195-201.

Classen, M.; Althaus, H.–J.; Blaser, S.; Scharnhorst, W.; Tuchschmid, M.; Jungbluth, N.; Faist Emmenegger, M. (2009). "Life Cycle Inventories of Metals," ecoinvent v2.1 report No. 10, Dübendorf, 2009.

Ecoinvent (2019). Ecoinvent database. Swiss Centre for Life Cycle Inventories. St. Gallen, Switzerland. http://www.ecoinvent.org [04/2019].

EU Directive (2009). Directive 2009/125/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for the setting of ecodesign requirements for energy-related products(recast).

Franz, M.; Atassi, I.; Maric, A.; Balluch, B.; Weilguni, M.; Smetana, W.; Kluge, C. P.; Radosavljevic, G. (2012). Material characteristics of the LTCC base material CeramTape GC. IEEEXplore, Proceedings of the IEEE 35th International Spring Seminar on Electronics Technology, pp. 276 - 281.

Frischknecht, R., Tuchschmid, M., Faist–Emmenegger, M., Bauer, C. (2007): Strommix und Stromnetz, Data v2.0 (2007). Ecoinvent report No. 6 / Teil XVI, Uster, December 2007, Swiss Centre for Life Cycle Inventories, Switzerland.

Gontermann, Andreas (2012). Die deutsche Elektroindustrie im globalen Vergleich. ifo Schnelldienst, 18, 2012, p. 22–26.

Herrmann, C.; Gediga, J.; Warburg, N. (2001). Eco-comparison between ceramic and epoxy based populated PWBs. Proceedings of the 2001 IEEE International Symposium on Electronics and the Environment. 2001 IEEE ISEE. pp. 303-308.

Heunisch, A.; Dellert, A.; Roosen, A. (2010). "Effect of powder, binder and process parameters on anisotropic shrinkage in tape cast ceramic products," Journal of the European Ceramic Society, Vol. 30, 2010, pp. 3397–3406.

Hosokawa, N.; Yokoyama, N. (2007). "Nanoparticle Technology Handbook," Elsevier, 2007.

Ibanez–Garcia, N.; Martinez–Cisneros, C. S.; Valdes, F.; Alonso, J. (2008). "Green–tape ceramics. New technological approach for integrating electronics and fluidics in microsystems," Trends in Analytical Chemistry, Vol. 27, No. 1, 2008, pp. 24–33.

Imanaka, Yoshihiko (2010). "Multilayered Low Temperature Cofired Ceramics (LTCC) Technology," Springer Science+Business Media, Inc., 2010.

Jantunen, H.; Hu, T.; Uusimäki, A.; Leppävuori, S. (2004). "Tape casting of ferroelectric, dielectric, piezoelectric and ferromagnetic materials," Journal of the European Ceramic Society, Vol. 24, 2004, pp. 1077–1081.

Jungbluth, N. (2007): Erdöl. Ecoinvent report No. 6–IV, Swiss Centre for Life Cycle Inventories, Duebendorf, Switzerland.

Khoonga, L. E.; Tan, Y. M.; Lama, Y. C. (2010). "Overview on fabrication of three–dimensional structures in multi-layer ceramic substrate," Journal of the European Ceramic Society, Vol. 30, 2010, pp. 1973–1987.

Salam, L. A.; Matthews, R. D.; Robertson, H. (2000a). "Pyrolysis of poly–methyl methacrylate (PMMA) binder in thermoelectric green tapes made by the tape casting method," Journal of the European Ceramic Society, Vol. 20, 2000, pp. 335–345.

Salam, L. A.; Matthews, R. D.; Robertson, H. (2000b). "Pyrolysis of polyvinyl butyral (PVB) binder in thermoelectric green tapes," Journal of the European Ceramic Society, Vol. 20, 2000, pp. 1375–1383.

Schmidt-Bleek, Friedrich (1993). MIPS—A universal ecological measure. Fresenius Environmental Bulletin, 1 (1993), pp. 306-311.

Sea Rates, (2011): Online Freight Exchange, Fast Freight INC USA "West Coast". http://www.searates.com [05/2011].

Sebastian, M. T. (2008). "Dielectric Materials for Wireless Communication," Elsevier, 2008.

Spielmann, M., Bauer, C., Dones, R., Tuchschmid, M. (2007): Transport Services, Data v2.0 (2007). Ecoinvent report No. 14, Villigen and Uster. Swiss Centre for Life Cycle Inventories. St. Gallen, Switzerland.

Tietz, H.–D. (1994). "Technische Keramik. Aufbau, Eigenschaften, Herstellung, Bearbeitung, Prüfung," VDI Verlag, 1994.

Unruh, R. (2008): Transporter Test: Ford Transit FT 350 L TDCI KFZ–Anzeiger 5/2008 – Stünings Medien GmbH, Krefeld: 18–21.

US Patent (2005). "Materials system for low cost, non wire–wound, miniature, multilayer magnetic circuit components", US Patent US6914513, 2005.

Wang, X.; Balluch, B.; Smetana, W.; Stangl, G. (2008). "Optimization of Cavity Fabrication for Micro–Fluidic Systems," Proceedings of the International Spring Seminar on Electronics Technology (ISSE), Budapest, Hungary, ISBN: 978-963-06-4915-5, 2008, pp. 420–425.

Yuping, Z.; Dongliang, J.; Greil, P. (2000) "Tape casting of aqueous Al2O3 slurries," Journal of the European Society, Vol. 20, 2000, pp. 1691–1697.

Zeitzen, F. (2007): Test: Mercedes Actros 1855 LS Megaspace. Lastauto Omnibus, Sonderdruck 2007 / 1: 2–7.

10. Life Cycle Assessment Studies for LED–Lamps: Review and Environmental Considerations

Abstract: *Resource scarcity, hazardous waste and climate change are the driving forces for developing energy efficient and low–toxic lighting sources. Currently, solid state lighting based on light–emitting diodes is expected to become the most dominant lighting technology of the future. Parallel to the ongoing development of light-emitting diode based lighting sources, a number of single case and comparative life cycle assessment studies of LED–lamps and components in varying study settings were carried out. However, these studies mostly refer to specific lamp designs, which limits general conclusions. This work includes a review and comparison of published life cycle assessment studies. In addition, different functional units for LED–lamps and environmental parameters of rare earth oxides are taken into consideration. The study addresses the utilisation of different functional units as well as simplifications, neglections, and uncertainties of LCI data and their consequences. The results show that LED–lamps have a better environmental performance than other efficient lamp technologies only for long life time combined with high luminous efficacy, which does not correspond to the real market and technology development. There are high data uncertainties concerning the manufacturing of LED–packages which significantly influences the total life cycle assessment results and the relations between manufacturing and using phase.*

10.1. Introduction

High-power LED–lamps have been on the market for more than 10 years. Estimates of the economic global market share of LED–lamps in the general lighting sector are 31 % (LEDinside, 2014) and 45 % (McKinsey, 2012) for the period 2015/16, and 67 % to 80 % in 2022 (Zissis & Bertoldi, 2018). However, there are still technical problems concerning the long–term reliability of LED–lamps, such as thermal overload, colour shift, decrease of luminous efficacy, and other material fatigue problems (Chang, 2012; Khanna, 2014; Fulmek et al., 2016; Mehr et al., 2015).

In the European Union (EU) the gradual replacement of incandescent lamps by compact fluorescent lamps (CFL) and by LED–lamps was enforced by the EU household lamp regulation (Comm. Reg., 2009). This regulation is based on the EU Ecodesign Directive (EC Directive, 2009) with the aim to reduce electrical energy consumption by energy efficient lamp technologies.

In literature there are various data on the global electrical energy consumption of lighting. The International Energy Agency estimates that the global electrical energy consumption in 2030 will be 5000 TWh under no energy–efficiency policies and half of that under least life–cycle cost policies (IEA, 2006). A McKinsey study reports that the global lighting market consumes approximately 20 % of the total electric power generated (McKinsey, 2012). The British daily newspaper 'The Guardian' reports that due to legally permitted tolerance values of up to 10 % in the mandatory production information of the lamps, the declared values for luminosity and power consumption may be responsible for wrong consumption rates of up to 25 % (Comm. Reg., 2009; Neslen, 2015).

Based on the energy–saving background, a number of life cycle assessment studies on lighting have been performed by manufacturing companies, consulting companies, and scientific researchers. These studies were carried out with different scopes on objects of comparison, assessment methods, assumptions and data transparency.

This work aims to provide a comprehensive comparison of published LCA studies of LED–lamps and other efficient lamp technologies for general lighting. The key questions are the relations of the total environmental performance between LED–lamps and other efficient lamps, the assumptions on material composition, manufacturing efforts, and local energy consumption. A critical discussions is provided about the technical assumptions, system boundaries in terms of luminous efficacy and lifetime of LED–lamps.

In the frame of the thesis the study addresses the utilisation of different functional units as well as simplifications, neglections, and uncertainties of LCI data and their consequences, respectively. Furthermore, the role of the grey energy and the consequences of incorrect prognosis of future technology and market development reflected in decision making LCA studies are investigated.

10.2. Materials and Methods

The environmental impact of LED–lamps has been assessed by several LCA studies in the last decade. In this review, 13 life cycle assessment studies for LED–lamps and partly other efficient lamp technologies, and a number of further LCA relevant studies are analysed and their assumptions and outcomes are compared and discussed. Since the studies are in most cases not directly comparable, a number of new comparative calculations are performed and visualised. This work is divided into the following topics:

- Functional units for LED–lighting
- Scopes and system boundaries of the reviewed LCA studies
- Life cycle inventories for LED–packages
- Life cycle assessment results for LED–packages
- LICA results for selected rare earth oxides
- Particularities of the LCA studies for LED and other efficient lamps
- Comparison of characterisation category results for LED–lamps
- Comparative results of LCA studies for all efficient lamp technologies

10.3. Functional Units for LED–Lighting

Traditional lighting is characterized in that the lamp and the luminaire are independent systems whose interface is a standardised thread or plug–in systems. Luminaires were often used for decades as a piece of furniture or inventory, while the light source is periodically exchangeable for maintenance. The environmental assessment of lighting was meaningfully always related to the lamp itself as a consumer good and energy consumer. Therefore, the functional unit for lighting service is referred to the lamps with their different illuminances and energy consumptions. In the course of the market launch of LED–lamps, this separation between the lamp and luminaire is no longer clear and presents new challenges in the comparability of the service of lighting.

The definition of the functional unit determines the direction of the considered environmental aspects. The chosen functional unit is an important factor for the comparability of LCIA results of different studies. In the reviewed LCA studies following functional units are applied: (a) *n x (lumen x hour)*, (b) *1 lux for n x hour*, (c) *n x hour* and (d) *1 km–lit–road*, whereby *n* is a variable parameter.

In comparative studies the defined number of hours of the functional unit requires a different number of lamps dependent on the respective lifetime of the technology. The shorter the lifetime of a lamp type the higher the number of lamps for proper comparison. As a result of this imbalance in the number of pieces, the environmental impact of the manufacturing phase gains a greater influence on the LCA result if the life time is shorter. This is to be considered in case of LED–lamps, which have been estimated with a very long lifetime in most studies.

In this context, the product–specific design, which is associated to the functional unit, that is the total mass and its relations between the LED–chip and the remaining part of the lamp/luminaire system, is determinant for LCI data and varies in a wide range within the LED–lamp sector. The following LED–lamp types are available and used in the LCA studies:

- The LED–lamp in the shape of a removable light bulb with standard threads (so called "retrofit" lamp). This lamp type is investigated i.a. in the studies (Navigant, 2009; Osram, 2009; Scholand & Dillon, 2013) (see figure 10.1.a).
- LED–lamp and luminaire are one package whereby the parts of the luminaire serve as heat sink of the LEDs ("dedicated" lamp). The LEDs are not removable and the lamp is a kind of a throw-away product. The boundary between lamp and luminaire is not clearly identifiable (see figure 10.1.b). This lamp type is investigated i.a. in the studies (Abdul Hadi et al., 2013; Navigant, 2009).

- LED–components and their different technologies (light emitting diode and LED-chip, see figure 10.1.c) e.g. (Ecoinvent, 2019; Osram, 2009):
  - Three or more single colored LEDs for additive color mixture, optionally combined with a color conversion layer
  - Blue or UV light combined with a color conversion layer

Figure 10.1. Illustration of different functional units of LED–lamps a) retrofit lamps b) dedicated, one-package LED–lamps c) LED–components.

- Unlike the former examples, a further functional unit can be the lifetime of e.g. 25,000 hours of a specified luminosity. The investigated comparative studies use a combination of lamp type and lumen-hours (e.g. Osram, 2009; Tähkämö, 2015).

The functional units do not consider a different consumer behavior dependent on the lamp type. All studies considered the technical lifetime of the respective lamp technology decoupled from the real lamp operation time and total illumination e.g. per year. However, it has turned out that the number of parallel lamps and the annual operation time are not neutral factors as assumed in the upper functional unit definitions. As discussed in Chapter 7, Section 7.8, in the course of the market introduction of LED–lamps the consumer behaviour has obviously changed into the direction of more illumination causing a significant rebound effect, which is not reflected in current LCA methods.

In comparative studies, the defined functional unit requires that the compared products of different technologies fulfil the same function. In the field of lighting, this is not as simple to define. In addition to the luminous flux measured in lumen and the energy consumption, many other parameters play a role that are difficult to quantify and often need to be collected on a case-by-case basis through extensive measurements and user involvement. This affects the light temperature, the beam angle, the glare behaviour of the LED–lamp, and the parallel new installation of light sensors, which can also lead to changed user behaviour.

Finally, the specified lifetime of a lamp refers only to the bulb. In the case of dedicated LED–lamps, however, the mechanical lifetime of the lamp-luminaire system, that is switches, threads, and brittleness of plastics, plays a role in case of very long lasting lamps and has to be also considered.

10.4. Scopes and System Boundaries of the Reviewed LCA Studies

The published LCA studies have different scopes and system boundaries that do not make them directly comparable. The investigated 13 life cycle assessment studies are analysed and their assumptions and outcomes are compared. Two of the studies are single case studies, including a full life cycle assessment of an indoor LED–downlight luminaire (Tähkämö et al., 2013) and a gate–to–gate study for LED–chip manufacturing (Matthews et al., 2009). One supplementary study includes a detailed process description for colour converter manufacturing (Pradal et al, 2013). All other analysed studies are comparative life cycle assessments, comparing the following lamp technologies in various configurations: (a) retrofit or integrated LED–lamp (Int–LED), (b) dedicated LED–lamp (Ded-LED), (c) incandescent lamp, (d) halogen lamp, (e) compact fluorescent lamp (CFL), (f) linear fluorescent lamp (FL, FLL, or T5), (g) ceramic metal halide lamp (CMH), (h) high–intensity discharge lamp (HID), (i) induction lamp, and (j) high–pressure sodium lamp (HPS). One of the studies does not cover LED–lamps, but it is included because of the poor comparative data situation for linear fluorescent lamps (Welz et al., 2011).

Life cycle inventory data differ in a wide range between the studies. This concerns variations of (a) lamp type and associated application, (b) technical lamp parameters, (c) light propagation and intensity, (d) level of details of the LED–package, (e) considered electrical energy mixes, and (f) waste management. Further differences arise from different results of the life cycle impact assessments, which are dependent on (g) the chosen functional unit, (h) the assessment method, (i) the considered environmental impact categories, (j) the used data sources, and (k) the nature and details of the results.

The LCA studies considered 39 different midpoint impact categories in various combinations. The aggregated characterisation method Eco–indicator 99 was applied in four studies (Abdul Hadi et al., 2013; Sangwan et al., 2014; Tähkämö et al., 2015; Welz et al., 2011). Furthermore, the studies applied 20 different energy mixes for the electrical energy consumption and almost all studies distinguish between manufacturing country and consumer country and/or calculate different energy source scenarios.

The intended application of the assessed LED–lamps is in most cases indoor or not specified. LED–street lamps are investigated in four studies (Abdul Hadi et al., 2013; Dale et al., 2011; Hartley et al., 2009; Tähkämö & Halonen, 2015). One study investigated a specific LED–downlight luminaire (Tähkämö et al., 2013), another one a specific working place lamp setting (Principi et al., 2014).

Figure 10.2. shows the distribution of the assumed luminous efficacies and the respective lifetime of efficient lamps for all studies. The lifetime of LED–lamps is in the range from 15000 h to 59000 h.

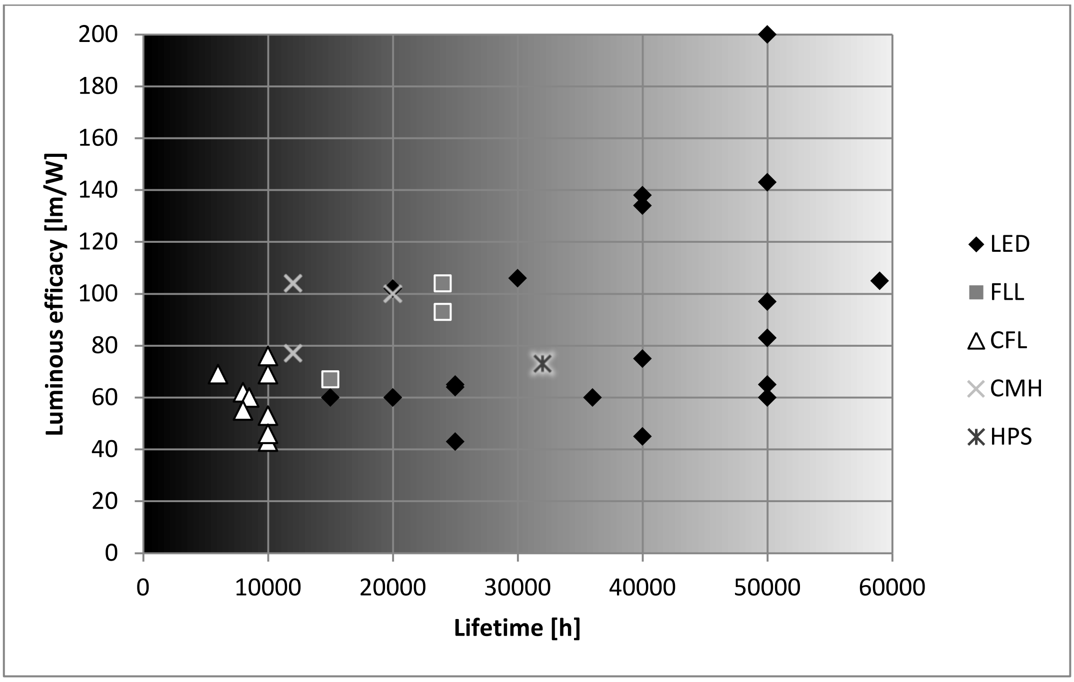


Figure 10.2. Distribution of the assumed lamp parameters throughout the LCA studies.

The assessed LED–lamps have largely varying technical parameters. The considered luminous efficacies, which are directly connected with the environmental impact of the using phase, are in the range from 43 lm/W to 200 lm/W. Whereby, the values of 134 lm/W is forecasted for 2017 (Navigant, 2012) and of 200 lm/W for 2020 (Tähkämö et al., 2015). LED–lamps show the largest variance of their assumed technical parameters compared to the other efficient lamps. In 2016, the best practice available luminous efficacy with sufficient lifetime was about 100 lm/W for neutral white colour (U.S. DOE, 2016). In 2019, a short market review shows that the highest available luminous efficacy for warm white LED–lamps is in the range of 115 lm/W at a lifetime of 15,000 h made in filament technology. This shows that the respective luminous efficacy data in the reviewed studies seem to be above current realistic values, as are the life time assumptions.

10.5. Life Cycle Inventories for LED–Packages

The LED–package life cycle inventory data most commonly used are provided by the ecoinvent database (Ecoinvent, 2019). Six of the LCA studies applied this dataset on a mass basis directly or in a modified form. New LCI data were determined by four studies (Matthews et al., 2009; Navigant, 2012; Osram, 2009; Pradal et al., 2013), whereby the last reference did not consider wafer manufacturing. Two studies applied cost based LCI data (Dale et al., 2011; Hartley et al., 2009).

The ecoinvent database provides life cycle inventory data for the component "*light emitting diode, LED, at plant*" (Hischier et al., 2007) based on a conventional 5–mm LED for through–hole mounting with a mass of 0.35 g and a luminous flux of 4 lm. The dataset is identical to "*Diode, glass-, for through-hole mounting, at plant*" of the ecoinvent database. Figure 10.3. shows the material composition of the ecoinvent LED–dataset and the referring production efforts according to the dataset "*production efforts, diodes*".

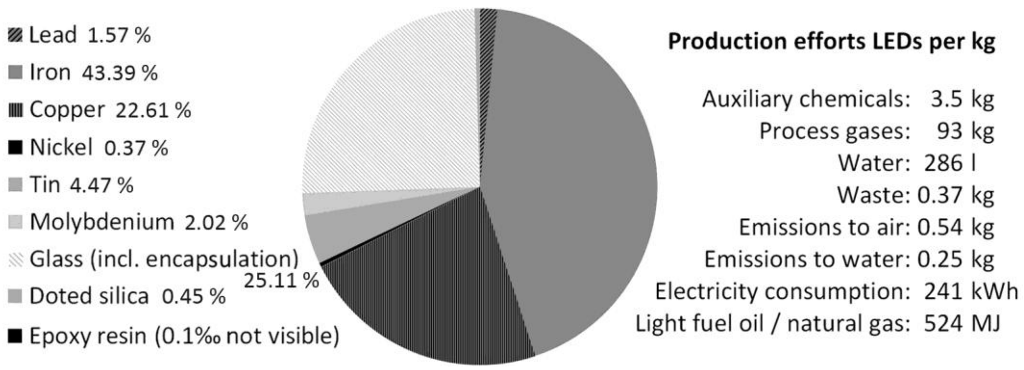


Figure 10.3. Life cycle inventory data of LEDs in % and production efforts, according to the ecoinvent dataset "*light emitting diode, LED, at plant*". (Own work; data source: Ecoinvent, 2019; Hischier et al., 2007)

The material "*doted silica*" is defined by the dataset "*silicon, electronic grade, at plant*", which is reported in the Photovoltaic Report of the ecoinvent database (Jungbluth, 2009). This dataset does not include wafer fabrication. The LED material composition of the ecoinvent database differs strongly from modern white light LEDs with blue light and phosphor layers. References for currently used LED–chip materials can be found in Chapter 7, Sections 7.3.–7.5. as well as in (Franz & Wenzl, 2017), respectively. Unfortunately, there are no detailed mass data for modern LED–chips according to these references available to be directly compared with figure 10.3. The application of the ecoinvent LED–dataset is therefore problematic.

The study by Matthews et al. (2009) provides new life cycle inventory data for the manufacturing of 1 mm^2 high brightness LED–packages in terms of the final electrical energy consumption from wafer growing to bonding and packaging. The functional unit was a 300 mm silicon wafer with 50 % yield rate, which results in 1000 pieces of LEDs per single wafer. The authors state that there is a lack of data for the colour conversion layer and it was assumed that the energy consumption is very low. The final electrical energy consumption results are 0.02 kWh/LED on a laboratory scale and 0.07 kWh/LED on a production scale. It is notable that the electrical energy consumption of the production scale according to this study is more than three times higher than of the laboratory scale.

The study by the U.S. Department of Energy (Navigant, 2012) Part 2 provides new manufacturing data for the LED–package. The assumed LED–technology was GaN on sapphire substrate with a remote YAG:$Ce3^{+}$–colour converter. More than 75 processing steps were considered from wafer and semiconductor fabrication to LED–packaging. All data relate to one 3 inch wafer. The wafer fabrication losses were assumed to be 63 % of the raw material consumption. The number of LED–chips was assumed with 2438 units per wafer (forecasted: 3250 units in 2017). The (end–)energy consumption for the wafer and the LED–die fabrication is 60.87 kWh/wafer. Herein, the wafer fabrication has a share of 30 %. Including the assembly efforts, one single LED–package requires 0.055 kWh electrical energy, which compares well with Matthews et al. (2009).

Tähkämö et al. (2013) conducted a life cycle assessment study for a LED–downlight luminaire with a remote colour converter. The authors state that the ecoinvent LED–data do not comply with the state of the art and considered that the data are highly underrated. Therefore, the life cycle inventory data for the LEDs were multiplied with a correction factor of 5.

10.6. Life Cycle Assessment Results for LED–Packages

The primary energy demand for LED–package manufacturing was evaluated and compiled by the U.S. Department of Energy (Navigant, 2012; Part 1). Figure 10.4. shows these results extended by further results of the same author in Part 2 per LED–package, and the Ecoinvent database per gram.

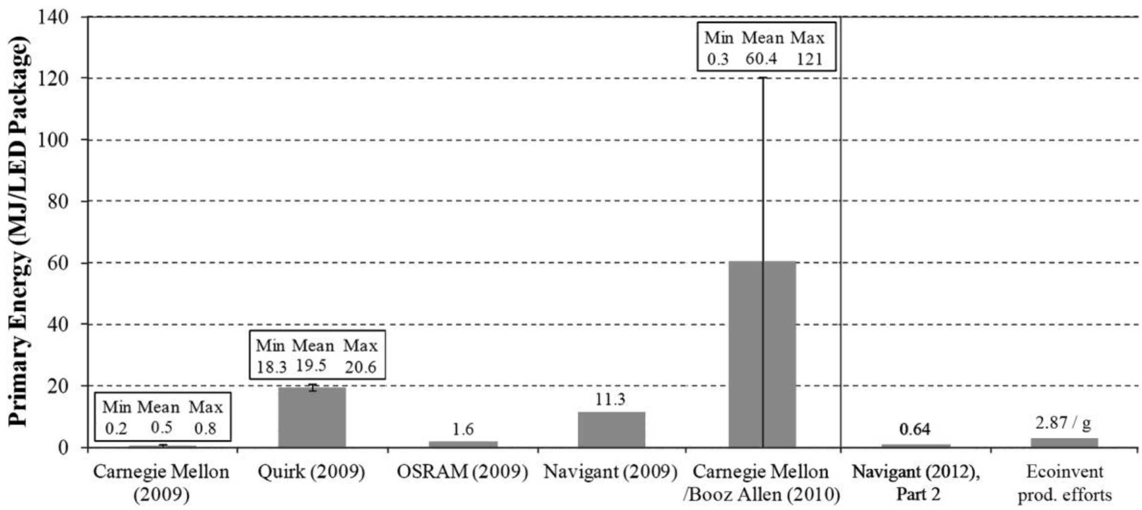


Figure 10.4. Primary energy demand of the manufacturing of one LED–chip/package (Navigant, 2012; Part 1), extended by Navigant (2012; Part 2) and Hischier et al., (2007).

The results show that there are very high uncertainties concerning the manufacturing efforts of one LED–package in the range between 0.2 to 121 MJ, that is an uncertainty factor of 600. Moreover, it can be assumed that non of the studies considered the manufacturing of nanoparticles and the complex structure of the colour converter layer. The environmental impact of the LED–chip is a key part of the environmental performance of the LED–lamp and the available LCI data are insufficient in a wide range. The published gate–to–gate energy data for LED–chip manufacturing cannot replace a high aggregated cradle–to–grave life cycle assessment. Furthermore, the studies calculated the LED–packages differently, on the one hand as piece unit and on the other hand in grams or mm^2 LED–dice, which makes a comparison and generalisation difficult.

The high uncertainty of the LED–package LCI data have a large influence on the total environmental assessment results and comparative studies as can be seen from figures 10.5. to 10.7. Almost all studies applied relatively low LCI data and it is expected that the manufacturing impact is significantly higher. There is no estimation possible and it is recommended to newly evaluate LCI data for LED–packages.

10.7. LICA Results for Selected Rare Earth Oxides

The semiconductor and phosphor layer of the LED–component can be produced with different technologies and includes various rare earth materials. Therefore, there are different properties, efficiencies and reliabilities of LED–lamps on the market. The respective material composition is not distinguishable for the consumer. Table 10.1. gives an overview on selected rare earths incorporated in LEDs according to Chapter 7, Section 7.5 and available results of life cycle inventory data and midpoint assessment results of their oxides. The investigations were made for rare earth elements produced in Bayan Obo, China using price–based and mass–based allocation (Koltun & Tharumarajah, 2014).

Table 10.1. LCI and environmental impact for 1 kg rare earth oxide raw material production using price–based allocation (Data source: Koltun & Tharumarajah, 2014).

Rare earth oxide	Energy consumption [MJ]		Water consump-tion [10^3 l]	Env. Impact	
	Electr.	Heat		GHG em. kg CO_2e	Resource depletion MJ surplus
CeO_2	24	32	12	10	12
Eu_2O_3	3472	5687	2080	1622	1884
Gd_2O_3	121	198	73	57	66
Lu_2O_3	N/A	N/A	N/A	N/A	N/A
Y_2O_3	81	132	49	38	44

Table 10.2. LCI and environmental impact for 1 kg rare earth oxide raw material production using mass–based allocation (Data source: Koltun & Tharumarajah, 2014).

Rare earth oxide	Energy consumption [MJ]		Water consumption [10^3 l]	Env. Impact	
	Electr.	Heat		GHG em. kg CO_2e	Resource depletion MJ surplus
Light REOs	74.08	102.51	38.06	32.29	36.50
Medium REOs	66.68	102.51	37.64	30.29	34.99
Heavy REOs	76.98	102.51	37.64	34.49	39.27

The classification of light, medium, and heavy rare earth oxides (REOs) relates to their atomic numbers and it is not standardised. The results show that on price–based allocation the europium oxide has by far the highest environmental impact in all characterisation categories of all other rare earth oxides. Using mass–based allocation, all rare earth oxide group results are in the same range in the respective impact categories. Only yttrium oxide shows similar results independent of the allocation method, the other oxides have significant lower values on mass–based level. The results of table 10.2. are too undifferentiated to further compare the rare earth oxides.

The study further determined, that the mass–based GHG emissions for the reduction from REO to rare earth elements (REE) is 5.64 kg CO_2e /kg_{REE} for light REOs and 5.04 kg CO_2e /kg_{REE} for medium and heavy REOs (Koltun & Tharumarajah, 2014).

In comparison to the upper assessment results, the following calculations show the environmental impact of rare earth oxides as included in the ecoinvent database 3.4 (Ecoinvent, 2019). The available processes concerning LED–lamps are "*rare earth oxides production from bastnäsite concentrate | cerium concentrate, 60% cerium oxide*" and "*rare earth oxides production from bastnäsite concentrate | samarium europium gadolinium concentrate, 94% rare earth oxide*".

Table 10.3. shows the data background of these two processes according to Althaus et al. The cerium concentrate consists of min. 60 % CeO_2, max. 20 % La_2O_3 and others. The Sm-Eu-Gd concentrate consists of min. 50 % Sm_2O_3, 8.5 % Eu_2O_3, 14 % Gd_2O_3, and others. Background considerations of the life cycle inventory data are that due to the material composition of bastnasite, all oxides are simultaneously produced independent of the market demand. Therefore, allocation factors for all LCI data are determined based on the oxides with high market demand pressure, which is neodymium oxide or europium oxide. (Althaus et al., 2007)

Table 10.3. Mass ratio and applied market demand allocation factor of rare earth oxides of 1 kg rare earth concentrate of bastnasite based on the ecoinvent database. (Data source: Ecoinvent, 2019; Database version 3.4.; Althaus et al., 2007)

Rare earth oxides	Amount	Mass ratio	Allocation factor
Cerium concentrate, 60 % cerium oxide	0.493 kg	50 %	22.1 %
Lanthanum oxide	0.282 kg	28 %	14.7 %
Neodymium oxide	0.154 kg	15 %	40.9 %
Praseodymium oxide	0.051 kg	5 %	14.6 %
Samarium europium gadolinium concentrate, 94 % rare earth oxide	0.020 kg	2 %	7.8 %
Total	1.000 kg	100 %	100 %

Table 10.4. shows the determined life cycle assessment results for the cumulative energy demand (CED) and the midpoint characterisation results (ReCiPe Midpoint (H)–method) of the climate change and water depletion for cerium concentrate and samarium–europium–gadolinium concentrate. The environmental impacts of the newly calculated pure oxides marked with *) are mass–based allocated to a share of the cerium concentrate of 75 % cerium oxide and 25 % lanthanium oxide. In case of Sm-Eu-Gd concentrate, the europium oxide is mass–based allocated with a share of 11.76 %. Cerium oxide and europium oxide are then normalised to 1 kg.

Table 10.4. Environmental impact results for 1 kg rare earth concentrate based on the ecoinvent database (Data source: Ecoinvent, 2019; Althaus et al., 2007).

Rare earth oxides [1 kg]	CED [MJ]	Water depletion [m^3]	Climate change [kg CO_2e]
Cerium concentrate (60%)	125	16.58	5.83
Cerium oxide *)	208	27.63	9.72
Sm-Eu-Gd concentrate (94 %)	383	50.72	17.83
Europium oxide *)	18024	2387	839

*) The respective allocated mass share of Ce oxide and Eu oxide of the concentrates is normalised to 1 kg

Compared to table 10.1., the results of table 10.4 show for cerium oxide equal results in the category climate change. The water consumption is 2.3 times higher and the CED is 3.7 times higher than the energy consumption of table 10.1. In the case of europium oxide, CED and climate change are twice as much and the water depletion is 47 % higher. It is not clear to what extent the LCI data and allocations overlap since table 10.1. is determined on basis of the SimaPro database (SimaPro, 2019) without further details. For both tables applies that from the view of rare earth manufacturing for LEDs, the production of europium is an environmental hot spot. Nevertheless, it is recommended to investigate in detail all other utilised rare earths for LEDs according to Chapter 7, Section 7.5 as well as (Franz & Wenzl, 2017).

The cumulative energy demand is based on the primary energy consumption, which is dependent on the final energy consumption, the energy transformation and transport, and the applied energy source or energy mix. It is not clear what type of energy, primary energy or final energy, is calculated in table 10.1. In any case it is recommended to strictly separate and indicate the type of energy and to provide results of the final energy consumption to make the results more usable for further research.

The allocations of the ecoinvent rare earth datasets show that one should not apply LCI data offhand as true process data for other processes. For example, in the case of cerium oxide the LCI data would be much too low and in the case of neodymium oxide much too high according to table 10.3. These discrepancies can only be recognised by studying the background documentation, in this case available by (Althaus et al., 2007), and they are not included in the standard dataset documentations.

Since mining of rare earths is associated with high environmental damage in terms of ecotoxicity, as stated in Chapter 7, Section 5, the energy consumption and the midpoint environmental categories climate change, water use, and resource depletion are insufficient to make conclusive environmentally relevant statements. Recycling technologies and circular economy of rare earths is still an unsolved issue and it is a great challenge to find technical solutions to separate smallest amounts of dopants and nano–particles of rare earths that are widely spread in electronic equipment.

10.8. Particularities of the LCA Studies for LED–Lamps and other Efficient Lamps

The following comments emphasise unique assumptions, results, and statements of the studies to be considered when interpreting the LCA results.

The work published by Osram calculated the primary energy consumption for manufacturing of one LED–lamp to 9.9 kWh. The fabrication of the LED–packages has a share of 30 % of the lamp production. The LED–chip was fabricated in Germany (frontend) and Malaysia (backend) and the final production of the LED–lamp was in China. It was assumed that the production of 1 kWh of final electrical energy requires 3.3 kWh primary energy. (Osram, 2009)

The study by Quirk states the problem, that countries with increasing energy efficiency or renewable electrical energy production, tend to outsource the manufacturing emissions to foreign countries with carbon intensive electrical energy mixes and there is no research about the imported embedded energy of products. This fact leads to distortion of the domestic emission inventories of up to 20 %. Furthermore, the primary energy demand of LED–lamp manufacturing is 65 kWh (± 30 %) per bulb, with about 50 % associated to the printed circuit board and the other half to the LED–chip. The variances result from uncertain energy data for printed circuit board and LED–chip production. The production of other lamp parts needs less than 2 % energy each. (Quirk, 2009)

Welz et al. conducted a comparative life cycle assessment study for four different lighting technologies. Although no LED–lamp was assessed, the study results are relevant for this review due to the fact that there are environmental impact results of linear fluorescent lamps available. They are rarely compared in comparative life cycle assessment studies on LED–lamps despite of their good environmental performance. The external electronic ballast was included, but the luminaire was not considered (cf. Navigant, 2009). The results of the study show, that the manufacturing impact of the linear fluorescent lamp has a share of 8 % (Swiss energy mix) and 2 % (European UCTE energy mix) of the total life cycle impact. (Welz et al., 2011)

The study further investigated the effects of different lamp qualities on the environmental impact of compact fluorescent lamps. The electronic part of the lamps included a different number of components and therefore, the environmental impact was not equal. Nevertheless, compared to the full life cycle including the use phase, these variations were not significant.

The study by Abdul Hadi et al. compared two lamps with a lifetime dependent luminous flux of 25000 lm – 17500 lm (CMH–lamp) and 19000 lm – 15000 lm (LED–lamp). These two lamps were considered as equal. The mean luminous flux of the LED–lamp with 17000 lm is 15 % lower than of the CMH lamp with 20000 lm (GE Lighting, 2011). The study defines that the LED–lamp matches the minimum luminous flux of the CMH lamp at the end of the life time. Increasing the calculation values of the LED–lamp by 15 % to an equal mean level of the CMH lamp, the eco–indicator points are almost equal. (Abdul Hadi et al., 2013)

The study by Tähkämö et al. provides detailed manufacturing results of a LED–downlight luminaire for 16 environmental impact categories. The lifetime of one LED–lamp was modelled with different numbers of hours, which causes significant influences on the full life cycle impact. The results for the global warming potential show that the manufacturing impact share for a lifetime of 50,000 h is 16 %, for 36,000 h it is 27 %, and in case of a lifetime of 15,000 h the impact share is 40 % (Tähkämö et al., 2013). The last scenario is currently state of the market (see Chapter 7, Section 7.7.2.).

The work by Tähkämö et al. compared streetlamps with three different functional units: per luminaire, per lm x h, and per km–lit–road. The application of which have significant influence on the environmental impact classification of the lamp technologies. The technological standards for street lighting require a pole spacing of 50 m for HPS lamps and 36 m for LED–lamps, because the LED–lamps have a smaller illumination angle. (Tähkämö et al., 2015). This results in an increased amount of installed LED–lamps by 39 %. The study did not consider the poles as they were assumed to be equal for all lamp technologies and therefore neglectable according to the LCA rules. In the case of the km–lit–road scenario the results for the LED–lamps have to be adapted by a plus of 39 % for pole manufacturing due to omissions of the study, as a result of own recalculations. A further interesting result is that the cover of the LED–luminaire shows a manufacturing impact share of more than 90 %. Depending on the impact category, only 1 % to 6 % is assigned to the LED–package production. That means, the increased number of lighting points per km for LED–lamps is a significant environmental impact factor.

10.9. Comparison of Characterisation Category Results for LED–Lamps

This section analyses the bandwidth of environmental impact characterisation results of the investigated LCA studies concerning only LED–lamps. The global warming potential GWP was investigated by most of the LCA studies. Therefore, it is the most representative comparison of all study results. The manufacturing share is between 2 % and 82 %. Basically, the differences depend on the following variable parameters: The *luminous flux* and the *lifetime* influence the number of lamps to be produced, that is proportional to the share of the manufacturing impact. The *luminous efficacy* and the *electrical energy mix* affect the share of the using phase impacts. The LED–packaging fabrication share of the manufacturing phase is in four cases 50 % to 60 % (Quirk, 2009; Navigant, 2009), in one case 15 % (Hartley et al., 2009), and in one further case 2 % (Tähkämö et al., 2015). The manufacturing influence of the lifetime is illustrated by the comparison of 15000 h, 30000 h, and 50000 h. The manufacturing share increases from 15 % to 42 % with decreasing lifetime (Tähkämö et al., 2013). The influence of the selected energy mix is reflected in the comparison of the EU mix with hydropower based electrical energy, where the using phase share decreases from 88 % to 18 % (Tähkämö et al., 2015). Another electrical energy scenario shows that the end–of–life battery treatment of remote photovoltaic systems significantly increases the using phase impact compared to natural gas electrical energy generation (Abdul Hadi et al., 2013). Other variations of the impact share seem to depend on different lamp modelling, which is not only explainable by physical lamp parameters and electrical energy mixes (e.g. Osram, 2009; Quirk, 2009).

The relations between manufacturing and use phase highly vary between the studies. The lamp manufacturing has a low influence on the global warming potential compared to the other characterisation categories. On the other hand, the (eco)toxicity potential, the photochemical oxidation potential, and the eutrophication potential are significantly influenced by manufacturing. Outstanding manufacturing impact results with a share of

82 % and 96 % in the category human toxicity potential shows the only study applying the input/output (I/O) database for the life cycle inventory. The I/O method approximates the use of materials and energy consumption based on economic data in terms of the producer price of a product. (Hartley et al., 2009).

The only aggregated assessment method Ecoindicator´99 is applied by three LCA studies for LED–lamps. The manufacturing impact share is between 10 % and 48 % (see figure 10.7). These results correlate with the environmental damage of the electrical energy generation in the following order: hydropower, Swiss mix, natural gas (Abdul Hadi et al. 2013), EU mix, and India mix. The end–of–life impact share amounts to 1.8 % in case of hydropower and 0.6 % in case of the India mix, which does not correlate to the other relations.

Figures 10.5.–10.7. show the comparative study results of the global warming potential (GWP), the human toxicity potential (HTP), and the eco–indicator 99 (EI`99) regarding all life cycle stages. A summary of the applied electrical energy mixes and all other assumptions and system boundaries of the respective studies are provided in (Franz & Wenzl, 2017). The manufacturing phase is represented by the sum of dark grey and black coloured bars. If there is no black section separately visualised, the LED–chip manufacturing is included in the dark grey bar in case of those studies, where it is not explicitly distinguished.

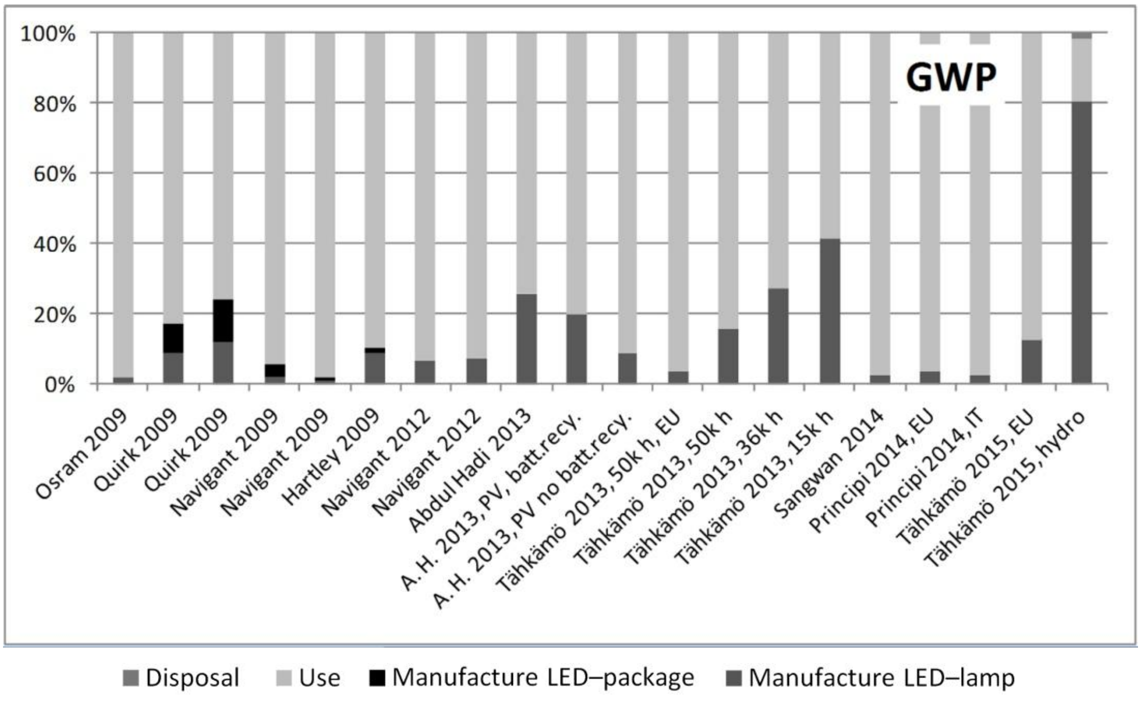


Figure 10.5. Comparison of the global warming potential (GWP) study results (own work).

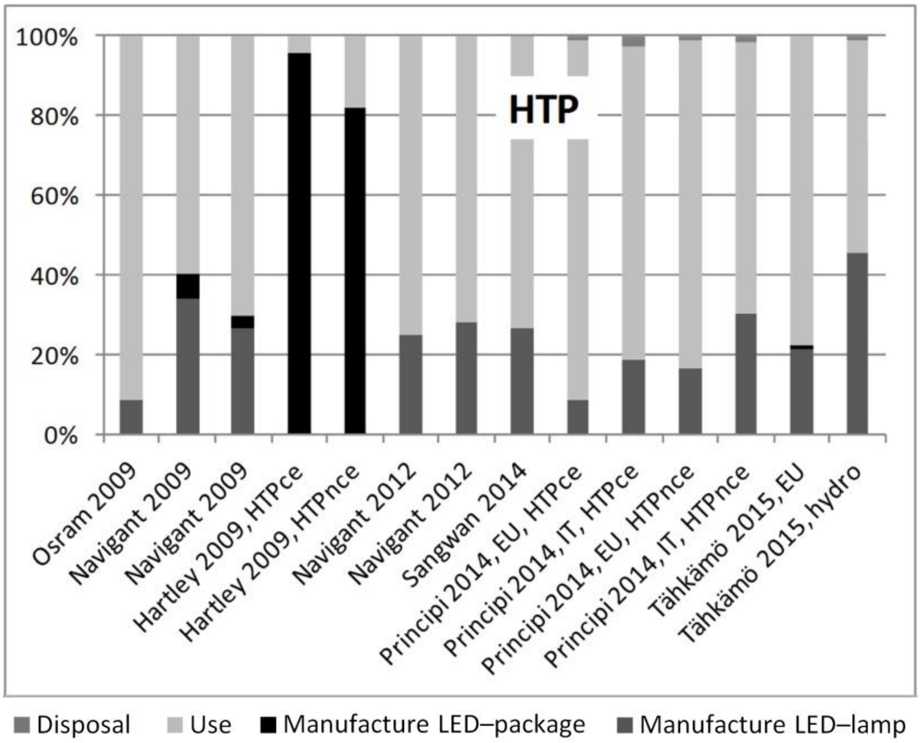


Figure 10.6. Comparison of the human toxicity potential (HTP) study results (own work).

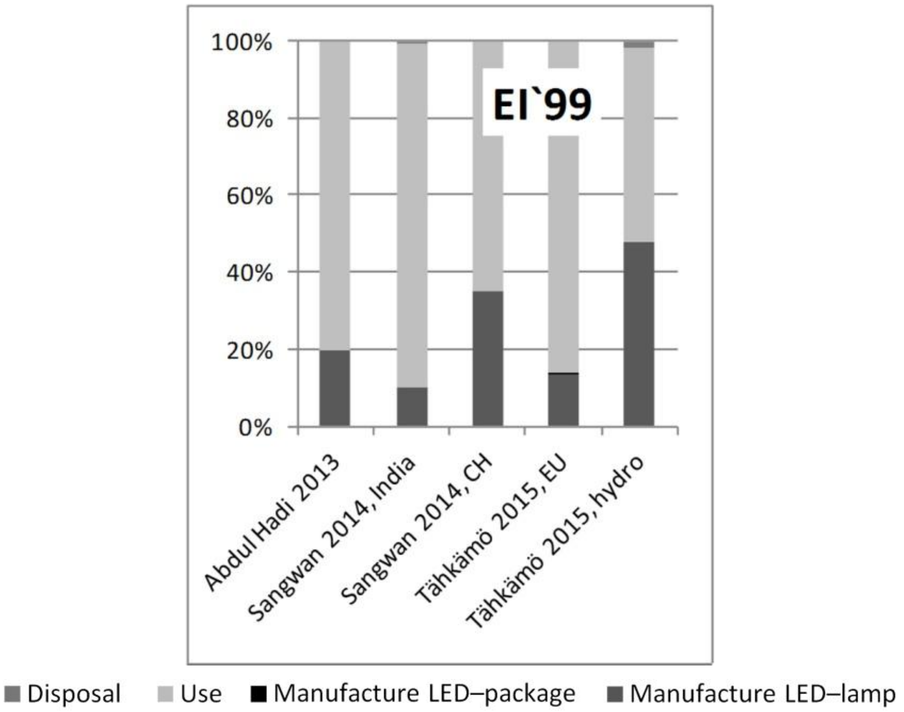


Figure 10.7. Comparison of the eco–indicator 99 damage points (EI`99) study results (own work).

10.10. Comparative Results of Selected LCA Studies for Efficient Lamp Technologies

The comparison of the LCA studies for LED–lamps and other efficient lamp technologies in their overall picture is relatively difficult, since their assumptions, system boundaries, and provided LCI and impact assessment data are very different. Two studies are found which have compared all LCIA results for a wide range of lamp technologies and number of environmental impact characterisation factors. One study of a manufacturer provides only the executive summary with no detailed data (Osram, 2009). In other studies the assumptions and data are published in detail (e.g. Navigant, 2012; Tähkämö & Halonen, 2015).

Two midpoint LCA studies were performed comparing 15 characterisation factors of LED–lamps with other efficient lamp technologies (Navigant, 2009; Navigant, 2012). Figure 10.8. shows newly compiled cradle–to–grave LCA study results which are based on different lamp technologies as well as different assumptions for the luminous efficacy and lifetime. The reference of 100 % is for all studies an incandescent lamp which makes the results comparable. The figure shows an excerpt from 0 % to 50 % for all impact categories.

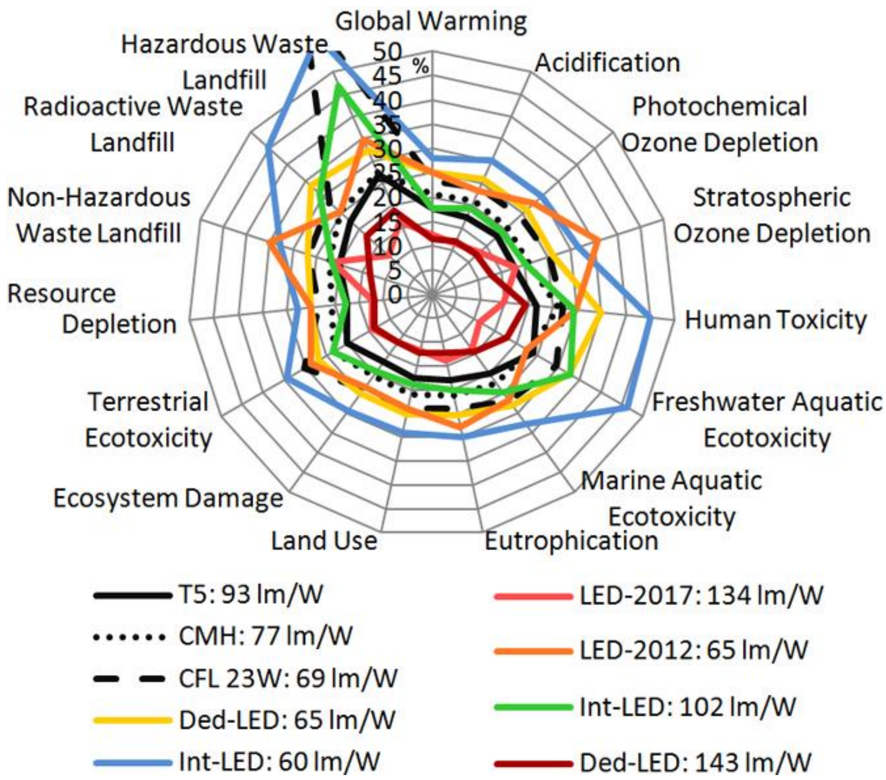


Figure 10.8. Compilation of full life cycle LCA results of LED–lamps compared with fluorescent lamp T5 / ceramic metal halide lamp CMH / compact fluorescent lamps CFL. The relative impact values are compared to an incandescent lamp with 100 % of all impact categories. (Own work; Data source: Navigant 2009/2012)

The results of figure 10.8. show that LED–lamps with a luminous efficacy > 134 lm/W have the lowest environmental impact in all categories (red and dark red lines), but this assumed value is not state of the market. Considering real market offers, long fluorescent lamps (black line) have the best environmental performance followed by CMH lamps (black dotted line) together with LED–lamps with a luminous efficacy > 102 lm/W (green line). Compact fluorescent lamps have a significantly poorer environmental performance (black dashed line), but in this case this could be caused by the relatively low luminous efficacy.

Compared with an incandescent lamp which is normalised to 100 %, representing de facto the energy in the using phase, almost all other lamps show within their impact relations a relatively high environmental pressure in the direction of waste management and human and freshwater toxicity.

The total environmental performance of T5 fluorescent lamps and CMH lamps compared with LED–lamps is equal or better regarding the state–of–the–art of 2019. The assumption of the studies include expected future technology progress that is not yet achieved for luminous efficacies > 115 lm/W and a lifetime > 20000 h. Table 10.5. shows the applied data for the luminous efficacy and the lifetime of the study results in figure 10.8. The values of the years 2014 and 2017 are estimated and in the future expected luminous efficacies and lifetimes. The respective reference is added in blue brackets. A comparison with available lamps on the market is shown in Chapter 7, Section 7.7.2.

Table 10.5. Luminous efficacy and lifetime of different lamp types according to figure 10.8.

Assumed luminous efficacy in LCA studies [lm/W]							
Int–LED (1)		Ded–LED (1)		T5–Lamp (1)		LED (2)	
2009	2014	2009	2014	2009	2014	2012	2017
60	102	65	143	93	102	65	134
20000 h		50000 h		24000 h		25000 h	40000 h

LED–lamps with a forecasted luminous efficacy > 134 lm/W and a lifetime > 40000 h have the best environmental performance for all impact categories, because the manufacturing impact, which is consistently higher for LED–lamps, then has less significance. Considering commercially available lamps, the T5 long fluorescent lamp shows better or equal impact values than all other technologies. Compared to linear fluorescent lamps, LED–lamps ≤ 102 lm/W show a weaker performance for almost all impact categories (Navigant, 2009; Sangwan et al., 2014).

High pressure sodium (HPS) lamps show in three cases a weaker performance for all impact categories than LED–lamps (Hartley et al., 2009; Dale et al., 2011, Tähkämö et al., 2015). One scenario with a functional unit of 1 km–lit–road shows an equal environmental performance, because a larger number of LED–lamps is necessary to achieve equal illuminance (Tähkämö et al., 2015).

The comparison with ceramic metal halide (CMH) lamps shows a varied picture. One study shows different results throughout the impact categories for a LED–lamp with 102 lm/W: two categories show better results (GWP, ARD), four categories show worse results (HTP, FAETP, RWL, HWL), all other results are equal (Navigant, 2009). Another study shows better results for EI99, equal results for energy/CO_2, and worse results for water consumption (Abdul Hadi et al., 2013). Concerning MH lamps, two studies show a better environmental performance for LED–lamps for all impact categories (Hartley et al., 2009; Dale et al., 2011).

The applied electrical energy mix for the manufacturing and the using phase has a large influence on the final impact results and the relations between the impact categories. The electrical energy from coal has a higher global warming potential by 58 % than the US average mix and the coal ecotoxicity impact is 60 times higher. Concerning the respiratory effects, the US average mix impact is three times higher than wind power and the coal impact is 20 times higher than the US average (Hartley et al., 2009). A further study found that the EU UCTE mix has a 3.8 times higher impact than the CH mix for EI99 (Welz et al., 2011). The Indian electrical energy mix shows a 5 times higher EI99 impact than the Swiss electrical energy mix (Sangwan et al., 2014). There is no comparison between the US and the EU electrical energy mixes.

The manufacturing phase of lamps shows different impact relations to the full life cycle. LED–lamp manufacturing tends towards a significant weaker environmental performance than the full life cycle for all characterisation categories.

10.11. Discussion of the Comparative Results

The published life cycle assessments of LED–lamps vary between comparative and single case studies with very different theoretical and measured data, assumptions, simplifications, applied assessment methods, and data sources. A small number of comparative studies include measurements for the particular lighting situation which shows, that by solely equating the luminous efficacies the results do not necessarily represent an equal illumination condition.

The comparative study results show that the using phase for LED–lamps has the lowest environmental impact of all lamp technologies for a luminous efficacy > 104 lm/W without considering the lighting situation. In terms of the manufacturing phase, linear fluorescent lamps have the lowest and LED–lamps have the highest environmental impact of efficient lamp technologies. The manufacturing impacts of LED–lamps vary between 1 % and 96 % of the total life cycle, dependent on the luminous efficacy, electrical energy mix, lifetime, and considered impact category. In detail, the manufacturing impact of GWP is between 1.5 % and 41 % (excluding hydropower energy mix), the eutrophication potential is between 3 % and 72 %, and the human toxicity is between 8 % and 96 %. More details can be found in (Franz & Wenzl, 2017). The determined impacts of transport and packaging are < 1 % each. The environmental impact of the waste disposal ranges from negligible to 27 %, or even as raw materials or energy benefit due to the recovery of the materials.

Compared to incandescent lamps, LED–lamps are far superior with respect to the environmental impact of the use phase electrical energy consumption. Compared to other efficient lighting technologies it is not possible to give a clear recommendation about the best environmental performance which have, in some cases of LED–lamps, proven to be too optimistic.

The specific application of a LED–lamp defines the environmental performance of the respective lamp. LED–retrofit lamps, dedicated floor lamps, ceiling lamps, and street lamps have different electrical (e.g. lightning, dimming), mechanical and thermal requirements. Only the LED–chip without colour converter of the lamps is likely to be equal, all other parts have to meet different requirements and therefore different materials are used.

Not all life cycle inventory data differentiate with respect to the quality of materials, components, and lamp parts, which has effects on manufacturing details and the lifetime of the product. Particularly LED–lamps require thermally well designed components and high quality LED–chips to meet the required and assumed properties of the respective life cycle assessment. There is no information about the global ratio of high quality LED–lamps with state–of–the–art luminous efficacy, colour rendering, lifetime etc., and the average LED–package market. One study found that in case of the investigated compact fluorescent lamps, there is a difference of the manufacturing impacts by 300 %, depending on the manufacturer (Welz et al., 2011).

The final luminous efficacy of the LED–lamp is the product of the LED–chip efficacy and the efficiencies of the optical lens, the driver, and the thermal efficiency. Particularly for dedicated LED–lamps with the technical potential of better values, it is not possible to define a universal value, due to the lack of a production quality standard. It can be assumed that the international mass market is not oriented on the best manufacturing practice. This fact reduces the global mean value of the efficiency.

Some comparative studies exclude lamp parts, materials and processes, if they are equal for all compared lamp types. This becomes problematic, if the results are compared to the using phase, and if the results are compared with other studies. Life cycle inventory data are often taken from other studies. This can lead to unnoticed chains of errors for the particular assessment situations.

Cleaner electrical energy production and higher lamp efficiency lead to a significant reduction of the total impact of a lamp, but increases the share of the manufacturing phase. Compared with all other lamp technologies, LED–lamp manufacturing has the highest environmental impact. From the manufacturing countries´ point of view, there is no environmental benefit from exported LED–lamps. This leads to the situation that LED–lamps produced for the own national market, reduce the environmental burden of the country, but LED–lamps produced for the export market increase the manufacturing country´s impact. The European ecodesign legislation does not consider the energy demand and the energy sources in the manufacturing countries outside the EU. It is recommended to introduce legislative world market correctives in the form of environmental equalisation calculation for the *grey energy*, that is the manufacturing energy consumption.

All comparative studies assume identical physical lighting parameters of the different lamp technologies, without taking into account a possible change of the user behaviour, standards or market development. There has been a rebound effect recognised in recent years which is discussed in more detail in Chapter 7, Section 7.8. The rebound effect cannot be directly calculated in the scope of a standardised streamlined life cycle assessment. However, it can be considered for example by changing the functional unit from direct comparison of lamps in terms of lumen x hours to the service of lighting per household and year.

10.12. Conclusion

The investigations of the published cradle–to–grave LCA studies for LED–lamps show that the scopes, system boundaries, and technical LCI data include variations in a wide range which makes the studies difficult to compare. In most cases it is only possible to compare the relative environmental impact shares between manufacturing stage and using phase. Transport and end–of–life treatment do not play a role within all study results and their respective system boundaries and applied LCI data. In general, the comparison of all efficient lamp technologies show that, regarding technical properties according to the state of the market in 2019, long fluorescent lamps have the best environmental performance in almost all environmental impact categories, followed by CMH–lamps and equally by LED–lamps. LED–lamps have most likely the best environmental performance for residential and outdoor spot light applications. LED–lamps for streetlamp application reach an equal or better environmental performance only if high colour temperatures are applied which is not recommendable according to the discussions in Chapter 7.

The chosen functional unit sets the direction of the investigated system model. Almost all studies considered the technical lifetime of the respective lamp technology on basis of an illumination point decoupled from possible technology dependent real lamp operation time and total illumination e.g. per year. One study found by applying the functional unit kilometer–lit–road that there is a higher number of LED–lamps necessary to reach the required illumination compared to other efficient lamps. The functional unit for lighting should include more lighting specifications besides lumen and lifetime such as colour temperature, beam angle, or glare behaviour.

The environmental assessment of LED–lamps is connected with serious problems of data uncertainties. This concerns on the one hand LCI data of the LED–packages and on the other hand technological forecasts. Since a number of published studies were made for technical policy decisions, accurate LCI data of the core components are of prime importance. The dataset for LEDs of the ecoinvent database is not applicable for the LED–packages and the few alternative determinations of LCI data do not give a complete picture of the LED–package manufacturing. This concerns the wafer and semiconductor fabrication as well as the colour converter with their sophisticated technologies and the use of various rare earths and other dopants. Consequently, the same problem arises for the phosphor manufacturing of fluorescent lamps which is also based on rare earths. Besides the problem of partially fragmentary data, the studies were performed within one decade and during this period the luminous efficacy of LED–lamps significantly increased from the range of 60 lm/W to 80 lm/W to the range of 80 lm/W to 90 lm/W for warm white retrofit bulbs and 115 lm/W for filament bulbs. The reliability of filament LED–lamps has to be observed since they are an interesting alternative to the other LED–lamp technologies. There is no information about changes and possible increase of the environmental burdens connected with the technology improvement.

Almost all studies modelled high values of the lifetime of LED–lamps in the range of 50,000 hours and also forecasted high values of the luminous efficacy in the range of 140 lm/W. Only these LED–lamps could reach a high environmental performance in the studies. The assumed parameters results in a relatively low share of the manufacturing impact of the full life cycle. The real technological and market development has stabilised at a lifetime of 15,000 hours. Together with the lower luminous efficacy, the manufacturing share of the full life cycle significantly increases and therefore, the manufacturing data uncertainties carry more weight. It is recommended to perform LCA studies including the actual technical parameters and LED–lamp designs.

LED–lamps are a further example besides e.g. photovoltaic modules for efficient technologies which reduces the global warming potential in Europe and increases the environmental burden in the manufacturing country which is in most cases in Asia. The introduction of legislative world market correctives in form of environmental equalisation calculations for grey energy and other environmental damages of the manufacturing countries could lead to a better environmental justice within the world market.

10.13. Acknowledgement

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10.14. References

Abdul Hadi, S.; Al Kaabi, M. R.; Al Ali, M. O.; Arafat, H. A. (2013). Comparative Life Cycle Assessment (LCA) of streetlight technologies for minor roads in United Arab Emirates; Energy for Sustainable Development, Volume 17, Issue 5, October 2013, Pages 438–450, 2013.

Althaus, H.-J.; Chudacoff, M.; Hischier, R.; Jungbluth, N.; Osses, M.; Primas, A. (2007). Life Cycle Inventories of Chemicals. Final report ecoinvent data v2.0 No. 8. EMPA, Swiss Centre for Life Cycle Inventories, Dübendorf, CH.

Chang M.-H.; Das D.; Varde P. V.; Pecht M. (2012). Light emitting diodes reliability review, Microelectronics Reliability 52, 762–782.

Commission Regulation (EC) (2009) No 244/2009 of 18 March 2009 implementing Directive 2005/32/EC of the European Parliament and of the Council with regard to ecodesign requirements for non-directional household lamps.

Dale, A. T.; Bilec, M. M.; Marriott, J.; Hartley, D.; Jurgens, C.; Zatcoff, E. (2011). Preliminary Comparative Life–Cycle Impacts of Streetlight Technology. Journal of Infrastructure Systems, 2011, 17(4), 193–199.

EC Directive (2009). Directive 2009/125/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for the setting of ecodesign requirements for energy–related products (recast).

Dupuis R. D.; Krames, M. R. (2008). History, development, and applications of high-brightness visible light emitting diodes, Journal of Lightwave Technology 26, 1154 – 1171.

Ecoinvent (2009). Jungbluth, N.; Stucki, M.; Frischknecht, R. (2009). Photovoltaics. In Dones, R. (Ed.) et al., Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. ecoinvent report No. 6–XII, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, 2009.

Ecoinvent Database (2019). http://www.ecoinvent.org/ [04/2019].

Franz, Manuela; Wenzl, Franz P. (2017). Critical review on life cycle inventories and environmental assessments of LED-lamps. Critical Reviews in Environmental Science and Technology, Volume 47, 2017 - Issue 21, pages 2017-2078.

Fulmek, P.; Nicolics, J.; Nemitz, W.; Schweitzer, S.; Sommer, C.; Hartmann, P.; Schrank, F.; Wenzl, F. P. (2016). The impact of the thermal conductivities of the color conversion elements of phosphor converted LEDs under different current driving schemes. Journal of Luminescence 169, 559–568.

GE Lighting, (2011). GE CMH ChromaFit Lamps. Datasheet.

Hartley, D.; Jurgens, C.; Zatcoff, E. (2009). Life Cycle Assessment of Streetlight Technologies. Mascaro Center for Sustainable Innovation, University of Pittsburgh; 72p. http://www.pitt.edu/news2010/Streetlight_Report.pdf [07/2016].

Hischier, Roland; Classen, Mischa; Lehmann, Martin; Scharnhorst, Wolfram (2007). Life Cycle Inventories of Electric and Electronic Equipment: Production, Use and Disposal. Data v2.0 (2007). ecoinvent report No. 18. / http://www.ecoinvent.org/ [04/2019].

IEA International Energy Agency (2006). Light's Labour's Lost. Policies for Energy–efficient Lighting, 2006.

Jungbluth, N.; Stucki, M.; Frischknecht, R. (2009). Photovoltaics. In Dones, R. (Ed.) et al., Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. ecoinvent report No. 6–XII, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, 2009.

Khanna, V. (2014). Fundamentals of Solid-State Lighting: LEDs, OLEDs, and Their Applications in Illumination and Displays. CRC Press.

Kölper, Christopher; Bergbauer, Werner; Strassburg, Martin; Linder, Norbert (2011). Die Licht(r)evolution. Weiße LEDs für die Allgemeinbeleuchtung. Phys. Unserer Zeit. Viley-VCH, 2/2011 (42), p. 92–98.

Koltun, P.; Tharumarajah, A. (2014). Life Cycle Impact of Rare Earth Elements; Hindawi Publishing Corporation, ISRN Metallurgy, Volume 2014, Article ID 907536, 2014.

LEDinside, a Business Division of TrendForce Corp. (2014). Global LED lighting market to reach US $25.7 billion in 2015, November. 6, 2014 – 18:14 – likchinlow.

Matthews, D. H.; Matthews, H. S.; Jaramillo, P.; Weber, C. L. (2009). Energy Consumption in the Production of High-Brightness Light-Emitting Diode, 2009 IEEE International Symposium on Sustainable Systems and Technology, 1–6, 2009; DOI: 10.1109/ISSST.2009.5156691.

McKinsey & Company (2012). Baumgartner, T.; Wunderlich, F.; Jaunich, A.; Sato, T.; Bundy, G.; Grießmann, N.; Kowalski, J.; Burghardt, S.; Hanebrink, J. (Eds.) (2012). Lighting the way: Perspectives on global lighting market. Second edition.

Mehr, M. Y.; Van Driel, W. D.; Zhang, G. Q. (2015). Progress in Understanding Color Maintenance in Solid-State Lighting Systems, Engineering 1, 170 – 178.

Navigant Consulting Europe, Ltd. (2009). Life–Cycle Assessment of Ultra–Efficient Lamps. A research report completed for the Department for Environment, Food and Rural Affairs.

Navigant (2012). U.S. Department of Energy. Life–Cycle Assessment of Energy and Environmental Impacts of LED Lighting Products. Part I–II.

Nakamura, S. (2015). Background story of the invention of efficient blue InGaN light emitting diodes (Nobel Lecture), Annalen der Physik 527, 335–349.

Neslen, A. (2015). Big brands 'cheating' consumers with false lightbulb efficiency claims. The Guardian, Thursday 17 December 2015 12.44 GMT (online). [02/2016].

Osram, Siemens Corporate Technology (2009). Life-cycle Assessment of Illuminants – a Comparison of Light Bulbs, Compact Fluorescent Lamps and LED Lamps, Executive Summary.

Pradal, N.; Chadeyron, G.; Potdevin, A.; Deschamps, J.; Mahou, R. (2013). Elaboration and optimization of Ce–doped Y3Al5O12 nanopowders dispersion. Journal of the European Ceramic Society, 33, pp. 1935-1945, 2013.

Principi, P.; Fioretti, R. (2014). A comparative life cycle assessment of luminaires for general lighting for the office – compact fluorescent (CFL) vs Light Emitting Diode (LED) – a case study. Journal of Cleaner Production, Volume 83, 15 November 2014, Pages 96–107.

Quirk, I. (2009). Life-Cycle Assessment and Policy Implications of Energy Efficient Lighting Technologies. http://nature.berkeley.edu/classes/es196/projects/2009final/QuirkI_2009.pdf [07/2016].

Sangwan, K. S.; Bhakar, V.; Naik, S.; Andrat, S. N. (2014). Life Cycle Assessment of Incandescent, Fluorescent, Compact Fluorescent and Light Emitting Diode Lamps in an Indian Scenario. Procedia CIRP 15 (2014) 467– 472.

Scholand, M. J.; Dillon, H. E. (2013). Life-Cycle Assessment of Energy and Environmental Impacts of LED Lighting Products, Part 1-3, U.S. Department of Energy, 2013.

SimaPro (2019). SimaPro. https://simapro.com/ [04/2019].

Tähkämö, L.; Bazzana, M.; Ravel, P.; Grannec, F.; Martinsons, C.; Zissis, G. (2013). Life cycle assessment of light–emitting diode downlight luminaire – a case study; Int. Journal of Life Cycle Assessment, 18, 1009-1018, 2013.

Tähkämö, L.; Martinsons, C.; Ravel, P.; Grannec, F.; Zissis, G. (2014). Solid State Lighting Annex: Life Cycle Assessment of Solid State Lighting. Final Report; Energy Efficient End-Use Equipment, Int. Energy Agency, 2014.

Tähkämö, L.; Halonen, L. (2015). Life cycle assessment of road lighting luminaires – Comparison of light–emitting diode and high–pressure sodium technologies. Journal of Cleaner Production 93 (2015) 234–242.

U.S. Department of Energy DOE (2016). Solid–State Lighting R & D Plan. June 2016. DOE SSL Program, "R&D Plan," edited by James Brodrick, Ph.D.

Welz, T.; Hischier, R.; Hilty, L. M. (2011). Environmental impacts of lighting technologies – Life cycle assessment and sensitivity analysis. Environmental Impact Assessment Review. Volume 31, Issue 3, April 2011, Pages 334–343.

Wilburn, D.R. (2012). Byproduct metals and rare-earth elements used in the production of light–emitting diodes – Overview of principal sources of supply and material requirements for selected markets: U.S. Geological Survey Scientific Investigations Report 2012–5215, 15 p., at http://pubs.usgs.gov/sir/2012/5215/ [03/2019].

Zissis, G. (2016). Quality and Health Aspects of LED Lamps for the European Residential Lighting Market. LED professional Review. Trends & Technologies for Future Lighting Solutions, July/August 2016, Issue 56, pp. 40–56.

Zissis, Georges; Bertoldi, Paolo (2018). Status of LED-Lighting world market in 2017, Ispra, European Commission, 2018, JRCxxxxx.

(1) Navigant Consulting (2009). Life Cycle Assessment of Ultra-Efficient Lamps; Department for Environment, Food and Rural Affairs, UK, 2009.

(2) Scholand, M. J.; Dillon, H. E. (2013). Life-Cycle Assessment of Energy and Environmental Impacts of LED Lighting Products, Part 1-3, U.S. Department of Energy, 2013.

11. Environmental Evaluation of Photovoltaic and Other Energy Systems by Mid–Point and Highly Aggregated Assessment Methods

Abstract: *The global application of photovoltaic (PV) systems has increased almost tenfold in the last seven years. Although renewable energy systems can rise a variety of questions, environmental sustainability is currently primarily linked to CO_2 emissions. This work deals with the life cycle assessment (LCA) of renewable and fossil energy sources, in particular photovoltaic plants. The study addresses the different findings of highly aggregated and midpoint assessment methods and their application to local infrastructure projects. The Sustainable Process Index (SPI) is compared with the Carbon Footprint and the Agricultural Land Occupation and the respective data situation and applicability are discussed. In the area of renewable energies, the SPI and the Carbon Footprint show that PV and biogas plants have significantly higher impacts than other renewable energy sources. Furthermore, the environmental impact of PV systems per generated kWh depends mainly on the place of manufacture and less on efficiency and operation location. It also shows that none of the methods alone can provide a comprehensive picture of environmental sustainability: An important factor for renewable energies is land consumption, which is visualised with the indicator Agricultural Land Occupation. It shows that biogas based on maize silage has the highest direct land consumption of all energy sources.*

11.1. Introduction

The *EurObserv´ER* reports that in Europe, the total cumulative installed nominal power of photovoltaic plants was 106.6 GW_p at the end of 2017 (EurObserv´er, 2018). In 2017, on global level, 70 % of the modules were produced in China and Taiwan, 15 % in Rest of Asia–Pacific & Central Asia, and only 5 % in Europe. 95 % of the modules are made in Si–wafer technology and 62 % in multi–crystalline silicon technology. Compared to 2015, the share of total produced mono–crystalline modules increased by 32 %. Figure 11.1. shows the growth of the global PV installation from 2010 to 2017. (Fraunhofer, 2016/2019)

In 2017, Europe had a total share of the cumulative global PV installation of 28 % with a regional production volume of only 5 % of the global market according to figure 11.1. This means, considering that the European module production share is for the own market, 5 % modules are assumed to be imported from America, that about 78 % of the upstream environmental impacts per produced kWh in Europe are assigned to the environmental manufacturing conditions of China and the Asia–Pacific region.

There are different methods to describe the environmental performance for photovoltaic modules. The environmental impact can be described by life cycle assessment per module unit, per m^2 or per produced kWh over the expected lifetime. The last method is dependent on the geographical location where the PV plant is installed, as is the energy payback time.

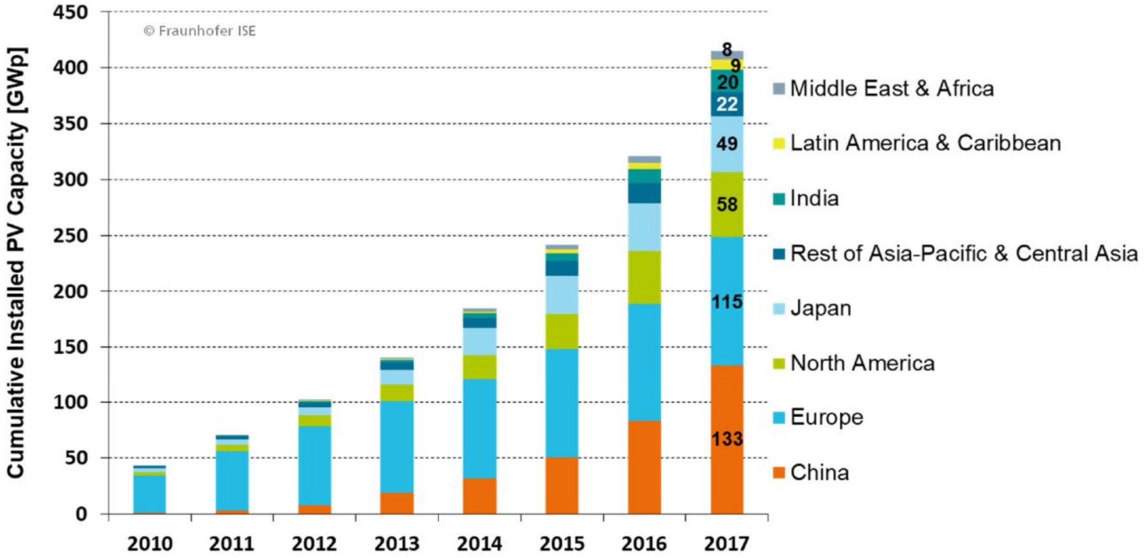

Figure 11.1. Annual global cumulative PV installation from 2010 to 2017. (Fraunhofer, 2019)

Due to various PV technologies, country specific different energy sources providing the manufacturing energy, manufacturer and technology dependent cell efficiencies, and location–dependent annual global horizontal irradiance (GHI) lead to different environmental impact values per produced kWh.

The environmental impact per produced kWh can be evaluated by midpoint or highly aggregated assessment methods. Midpoint indicators are useful to investigate specific environmental problems in detail at the expense of the overall view of the environmental performance of a device or service. Highly aggregated environmental indicators are characterised by the weighting decision being made between several different environmental characterisation factors prior to aggregation. Therefore, the assessment results provide a comprehensive picture of the environmental pressure of the device or service (Narodoslawsky & Stoeglehner, 2010), but are based on a normative weighting method. Both methodological approaches are investigated in this study for different energy sources.

In the common frame of the thesis, the study addresses the differences between the assessment methods Sustainable Process Index (SPI) as highly aggregated method and the Carbon Footprint as well as the Agricultural Land Occupation as midpoint assessment indicators. Further, the application of environmental assessment systems on local infrastructure projects and their correlation and informative value within the systematics of life cycle assessment is investigated.

11.2. Materials and Methods

The environmental characteristics of PV modules are examined on global level based on literature data and further calculated for local PV infrastructure in the area of the city of Eggenburg in Lower Austria. To compare the environmental relations between midpoint and highly aggregated assessment methods, the Global Warming Potential (or Carbon Footprint), the Agricultural Land Occupation, and the Sustainable Process Index (SPI) are determined for the most important conventional and renewable energy sources. The midpoint indicator global warming potential is chosen according to the European climate strategy. The Agricultural Land Occupation is additionally calculated for a free–field PV plant, two hydro power plants, and a biogas plant based on maize silage in Austria. The land use indicator is chosen to visualise and compare future land use pressures of renewable energy sources.

The investigations include the preparation and combination of literature data from scientific publications and own calculations. Furthermore, the environmental indicators Global Warming Potential (GWP), Agricultural Land Occupation (ALO), and Sustainable Process Index (SPI) are determined to compare the environmental impact results of different energy sources and their information value on local level. In literature, different environmental indicators are applied to describe the global warming potential. Although they do not base on the same calculation method, in this work the indicators GWP, Climate Change, and Carbon Footprint are equated. The used database for the newly calculated indicators ALO and GWP is ecoinvent 3.4 (Ecoinvent, 2019) and the assessment methods are ReCiPe Midpoint (H) (Goedkoop, 2013) and SPI (Narodoslawsky & Krotscheck, 1995; SPI, 2019).

The study is divided into the following sections:

Energy Amortisation Time of PV Systems

- Energy payback time (EPBT)
- Energy return on energy invested (EROI) factor of PV modules
- EROI of storage batteries

Midpoint and Highly Aggregated Assessment of Energy Sources

- Carbon Footprint of PV modules in Lower Austria
- Comparison of the Carbon Footprint of PV modules manufactured in Europe and China
- Carbon Footprint of renewable and fossil energy sources
- Sustainable Process Index of renewable and fossil energy sources
- Agricultural Land Occupation of renewable and fossil energy sources

11.3. Energy Amortisation Time of PV Systems

The energy amortisation can be used as a simplified sustainability parameter. The energy payback time (EPBT) indicates how many years operation are required to amortise the "grey energy", that is the energy needed for manufacturing. The energy return on energy invested (EROI) factor indicates the energy harvesting factor of electrical energy generation over the full lifetime. A meta–study conducted by Bhandari et al. found follwing range of results for PV systems (Bhandari et al., 2015):

- EPBT: 1.0 to 4.1 years
- EROI factor: 8.7 to 34.2

The results show that in both cases EPBT and EROI the uncertainty is up to a factor of 4. For local ratings, adjusted values must be calculated separately to be meaningful. The Fraunhofer Institute published reference values of the energy payback time differentiated between Northern and Southern Europe (Fraunhofer ISE, 2019):

- EPBT in Northern Europe: 2.5 years
- EPBT in Southern Europe: 1.5 years

Off–grid PV systems additionally require a battery to store the surplus produced energy. The PHOTOVOLTAIC AUSTRIA Federal Association provides data for storage batteries for off–grid PV plants (PV Austria, 2016):

- PV batteries: have a 9–fold storage capacity during the lifetime than the required manufacturing energy

11.4. Midpoint and Highly Aggregated Assessment of Energy Sources

11.4.1. Carbon Footprint of PV Modules

During operation in the use stage of the life cycle, there are almost zero greenhouse gas emissions of the PV plants. However, in terms of the overall balance, the efforts for manufacturing are assigned to each produced kWh over a nominal lifetime. De Wild–Scholten determined in her work the Carbon Footprint per kWh dependent on the module technology, materials and plant components, and the place of manufacture and operation, respectively, for the state of technology of 2011. Figure 11.2., left diagram, shows the relations for PV plant operation per kWh updated for Eggenburg (own calculations based on Chinese modules, data source for global irradiation and module efficiencies see Chapter 6 table 6.1.), and a comparison between European and Chinese (CN) production sites in the right diagram (De Wild–Scholten, 2014). The percent values represent the module efficiencies of the related technology year.

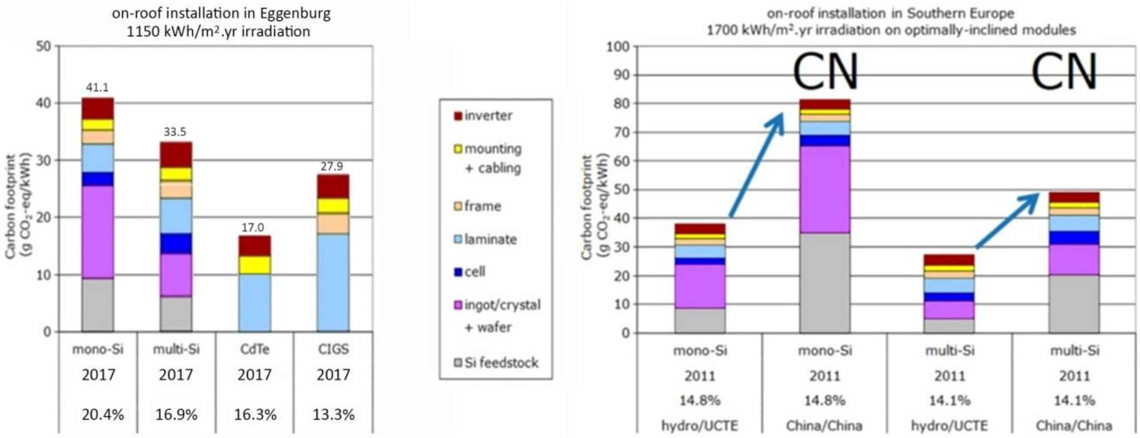

Figure 11.2. Carbon Footprint of PV modules per kWh, left diagram updated and own calculation for 2017. (De Wild–Scholten, 2014, modified)

Valid for large areas of Lower and Upper Austria north of the Alps, mono–Si PV modules produced in Europe with partly hydro power based energy show an environmental pressure of 41.1 g CO_2–e per produced kWh. Multi–Si PV modules emit 18.5 % less and cadmium telluride modules emit 59 % less CO_2–e/kWh. However, the lower CO_2–e emissions are at the expense of a larger area consumption for the produced energy in terms of a required greater number of modules.

It should be noted that the European manufacturing scenario shown in figure 11.2. assumes, that the manufacturing of the silicon feedstock (grey colour) of the Si–based modules is calculated with hydro power electrical energy (De Wild–Scholten, 2014), although it is not realistic for the whole sector**.** The wafer, cell, and module manufacturing is calculated with the average European electrical energy mix UCTE. The right side scenario of figure 11.2 shows that the same module production in China doubles the Carbon Footprint per produced kWh, it significantly increases the "embedded" carbon of the PV modules. The comparison between the technology years 2011 (right) and 2017 (left) shows that due to higher module efficiency but lower solar irradiation in Northern Austria the total Carbon Footprint per produced kWh increased only by 8 % in case of mono–Si modules.

The manufacturing location and the respective used energy sources are an important factor of the CO_2–e emissions of a photovoltaic plant. A doubling of emissions depending on whether the module is produced in Europe or China must not be ignored in the buying decision by focussing only on cheap manufacturing costs. Figure 11.2. shows that the module efficiency and the plant location in Europe has far less impact on CO_2–e emissions than the place of manufacture. This circumstance should have more attention in future environmental policy.

11.4.2. Carbon Footprint and SPI of Energy Sources, European Average

Greenhouse gas emissions relate to only one of a large number of characterisation categories. To describe the environmental performance of a product, further environmental indicators should be taken into consideration. Highly aggregated environmental indicators weights several different environmental characterisation factors prior to aggregation. As an example, figure 11.3. shows a comparison of the Carbon Footprint and the Sustainable Process Index (SPI) per generated kWh for renewable and fossil energy based electrical energy generation (Data sources: De Wild–Scholten, 2014; Narodoslawsky & Krotscheck, 1995; SPI, 2019). UCTE represents one calculation method for the average European electrical energy mix. In case of the Carbon Footprint, the energy source biogas is added by own calculation (data source: ecoinvent 3.3). Data source for the Carbon Footprint is ecoinvent 2.2 (De Wild–Scholten, 2014; Ecoinvent, 2019).

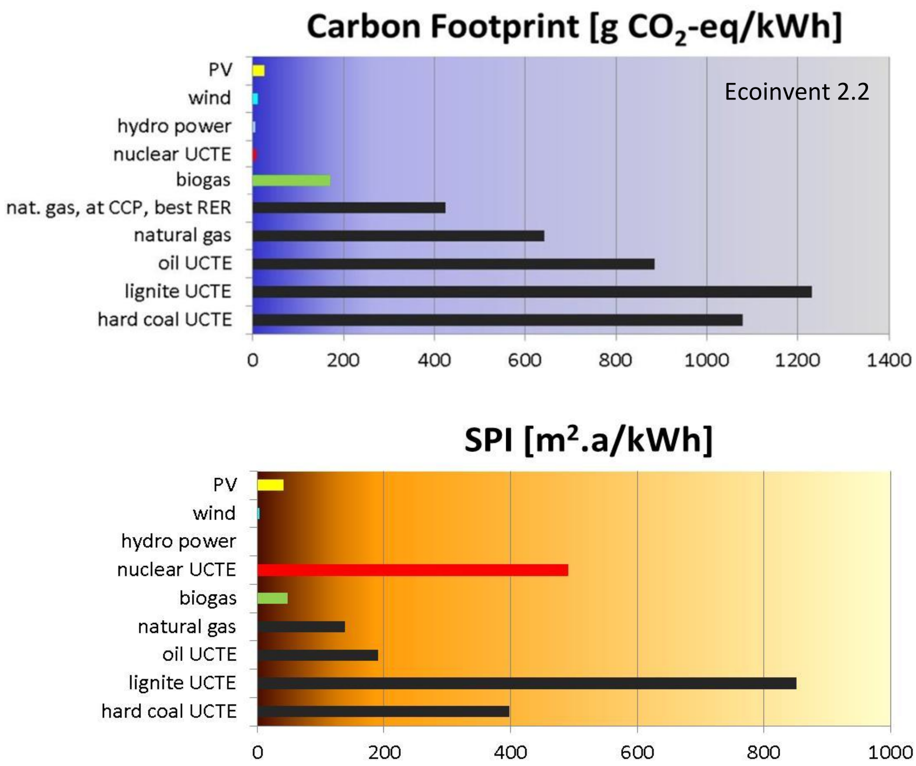

Figure 11.3. Comparison of the Carbon Footprint (De Wild–Scholten, 2014; modified) and the ecological footprint SPI (own calculation) of energy sources, average European data.

Both assessment methods show that all renewable energy sources have a significant lower environmental impact than all fossil based energy sources, but the relations are different. Within the group of renewable energy, both assessment methods show that hydro power followed by wind energy have the lowest environmental impacts. In case of the Carbon Footprint, PV systems emit twice as much CO_2–eq and biogas plants emit 15 times as much CO_2–eq as wind power plants.

The highly aggregated SPI method shows that PV systems and biogas plants generate a similar environmental pressure that is ten times as much as wind power. The results show further, that only considering the CO_2–equivalent is too one–sided and does not reflect real sustainability relations. The largest differences between the results of CO_2–eq and SPI are the relations compared to lignite and nuclear power. It shows the difference to a one–sided assessment standard such as CO_2–e which does not consider other emissions. The SPI clearly demonstrates that the energy source lignite is an environmental hot spot followed by nuclear energy and hard coal.

11.4.3. Carbon Footprint of Energy Sources, Austrian Average

The application of a portfolio of midpoint assessment methods have the difficulty which one to integrate and how to weight the assessment results. In the following, two very different environmental indicators are evaluated for renewable and fossil energy sources on local level in Austria (AT). The Carbon Footprint and the Agricultural Land Occupation (ALO) are calculated on basis of the ecoinvent 3.4. In case of PV plants, two silicon module technologies are compared as well as the difference beween roof–top and free–field plants which are both provided by the ecoinvent data base. Figure 11.4. shows the results of the Carbon Footprint for all main energy sources. Since Austria has no nuclear power plant and no lignite power plant, the Carbon Footprints of the next located ones in the Czech Republic (CZ) are determined as they are included in the Austrian power trading.

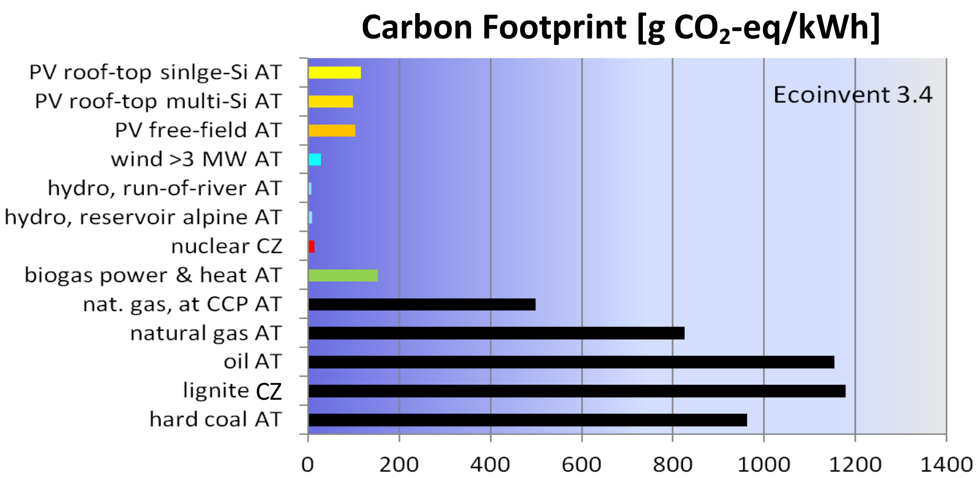

Figure 11.4. Comparison of the Carbon Footprint of energy sources, average Austrian data; CCP: combined cycle power plant (Own work, Data source: ecoinvent 3.4).

The results show that the Austrian relations based on Ecoinvent 3.4 are similar to the European results based on Ecoinvent 2.2 according to figure 11.3., but almost all values are higher. Based on the existing values, in case of renewable energy, the CO_2–eq emissions of PV plants are 5 times higher, the emissions of wind energy are 2.5 times higher and the emissions of biogas are reduced by 10 % compared to the European average values in figure 11.3. The newly calculated biogas is allocated into power and heat, whereby the efficiency shares are 0.37 for electrical energy and 0.53 for heat. Within the PV group, mono cyristalline PV modules emit 18 % more CO_2–eq than multi crystalline modules. In the field of fossil energy sources, natural gas and oil have a significant worse Carbon Footprint as the European average calculated with ecoinvent 2.2.

Nevertheless, it is not clear if the Austrian market and technologies are different and/or in general, the input data of the ecoinvent version 3.4 partly have been revised upwards. The European electrical energy mix UCTE for individual energy sources is no longer available in ecoinvent 3.4. To clarify this, other countries would have to be compared, too.

11.4.4. Agricultural Land Occupation of Energy Sources, Austrian Average

The Agricultural Land Occupation (ALO) is an important environmental indicator because land use will be an increasing future conflict factor between food and feed production, energy generation, chemical and textile feedstock, and preservation of natural habitats. Figure 11.5. shows the comparison of the Agricultural Land Occupation per kWh of renewable and fossil energy sources for Austria (AT) based on ecoinvent 3.4. For nuclear power and lignite power plants, which are not available in Austria, the ALO of the next located ones in the Czech Republic (CZ) are determined as they are included in the Austrian power trading.

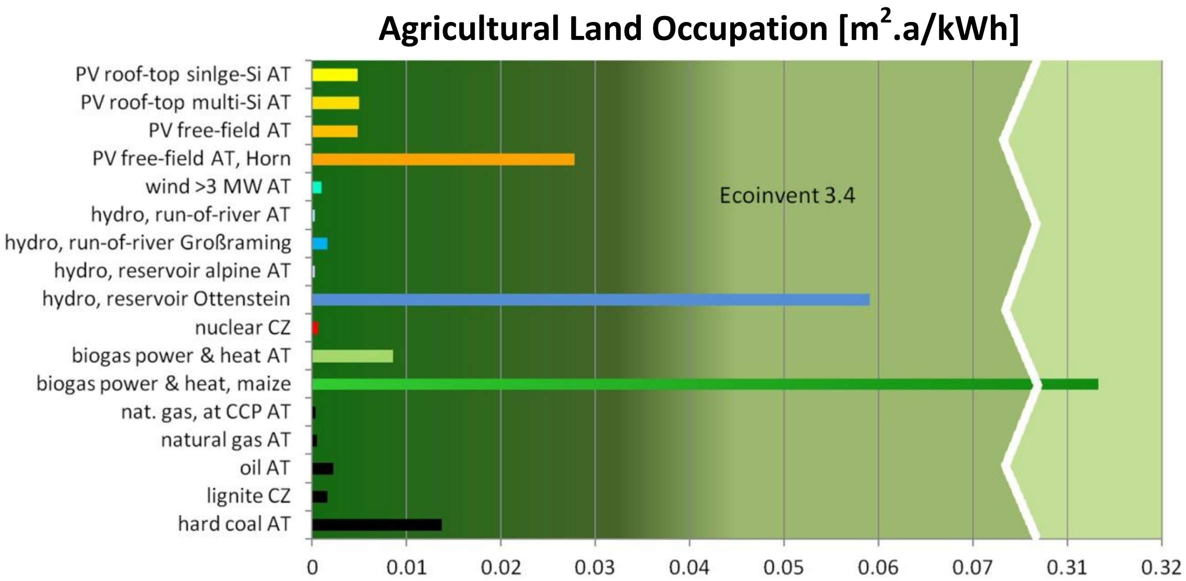

Figure 11.5. Comparison of the Agricultural Land Occupation of energy sources, average Austrian data and case studies (Own work; Data source AT, CZ: Ecoinvent 3.4).

Comparative ALO values are newly calculated for case studies of an Austrian free–field power plant, a medium size run–of–river power plant, a reservoir power plant, and a biogas power plant based on maize silage.

The database does not include the land use for free–field PV plants in operation, since the results are in the same range for all three PV scenarios. Because of that, new calculations were performed by the example of a free–field PV plant in Horn, Lower Austria. Figure 11.6. shows a satellite image of the free–field PV system in Horn with yellow marked area and the surrounding fence (Google Earth, 2019). The corresponding data are the following: The installed total power is 850 MW_p (NÖN, 2014), the fenced area measured in Google Earth is 21900 m^2, and the average annual electrical energy production is assumed with 1.119 kWh/W_p (see Chapter 6, section 6.4.3.). The resulting additional land use is 0.023 m^2.a/kWh.

Figure 11.6. Free–field PV plant and fenced area marked in yellow in Horn, Lower Austria. (Own work; Data source: Google Earth, 2019; Image recording data: 8/15/2017)

Since the standard ALO values for hydro power marked with AT according to figure 11.5. are very low, in addition to the original data, the direct land consumption is newly calculated by the example of two medium–sized hydropower plants in Austria.

The datasets of ecoinvent 3.4 for hydro power electrical energy do not include the land consumption of the reservoir. In case of run–of–river power plants, the ALO contribution tree of the original dataset shows that 64.1 % is related to dried sawnwood production followed by 8.71 % steel production. The newly calculated case study is based on the medium size run–of–river power plant in Großraming of the Enns river, Upper Austria.

Figure 11.7. shows the satellite image of the region of the National Park Kalkalpen with the outlined Enns river between the power plants Weyer and Großraming. The length of the backwater reservoir is 13.2 km measured in Google Earth. The reservoir is coloured in light blue and within the "original" river with an estimated width of 60 m is coloured in dark blue. The additional land occupation (A_{ALO}) is determined by measuring the reservoir area ($A_{reservoir}$) in Google Earth minus the estimated original river surface (A_{river}) without power plant: $A_{ALO} = A_{reservoir} – A_{river} = 1.17\ km^2 – 0.81\ km^2 = 0.36\ km^2$. The annual electrical energy producton of the power plant Großraming is 270.7 GWh (Verbund, 2019). The additional land use results to 0.0013 m^2.a. In this case, the direct land use during operation is 8 times higher per generated kWh than the ALO provided by ecoinvent 3.4.

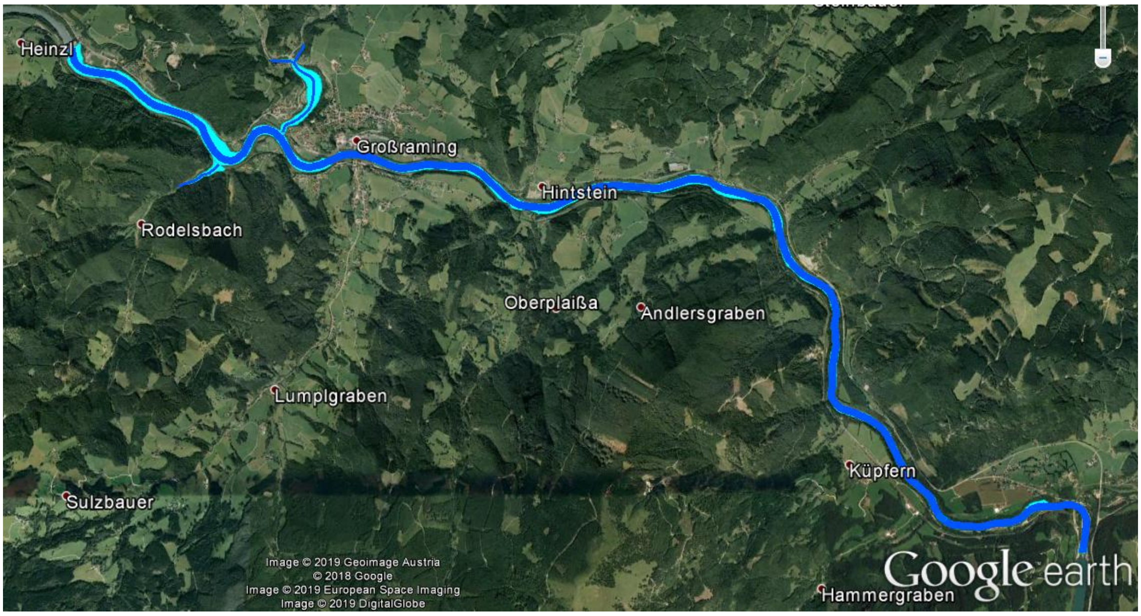

Figure 11.7. Run–of–river power plant in Großraming, Lower Austria.
Light blue: reservoir, dark blue: original river width. (Own work;
Data source: Google Earth, 2019; Image recording data: 1/1/2000)

The second case study for hydro power plants refers to the storage power plant in Ottenstein, Lower Austria. The Ecoinvent 3.4 dataset documentation defines in case of storage power plants in Austria the reservoir area with an average value of 0.00345 m^2.a per produced kWh (Ecoinvent, 2019). This value is not considered in the dataset itself and therefore, in the results of the calculated ALO which amounts to 0.00012 m^2.a per kWh. The reason is unclear, it could be because the dataset refers to alpine regions without agriculture. The ALO contribution tree of this original dataset result shows that 32 % of the area is related to steel production, 20 % to on–site electrical energy consumption, 15.2 % to cement production, and 12.6 % to transport and treatment of waste reinforced concrete.

The newly investigated storage power plant in Ottenstein is located in the region of Waldviertel Mitte in Lower Austria. Figure 11.8. shows the satellite image of the region with the reservoir outlined in light blue colour. The reservoir has a length of 9.7 km and a surface area of 4.3 km^2 (Land NÖ, 2017). The lost area usable for agriculture and forestry is actually much larger than the surface of the lake, since the shore slope would have to be included. However, as there are no exact data available, it will not be further considered. The electrical energy production is assumed with 72.4 GWh per year (Schlager et al., 2011). The ALO of the reservoir results to 0.059 m^2/kWh. The result is 500 times higher than the value of Ecoinvent 3.4. However, the ALO of reservoir power plants have worse values than run–of–river power plants because they are designed for storage and peak load balancing.

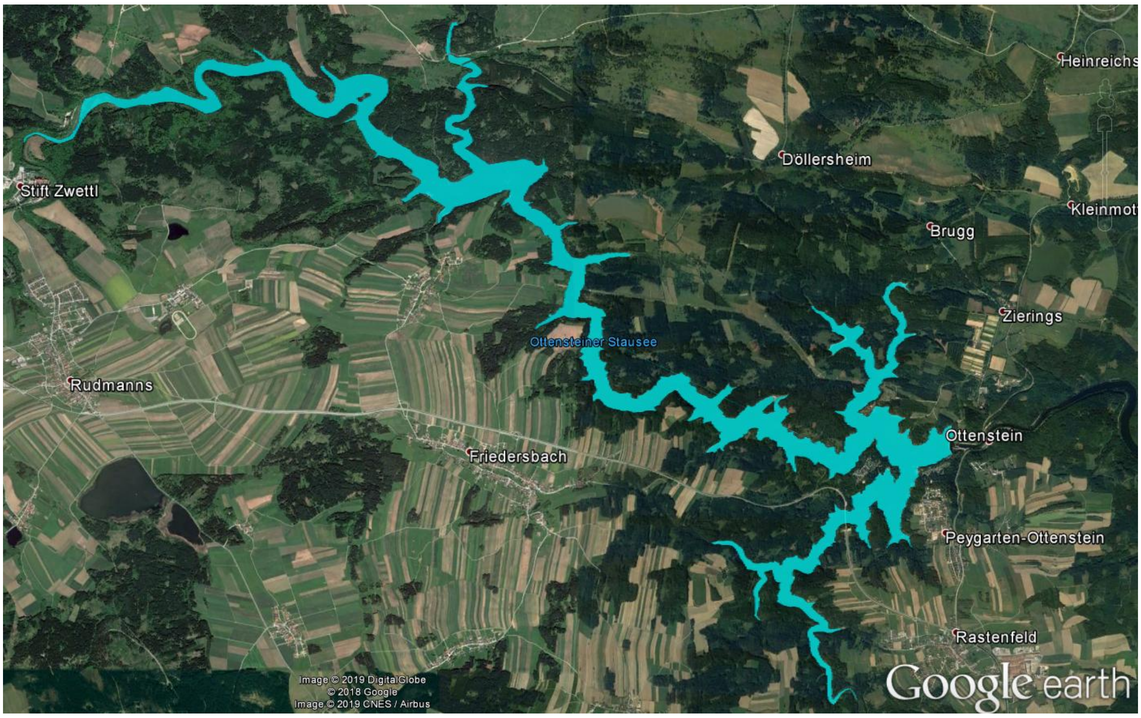

Figure 11.8. Reservoir power plant in Ottenstein, Lower Austria.
Light blue: reservoir. (Own work; Data source: Google Earth, 2019;
Image recording data: 8/15/2017)

The ecoinvent 3.4 dataset for biogas produced in Austria (biogas power & heat AT, see figure 11.5) is based on the treatment of a mixture of organic residuals such as biowaste, sewage sludge, manure, and used vegetable cooking oil. All waste materials have no further input flows and emissions, because they are all allocated to the product before it became waste. Therefore, concerning the agricultural land occupation, there are no direct land use input data. Modern biogas plants are fed with organic materials that are specially grown for this purpose. Therefore, a comparative value for biogas is newly calculated (biogas power & heat, maize, see figure 11.5.). The biogas production is based on maize silage which has the highest harvesting factor compared to grain or grass silage. The calculation of the ALO is based on following parameters: In Austria, the average harvest yield of the entire maize plant (green mass) is 47 t/ha (Resch et al., 2009). The weight loss of the maize silage is 10 % (BayLfU, 2007). To produce 1 m^3 biogas 5.71 kg maize silage is needed and the energy generation per m^3 biogas is 2.74 kWh thermal energy (64 %) and 1.57 kWh electrical energy (36 %) based on data by (Hutňan, 2015). Considering these parameters and the allocation between power and heat, the generation of 1 kWh electrical energy of maize based biogas relates to an ALO of 0.313 m^2.a.

For the sake of completeness, the direct land use of wind power plants in operation is also not considered in Ecoinvent 3.4. The calculation causes some difficulties, since the ground base of the plant is relatively small and in general, agriculture can be run in the vicinity of the plants. However, there are other restrictions on, for example, the colonisation, livestock farming, and wildlife habitat conservation due to noise emissions, infrasound, drop shadows or direct wing flapping. In the work of Dumke the direct land consumption of a 7.5 MW wind power plant is specified with 500 m^2. The land consumption of wind parks is defined by the total surrounding area of all individual installations (Dumke, 2017).

The results of the ALO show opposite relations than the Carbon Footprint concerning the group of renewable and the group of fossil energy sources. The newly calculated biogas based on maize, reservoir hydro power, and free–field PV plants including the actual land use while operating have an unacceptable large value of ALO compared to other energy sources. All other PV scenarios and biogas based on waste organic materials also show larger values compared to fossil energy sources. Run–of–river hydro power shows the best environemental performance of the renewable energy sources in terms of land use, wind energy is not considered because of the unclear calculation method, which however is expected to be higher than in figure 11.5. Except of hard coal, all fossil based energy sources show relatively low land use per generated kWh.

The relations of the Agricultural Land Occupation for PV and biogas should be an indicator for future environmental pressures and land conflicts by excessive expansion of free–field and maize based biogas plants which has not yet have come to light compared with the use of fossil based energy.

The comparison of Agricultural Land Occupation (ALO) with the Sustainable Process Index (SPI) clearly shows the different implications of both assessment methods although they have the same unit. It demonstrates the difference between a one–sided midpoint with a highly aggregated indicator. The direct land consumption of a roof–top PV plant is about 10,000 times lower and in case of fossil oil the ALO is about 80,000 times lower than the result of the SPI and therefore the ALO indicator is inappropriate for sustainable assessment of land consumption.

11.5. Conclusion

The study investigated the informative value and comparability of environmental parameters of photovoltaic plants and other renewable and fossil energy sources assessed by energy balance, midpoint and highly aggregated LCA methods. In detail, the energy amortisation time, Carbon Footprint, Sustainable Process Index and Agricultural Land Occupation were assessed and compared.

The Carbon Footprint of PV modules is strongly dependent on the technology and the place of manufacturing. Cadmium telluride modules emit lower CO_2–eq by the factors 2 and 2.5 than silicon based modules, but at the expense of the required total area. The technological increase in module efficiency and the installation location in Northern or Southern Europe have far less impact on the Carbon Footprint per kWh than the place of manufacture. The doubling of emissions depending on whether the module is produced in Europe or China has to be taken into account in the course of economic considerations of manufacturing costs.

The comparison of renewable energy sources in Austria shows that the Carbon Footprint of PV modules and biogas are 4.3 and 5.7 times higher than wind energy and up to 35 times higher than hydropower. A comparison between the ecoinvent database versions 2.2 and 3.4 shows that the input data of almost all energy sources have been revised upwards. In case of storage hydro power plants, the decades of greenhouse gas emissions from dying plants and organisms due to the flooding of the reservoir are not taken into account and should be evaluated.

The results of the highly aggregated SPI show similar trends, but the environmental pressure of PV systems and biogas plants is ten times as much as wind power. Further, nuclear power and lignite based energy are identified as environmental hot spots.

The midpoint indicator Agricultural Land Occupation gives a different picture of the environmental situation of all energy sources. Maize based biogas plants, hydro power reservoir plants, and free–field PV plants including the real land use while operating have an unacceptable large value of ALO compared to other energy sources. All other PV scenarios and particularly biogas also show large values compared to fossil energy sources. The SPI as highly aggregated sustainability indicator includes the environmental pressure of direct land use conflicts in the partial footprint "Footprint for direct area use and installations", but this value is very small compared to the total SPI and has to be actively visualised.

The different assessment methods show that none of the indicators alone is capable of providing a comprehensive visualisation and a proper weighting of the environmental performance of energy sources. Especially in the field of energy supply, apart from the climate relevance represented by the carbon footprint and the goal of reducing toxic emissions and non–renewable resources as represented by the SPI, direct land consumption has become much more of a priority. While toxic emissions can at least theoretically be dealt through suitable measures, the massive increase in land use for renewable energies creates new challenges.

11.6. References

BayLfU – Bayerisches Landesamt für Umwelt (2007). Biogashandbuch Bayern – Materialienband; Kap. 1.7, Kap. 2.2.3, Stand Mai 2007. https://energypedia.info/images/6/65/Biogashandbuch_Bayern.pdf [04/2019].

Bhandari, Khagendra P.; Collier, Jennifer M.; Ellingson, Randy J.; Apul, Defne S. (2015). Energy payback time (EPBT) and energy return on energy invested (EROI) of solar photovoltaic systems: A systematic review and meta-analysis. Renewable and Sustainable Energy Reviews 47 (2015) 133-141.
http://astro1.panet.utoledo.edu/~relling2/PDF/pubs/life_cycle_assesment_ellingson_apul_(2015)_ren_and_sustain._energy_revs.pdf [14.9.2017].

De Wild–Scholten (2014). Environmental aspects of photovoltaics. Smart Green Scans. Presentation at the 12. Österreichische Photovoltaik–Tagung Linz, Austria, 4 Nov. 2014.
http://smartgreenscans.nl/publications/deWildScholten-2014-Environmental-aspects-of-PV.pdf [10.1.2017].

Dumke, Hartmut (2017). Erneuerbare Energien für Regionen - Flächenbedarfe und Flächenkonkurrenzen. Dissertation, TU Wien, 2017.
Online available via https://repositum.tuwien.ac.at/ [04/2019].

Ecoinvent (2019). http://www.ecoinvent.org/ [03/2019].

Eurobserv’er (2018). Photovoltaic Barometer 2018. http://www.eurobserv-er.org [03/2019].

Fraunhofer ISE (2016). Photovoltaics Report, updated: 17 November 2016. https://www.ise.fraunhofer.de / de / downloads / pdf-files / aktuelles / photovoltaics-report-in-englischer-sprache.pdf [7.1.2017].

Fraunhofer ISE (2019). Photovoltaics Report, updated: 14 March 2019. https://www.ise.fraunhofer.de/de/veroeffentlichungen/studien/photovoltaics-report.html [03/2019].

Goedkoop, Mark; Heijungs, Reinout; Huijbregts, Mark; De Schryver, An; Struijs, Jaap; van Zelm, Rosalie (2013): ReCiPe 2008 – A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First Edition (revised). Ruimte en Milieu. Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer, 2013.

Google Earth (2019). https://www.google.com/earth/ [04/2019].

Hutňan, Miroslav (2015). Maize Silage as Substrate for Biogas Production. Intech open, 2015 pp. 173-196. http://dx.doi.org/10.5772/64378 [04/2019].

Land NÖ (2017). Statistisches Handbuch des Landes Niederösterreich. 41. Jahrgang 2017. NÖ Schriften 215 – Information. Amt der Niederösterreichischen Landesregierung, 2017.

Narodoslawsky M.; Krotscheck C.; (1995). The sustainable process index (SPI): Evaluating processes according to environmental compatibility; J. of Hazardous Materials, 1995, 41 (2+3), 383-397.

Narodoslawsky, Michael; Stoeglehner, Gernot (2010). Planning for Local and Regional Energy Strategies with the Ecological Footprint. Journal of Environmental Policy & Planning Vol. 12, No. 4, December 2010, 363–379.

NÖN (2014). Horn: Photovoltaik-Anlage versorgt 250 Haushalte. Nieederösterreichische Nachrichten NÖN, online Article 30. December 2014 (09:56). https://www.noen.at/horn/horn-photovoltaik-anlage-versorgt-250-haushalte-top-4305900# (German) [1.4.2019].

PV Austria (2016). PHOTOVOLTAIC AUSTRIA Federal Association. Stromspeicher. https://www.pvaustria.at/pv-speicher/. (German) [1.4.2019].

Resch, Reinhard; Wiedner, Günther; Tiefenthaller, Franz; Stromberger, Wolfgang; Frank, Peter; Meusburger, Christian (2009). Abschlussbericht Silageprojekt, Wissenschaftliche Tätigkeit Nr. 3561 (100535); Qualitätsbewertung von österreichischen Grassilagen und Silomais aus Praxisbetrieben. Evaluation of Austrian grass- and maizesilage. Lehr- und Forschungsanstalt für Landwirtschaft Raumberg-Gumpenstein, A-8952 Irdning des Bundesministeriums für Land- und Forstwirtschaft, Umwelt- und Wasserwirtschaft.

Schlager, Otmar; Brandner-Weiß, Renate; Lunzer, Horst; Sturm, Ansbert; Weltzl, Adolf; Brandner, Gottfried; Müllner, Markus; Müller, Silke (2011). Umsetzungskonzept für die Klima- und Energiemodellregion Kampseen. Energieagentur der Regionen (EAR) im Auftrag der Klima- und Energiemodellregion Kampseen. Waidhofen an der Thaya, 2011.

SPI (2019). Sustainable Process Index. http://spionweb.tugraz.at/ [03/2019].

Verbund AG (2019). Kraftwerke. https://www.verbund.com/de-at/ueber-verbund/kraftwerke [04/2019].

12. Environmental Assessment of a New Power Discrete Package in Embedding Technology

Abstract: *The study investigates the environmental performance of a new design for an electronic 50 W power discrete package fabricated in embedding technology for application in harsh environment in comparison with a commercially available sample in a D^{2}PAK package design. Power discrete packages include embedded power diodes or voltage limiters for e–mobility application. The applied assessment method is the Sustainable Process Index (SPI) provided by SPIonWeb of the Graz University of Technology. The aim of the work is to investigate the environmental trends for the fabrication of miniaturised electronic components. The study addresses the application of highly aggregated assessment methods as a tool in the product design phase. Furthermore, the question of LCI data uncertainties and the applicability of existing life cycle inventory data provided by commercial databases are discussed. The new package design shows an increase of the ecological footprint by a factor of three compared to the D^{2}PAK design per sample unit and by a factor of 20 on mass unit within the system boundary. The environmental hot–spot is the printed circuit board manufacturing of the embedding technology mainly caused by the electrical energy consumption based on fossil energy sources.*

12.1. Introduction

Electronic applications require high reliability in the e–mobility sector in harsh environments regarding mechanical stability, temperature, or humidity. A further challenge is dealing with restricted mounting space. A new packaging concept in embedding technology has been developed for thinned power transistor and power diode chips (Unger et al., 2015). The power components are applied as rectifiers or free-wheeling diodes for electric motors used in e–bikes and other electrical vehicles.

The significant innovation of the concept lies on an effective double–sided large–area chip cooling and on the reduction of switching losses due to minimisation of parasitic impedance of the embedded power circuit. The new developed process makes it possible to directly embed power semiconductors such as diodes in epoxy resin inside the printed circuit boards instead of populating the components in surface mount technology (SMT). This new technology allows significant volume and mass reduction of the power package, and an improved electronic performance.

In this study, the environmental performance of a novel embedding technology for the industrial fabrication of a 50 W power discrete package is investigated and compared with a functionally equal conventional SMT package design. In the frame of the thesis, the study investigates the application of a highly aggregated environmental assessment method of the ecological footprint family and further addresses data uncertainties and the applicability of existing life cycle inventory data provided by commercial databases.

12.2. Materials and Methods

The environmental assessment includes a cradle–to–gate attributional comparative LCA study of two power components made by different technologies. The investigated electronic components are a 50 W commercially available conventional low drop power Schottky rectifier with a housing of a D²PAK package and a new embedding technology package design of a demonstrator of equal function. In technology research, the term "demonstrator" is used for a prototype that serves as demonstration and proof–of–concept model for a new technology. Figure 12.1. shows an image of the samples soldered onto a test set–up. The new demonstrator is marked with a yellow rectangle. The test set–up itself is not within the investigated system boundary.

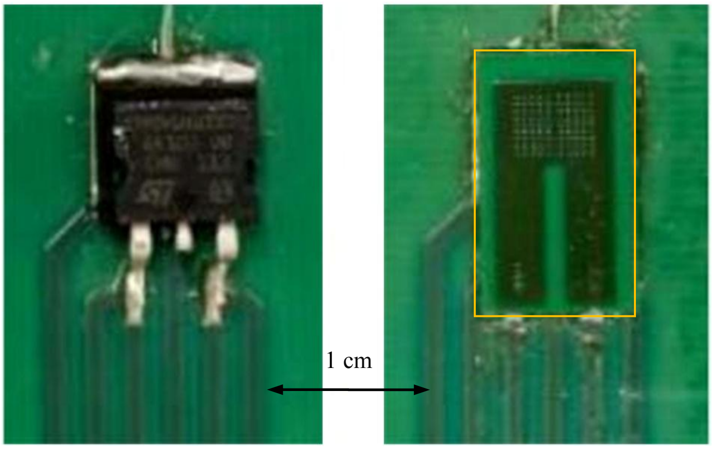

Figure 12.1. D²PAK (left) and the new 50 W demonstrator (right) samples soldered onto a reference printed circuit board.

The chosen environmental assessment method is the Sustainable Process Index (SPI) (Narodoslawsky & Krotscheck, 1995) as an example for highly aggregated assessment methods. The SPI is based on the assumption, that the only sustainability natural income to be used in a sustainable economy is solar radiation. This radiation on the earth's surface is converted into products or services. Therefore, the area as a limited resource is the measure for the SPI which makes it a member of the ecological footprint family of indicators. It measures the area necessary to embed a human activity sustainably in the ecosphere without changing its ability to convert the natural income of solar radiation into useful services. The Graz University of Technology provides an open SPI–software, which offers the opportunity to access a database and to create new case processes. For this study SPIonWeb (SPI, 2019) is used to calculate and compare the values of the SPI or "ecological footprint" of two different power component designs.

The functional unit is one sample of each technology as they are functionally identical. The life cycle inventory data are determined by mass and volume measurements, literature data, as well as material and geometry simulation data provided by project partners (EmPower, 2018).

12.3. Life Cycle Inventory and Sustainable Process Index

The measured and calculated material compositions of the 50 W power components are shown in table 12.1. The new demonstrator has a reduced mass by the factor 7.

Table 12.1. Determined material composition of the conventional D^{2}PAK (left) and the new power discrete package demonstrator (right).

Material composition of D^{2}PAK	
Copper	790 mg
Epoxy resin	620 mg
Silicon product	16 mg
Lead	8 mg
Aluminium	2 mg
Nickel	962 µg
Silver	206 µg
Tin	165 µg
Gold	24 µg
Titanium *)	14 µg
Sample weight	**1,4 g**

Material composition of 50 W demonstrator	
FR4	115 mg
Copper	66 mg
Silicone product	18 mg
Epoxy resin	6 mg
Nickel	5 mg
Gold	0,15 mg
Sample weight	**0,2 g**

*) Assumption for LCI-Data: TiO_2

Life cycle inventory data are determined including following data adjustments: There is no available dataset for titanium, therefore the dataset of TiO_2 is used. The life cycle inventory for TiO_2 is multiplied by the factor 1.67 to correspond to the included mass of titanium according to the atomic masses of titanium and oxygen. Although there are additional processing steps from TiO_2 to pure titanium, this is not further considered, because the mass contained in the sample and its impact is almost negligible for the total SPI result.

A further correction is made for using the SPI dataset for the FR4 printed circuit board material: The applied original dataset "*printed wiring board*" is based on a 6–layer PCB design with a specific mass of 3.26 kg/m^2 and a thickness of 1.6 mm (Ecoinvent, 2019; database version 3.4). The demonstrator consists of a thin multilayer FR4 PCB including the embedded component with an area of 1.5 cm^2 and a specific mass of about 0.8 kg/m^2, that is less than one quarter of the original dataset. The production steps of a printed circuit board depend on the mass, the area, and the number of layers. Therefore, a simple conversion of the area according to the reduced mass is not suitable. As a first approximation the LCI area value for the software input data is halved to 0.75 cm^2.

The "silicon product" represents the electronic core of the power component. Since it includes only the material, the process efforts are represented by the dataset "*transistor, auxilliaries and energy use*" as it is assumed to be most comparable with the processing of a power diode. These two processes are almost equal for both sample technologies. Nevertheless, it is included to give a more comprehensive picture of the footprint relations between the samples.

To make a fair comparison of the technologies, all production steps should be represented in a similar depth. However, one process remains ignored that is the manufacturing efforts for the liquid epoxy resin of the D^{2}PAK housing to a solid epoxy resin besides the material composition itself. There was no dataset found representing the input of hardeners and the curing process cycles. There is further no information on which basis the epoxy resin is made, such as glycerin or propylene, and what kind of additives are included, such as bisphenol A.

Tables 12.2. and 12.3. show the applied LCI data for both 50 W power component technologies, the used SPI datasets, the corresponding databases, and the respective SPI results. The input flow data represent the revised version of (Franz & Unger, 2016).

Table 12.2. Life cycle inventory data and SPI results for the commercial D^{2}PAK.

50 W Power Schottky Rectifier D^{2}PAK				
Material	**Mass [mg]**	**SPI dataset name**	**Database**	**a_{part} [m^2.a/Unit]**
Copper	90.30	Copper (new process)	"Ökoinventare von Energiesystemen" BEW (1996)	1.063
Package	20.00	Epoxy resin	ecoinvent	1.142
Microchip	16.05	Silicon, electronic grade, at plant	ecoinvent	0.819
	16.05	Transistor, auxilliaries and energy use		0.903
Lead	7.88	Lead 0% Recycling	"Ökoinventare von Energiesystemen" BEW (1996)	0.009
Aluminium	1.90	Aluminium 0% Recycling	"Ökoinventare von Energiesystemen" BEW (1996)	0.010
Nickel	0.96	Nickel	"Ökoinventare von Energiesystemen" BEW (1996)	0.015
Silver	0.21	Silver-gold mine operation with refinery	ecoinvent 3	0.155
Tin	0.17	Tin production	ecoinvent 3	0.001
Gold	0.02	Gold	ecoinvent 3	0.138
Titanium	0.02	Titanium dioxide production, sulfate process	ecoinvent 3	< 0.001
Total				**4.254**

Table 12.3. Life cycle inventory data and SPI results for the 50 W power module demonstrator.

50 W Power Discrete Package Demonstrator				
Material	**Mass [mg]**	**SPI dataset name**	**Database**	**a_{part} [m^2.a/Unit]**
Copper	65.86	Copper (new process)	"Ökoinventare von Energiesystemen" BEW (1996)	0.089
Package	6.16	Epoxy resin	Ecoinvent	0.011
Gold	0.15	Gold	ecoinvent 3	0.845
Nickel	4.70	Nickel	"Ökoinventare von Energiesystemen" BEW (1996)	0.072
FR4	0.75 cm^2	Printed wiring board, for surface mounting, Pb free surface	ecoinvent 3	10.017
Microchip	17.99	Silicone product, at plant	ecoinvent	0.916
	17.99	Transistor, auxilliaries and energy use		1.011
Total				**12.962**

Figure 12.2. and 12.3. show the results of the main contributors to the Sustainable Process Index of both samples. The total SPI results to 4.3 m^2.a/sample for the D^2PAK and 12.96 m^2.a/sample for the 50 W demonstrator in embedding technology. Concerning the commercial D^2PAK package, the microchip manufacturing (silicon and processing auxiliaries) has the largest share with 40.4 % followed by the liquid exopy resin (without processing) with 26.8 % and copper with 25 %. The new demonstrator shows an environmental hot spot in the embedding PCB manufacturing with a share of 77.3 % followed by the microchip (silicon and processing auxiliaries) with 14.9 %. The unconsidered processing of liquid epoxy resin will reduce the differences between the technologies in favour of the new demonstrator, but it is expected that the general trends will not be affected.

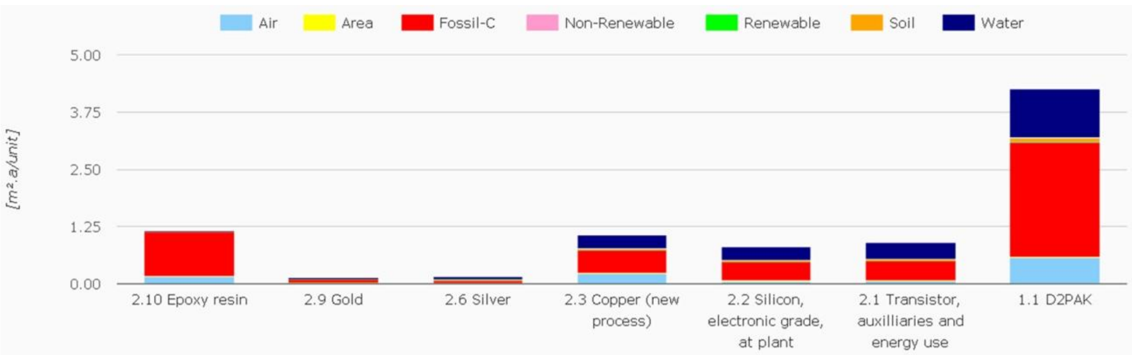

Figure 12.2. Environmental hot–spots and detailed SPI results of the D^2PAK package design.

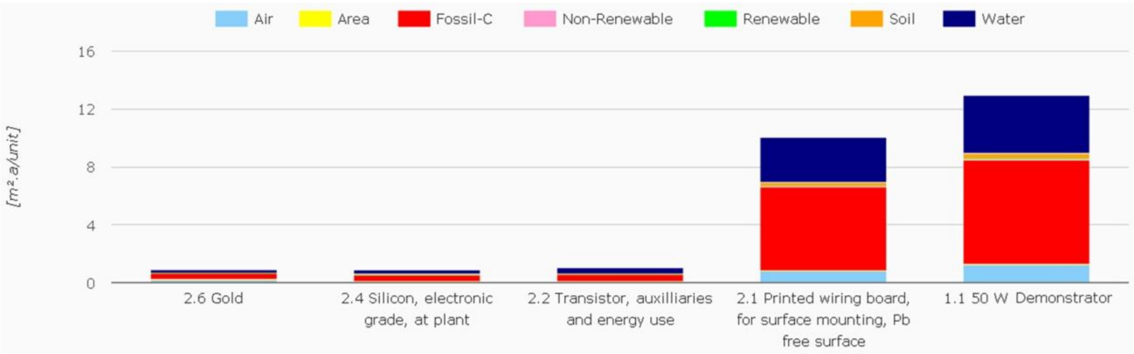

Figure 12.3. Environmental hot–spots and detailed SPI results of the new package demonstrator.

The results of the SPI are divided into seven different categories (see figure 12.3.): emission to air, area consumption, fossil carbon consumption, non–renewables consumption, renewables consumption, emissions in soil, and emissions in water.

The largest area consumption for all materials and processes concerns fossil carbon (red) with 58.9 % for the D^2PAK and 56.2 % for the new demonstrator, which is related to the energy consumption. A further environmental pressure results from emissions in water (dark blue). In case of the D^2PAK 25.3 % of the total area relates to water emissions caused by the manufacturing of copper and silicon, and process auxiliaries and energy. For the new demonstrator, 31.3 % of the total area relates to water emissions.

The overall environmental hot spots are the energy consumption of both technologies and the PCB production of the new demonstrator with an area contribution of 77 %. This leads to the recommendations firstly, to search for more energy efficient process parameters for the PCB production and secondly, to provide sustainable energy sources for the process steps.

The mass of the new demonstrator package amounts to 14 % of the D^2PAK package. Compared to that, in absolute values and within the system boundaries, the new demonstrator component has a worse ecological footprint than the commercial D^2PAK by at least the double, including an estimated generous share of the liquid epoxy resin processing.

The results show that the miniaturisation of the component does not necessarily reduce the environmental impact. Quite to the contrary, in this case the mass specific ecological footprint increases by the factor 15 to 20 for the new package.

12.4. Conclusion

This cradle–to–gate LCA study compared the Sustainable Process Index of two functionally equal 50 W power components with the focus on varying material compositions. The results for the commercial D^2PAK package design is 4.3 m^2.a/sample and for the new developed demonstrator fabricated in embedding technology is 13.0 m^2.a/sample within the system boundary. The samples have different masses, therefore, the mass specific ecological footprint of the new miniaturised power discrete package design is 20 times higher than the commercial D^2PAK package. The comparison is to show that the electronic components should not be arbitrarily interchangeable, but an application-specific consideration for environmental reasons is required. It further shows that the miniaturisation of components leads to a significant increase of the mass-dependent environmental impact which has to be considered using the limited choice of LCI data of commercial databases.

This study is performed with the focus on material data with low uncertainties. Consequently, there are process uncertainties concerning the microchip manufacturing and also bonding and assembly of the entire component. Further, the PCB manufacturing in embedding technology tends to require more energy than it is applied in the dataset, as it is shown in Chapter 14, Section 14.5. This might be already partly considered in the assumed LCI area of the PCB which has to be further investigated in detail. There is a reference in the ecoinvent dataset for SMD transistors (Ecoinvent, 2019), which could be comparable to some extent to a power diode, where three quarters of the calculated SPI relates to process auxiliaries and energy besides the material composition. Therefore, further research is recommended including more detailed process steps concerning both technologies and to make the results applicable for the scientific community.

Data uncertainties occure from possibly unknown auxiliary chemicals and process energy which could further increase the ecological footprint for both samples and might reduce the high difference of the assessment results. But it is expected that the determined trend to higher mass–specific environmental impact due to miniaturisation remains valid.

The Sustainable Process Index is a highly aggregated environmental assessment method which clearly shows environmental hot spots in the development phase of a product design. In the case of the new developed power discrete package in embedding technology the environmental pressure is along the gate–to–gate process of the printed circuit board manufacturing which is mainly related to the electrical energy consumption based on fossil energy sources. To reduce this impact it is recommended to find strategies for providing renewable energy sources for this process chain in the location of the mass production which is expected to be in Asia.

12.5. Acknowledgement

This study was supported by the Austrian Research Promotion Agency (Österreichische Forschungsförderungsgesellschaft–FFG) in the frame of the research project "EmPower" (Embedded Power components for electric vehicle applications) project No. 3665375.

12.6. References

Ecoinvent Database (2019). http://www.ecoinvent.org/ [04/2019].

EmPower (2018). Embedded Power components for electric vehicle applications-EmPower. Project funded by the Austrian Research Promotion Agency (FFG), project No. 3665375.

Franz, Manuela; Unger, Michael (2016). Ecological Aspects of a Novel Embedding Technology for Automotive Power Applications. IEEE 39th International Spring Seminar on Electronics Technology, Book of Abstracts, May 18-22, Pilsen, Czech Republic, University of West Bohemia in Pilsen, 2016. Extended Abstract; ISBN: 978-80-261-0618-0, S. 195 - 196.

Narodoslawsky M.; Krotscheck C. (1995). The sustainable process index (SPI): Evaluating processes according to environmental compatibility; J. of Hazardous Materials, 1995, 41 (2+3), 383-397.

SPI (2019). Sustainable Process Index. http://spionweb.tugraz.at/en/spi [04/2019].

Unger, M.; Nicolics, J.; Morianz, M.; Stahr, H.; Dosseul, F.; Manessis, D. (2015). Thermo-mechanical comparison of novel embedded high-power packages using FEM-simulations. Conf.-Proc. ISBN: 978-1-5108-0813-3, IMAPS Nordic 2015, Helsingor, June 8-9, 2015, pp. 126-133.

13. Sustainable Process Index of New Embedded Power Modules for e–Bikes and Automotive Application

Abstract: *Embedding technology enables miniaturisation of electric and electronic components together with improved reliability and thermal stability in harsh environment. This work investigates in the course of single case studies the environmental performance from cradle-to-gate of two newly designed and fabricated power modules for electric vehicle application applying the method of the Sustainable Process Index (SPI), a member of the "ecological footprint" family. The study addresses the application of highly aggregated life cycle assessment methods to identify environmental hot spots in the product design phase, to evaluate data requirements, and to show optimisation options. The SPI results to 461 m^2.a/sample for a 500 W power module demonstrator for e–bikes and 18,817 m^2.a/sample for a 50 kW power module demonstrator for automotive application. The life cycle impact assessment shows in case of the 500 W demonstrator, that the wafer manufacturing has the largest environmental pressure followed by the silver sinter paste, FR4 material, and copper. In case of the 50 kW demonstrator, the largest environmental impact is also caused by the wafer manufacturing followed by the silver sinter paste. These first results show, that there is further research needed to generate more precise life cycle inventory data for wafer fabrication, FR4 material processing, and the fabrication of metallisation sinter pastes including nanoparticles.*

13.1. Introduction

Electric circuits can be realised by different technologies. Most commonly used are printed circuit boards made of copper plated FR4 (flame retardant, class 4) compound materials populated with through–hole or surface mounted electronic components. Embedding technology is a further method to design electric circuits by incorporating and encapsulating thin shaped electronic components. Figure 13.1. shows the cross section of microchips embedded between several layers of FR4 material.

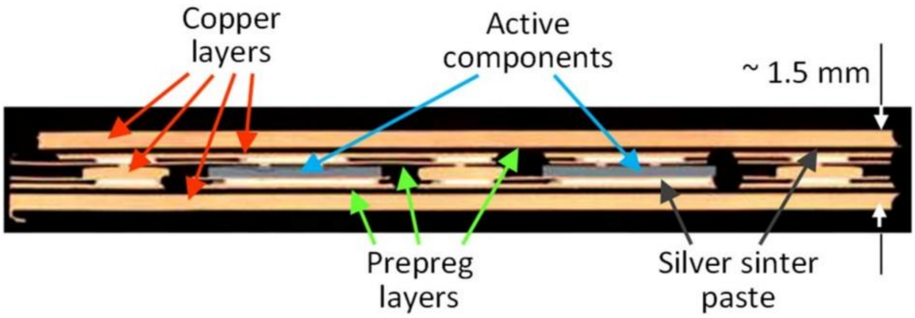

Figure 13.1. Principle of embedded components: image of the cross section of a power module. (Stahr et al., 2016b)

Embedding technology enables miniaturisation of electric and electronic components together with improved reliability and thermal stability in harsh environment. Besides miniaturisation and simultaneously increasing of the electrical performance, the design and packaging has to meet high environmental standards.

A newly designed 500 W power module demonstrator for e-bikes and a 50 kW power module demonstrator for automotive application were developed (Nicolics et al., 2017; Stahr et al., 2016a,c; Unger et al., 2016). The high–power control modules are applied as inverters for drive motors, battery chargers and electronic power controllers in auxiliaries such as power steering and brake systems.

This work provides a first environmental assessment of these new high–power modules, which are based on the embedding of thin MOSFET (metal–oxide–semiconductor field–effect transistor), IGBT (insulated–gate bipolar transistor), and diode chips into an enclosing package. The package is characterised by stacking different thin layers including prepregs, power core, metallisations, insulated metal substrates (IMS), and dielectric filling materials. A prepreg is an incomplete cured, flexible thin layer made of FR4 glass fibre reinforced epoxy resin in contrast to the cured rigid core material. The functional high power components (MOSFETs, IGBTs, diodes) are wafer fabricated. The innovation of the new packaging concept lies on double–sided cooling and vertical current flow to improve heat dissipation of the embedded power cores.

In the frame of the thesis, this study addresses the application of highly aggregated life cycle assessment methods to identify environmental hot spots in the product design phase. Optimisation options are determined and further data requirements are discussed.

13.2. Goal and Scope Definition

The goal of this study is to evaluate the environmental hot spots of the new power module demonstrators with 500 W and 50 kW by two independent case studies. Figure 13.2. and Figure 13.3. show the top views, the layout concepts, and the cross sections of the embedding layer stacks and of the wafer fabricated MOSFET, IGBT, and diode components, respectively. The power module parts, which are considered in the environmental assessment, are marked with a yellow line in the top views.

The environmental assessment is conducted as a streamlined life cycle assessment (SLCA). In contrast to a full life cycle assessment from cradle–to–grave, the SLCA shortens the procedure by targeting and limiting the goal of the study. In this case, the manufacturing stages of the two power modules, as described above, are evaluated by a cradle–to–gate LCA. The use phase, the e-waste management, transport, and packaging are not considered. The functional units are one sample of the respective power module.

The applied environmental assessment method is the Sustainable Process Index (SPI), which is provided by the Graz University of Technology, Austria. The method is based firstly, on the assumption that a sustainable economy builds only on solar radiation as natural income, and secondly, on the comparison of natural and anthropogenic material flows. It calculates the cumulative area [m^2] needed to embed the impact of all material and energy flows, as well as all produced emissions, sustainably into the biosphere (Narodoslawsky & Krotscheck, 1995). Life cycle inventory (LCI) data are taken from the databases of SPIonWeb (SPI, 2019), own measurements, and simulation raw data provided by project partners.

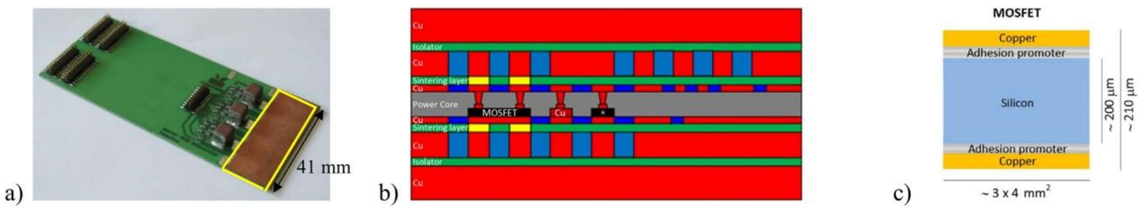

Figure 13.2. 500 W motor driver module demonstrator; a) top view, b) schematic cross sections of the power module part, and c) schematic multilayer set–up of the embedded MOSFETs (Boettcher et al., 2016; Stahr et al., 2016b).

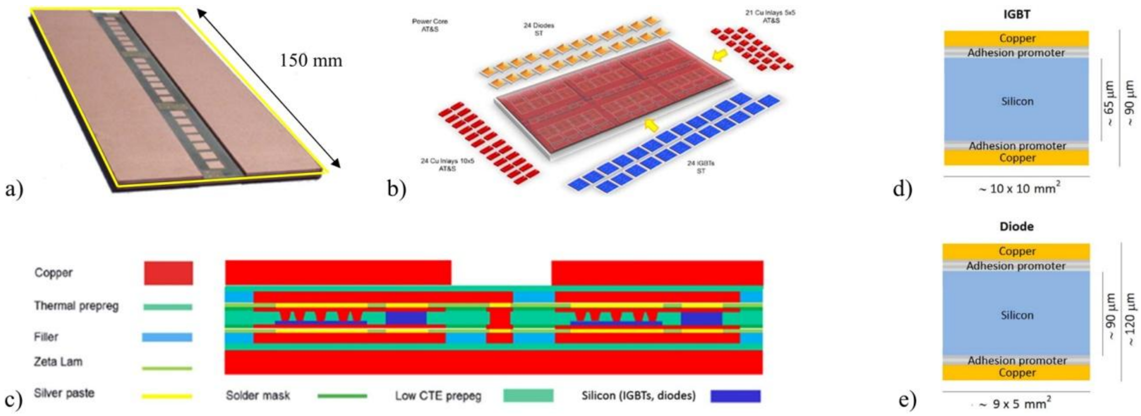

Figure 13.3. 50 kW demonstrator; a) top view of the power module, b) schematic illustration of power core, power semiconductors, and copper inlays, c) schematic cross sections of the power module (Boettcher et al., 2016), d) schematic multilayer set–up of the embedded IGBTs and e) of the embedded diodes.

13.3. Life Cycle Inventory (LCI)

The manufacturing process for the demonstrators includes three main steps, the chip manufacturing, the power core fabrication, and the power module fabrication, which is schematically shown in figure 13.4. The sub–processes are similar for all demonstrators.

The life cycle inventory consists of two parts. The first part schematically describes the flow of the power module manufacturing process, which is similar for all demonstrators. The second part includes detailed material composition data of the demonstrators on basis of simulation data. However, the manufacturing process data are taken from the SPI database, which are not necessarily identical with the actual flow–chart due to unavailable data. The assumptions are made with the best possible approximation to the real processes.

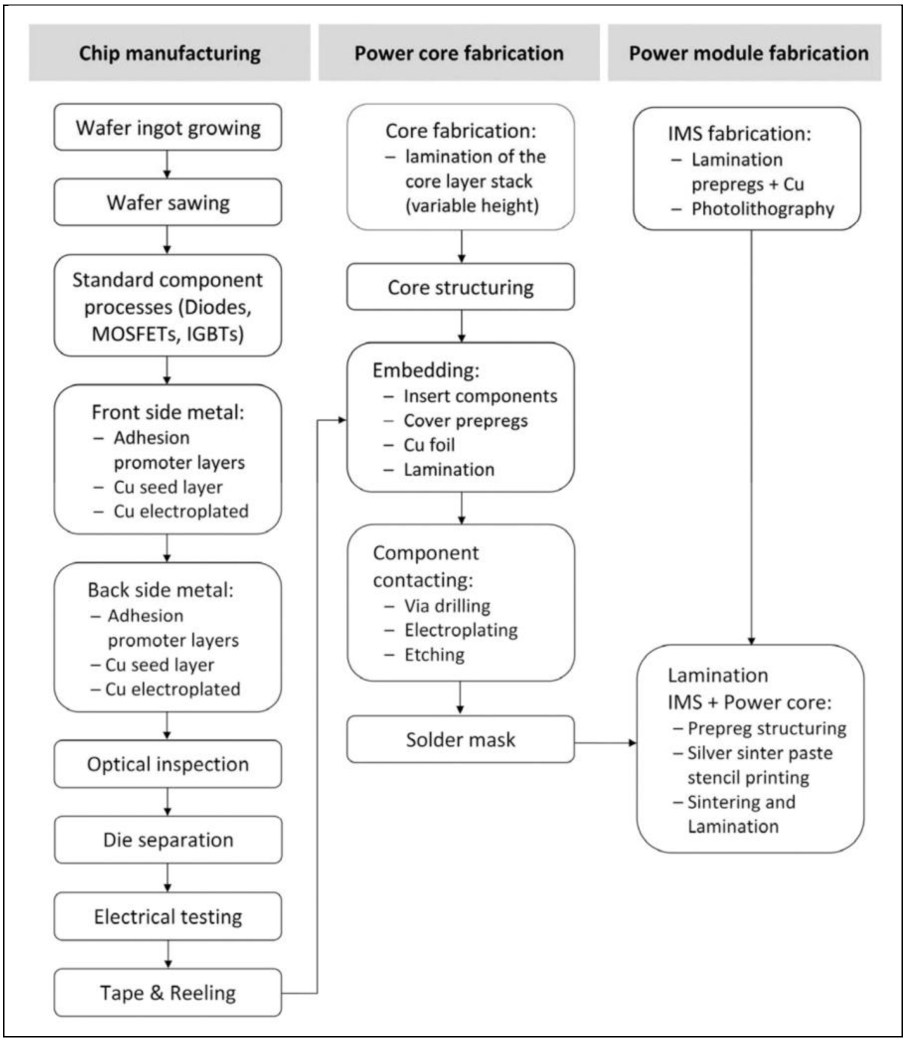

Figure 13.4. Process flow of Chip manufacturing (Doering & Nishi, 2008), Power core fabrication (Morianz & Stahr, 2015), and Power module fabrication (Manessis et al., 2015).

The SPI database requires material input data in the unit “kg” or in a few cases “m^2”. The material composition data based on simulation results are provided in terms of volume data. Table 13.1. shows the density conversion factors used for all materials in this study.

Table 13.1. Density of materials used in the power modules.

Material	**Density**	
Copper	8.94 g/cm³	(Römpp, 2019a)
FR4 (Prepreg / Core)	2.00 g/cm³	(Rotek, 2008)
Silicon	2.33 g/cm³	(Römpp, 2019b)
Silver	10.50 g/cm³	(Römpp, 2019c)

13.3.1. Life Cycle Inventory for the 500 W Demonstrator

The material composition and the corresponding datasets for the 500 W power module demonstrator is shown in Table 13.2. The functional electronic component is modelled by a “silicon product” and new estimations for wafer manufacturing. The adjustments of the datasets are following:

- The SPI database, which is based on the ecoinvent data for electronics, does not consider variable thicknesses of PCBs. The standard PCB design has a specific mass of 3.26 kg/m^2 and a thickness of 1.6 mm. For the 500 W demonstrator the dataset: “Printed wiring board, forsurface mounting, Pb free surface”, is applied, which has the input unit of m^2. As a first approximation, the thin FR4 material of the demonstrator with a thickness of 75 µm is converted by mass to a standard 1.6 mm thick material.
- The embedded MOSFET components are described by a silicon product, fabricated similar to the wafer and semiconductor manufacturing procedure of a LED-dice, which is described in detail in a publication of the U.S. Department of Energy (U.S. DOE, 2012). According to discussions with technicians working in this field, as first approximations, the dataset values are reduced by 50 % assuming less production efforts at larger wafer diameters for the demonstrator components than for a LED-dice.
- The silver sintered paste is composed of a nano-silver matrix, which is assumed to contain 50 wt% silver. The organic matrix is not considered due to unavailable data.

13.3.2. Life Cycle Inventory for the 50 kW Demonstrator

The material composition and the corresponding datasets for the 50 kW demonstrator power module is shown in table 13.3. The functional electronic components, 24 diodes and 24 IGBTs, and the other materials are modelled by following the same approach as described for the 500 W demonstrator in section 13.3.1.

Table 13.2. Material composition and corresponding datasets for the 500 W demonstrator.

500 W Power Module Demonstrator			
Material Composition	***Volume***	***SPI Database: Applied Datasets***	***LCI Data***
Copper	2088 mm^3	Copper	18.63 g
Printed Circuit Board FR4 (75 μm)	2036 mm^3	Printed wiring board, for surface mounting, Pb free surface (converted to 1.6mm)	12.72 cm^2
MOSFET 6x	6x12.3 mm^3	Silicon product	28.7 mg
MOSFET 6x	6x12.56 mm^2	Wafer/semiconductor manufacturing (50% area)	37.7 mm^2
Silver sintered paste (metallisation)	14.4 mm^3	Silver (50% due to nano–silver matrix)	75.53 mg

Table 13.3. Material composition and corresponding datasets for the 50 kW demonstrator.

50 kW Power Module Demonstrator			
Material Composition	***Volume***	***SPI Database: Applied Datasets***	***LCI Data***
Copper	24262 mm^3	Copper	216.4 g
Prepreg FR4 (100 μm)	1279 mm^3	Printed wiring board, for surface mounting, Pb free surface (converted to 1.6mm)	7.99 cm^2
Core FR4 (156 μm)	1989 mm^3	Printed wiring board, for surface mounting, Pb free surface (converted to 1.6mm)	12.43 cm^2
Prepreg FR4 (100 μm)	1279 mm^3	Printed wiring board, for surface mounting, Pb free surface (converted to 1.6mm)	7.99 cm^2
Diodes 24x (sputtered)	116 mm^3	Silicon product	272 mg
Diodes 24x (sputtered)	24 x 51 mm^2	Wafer/semiconductor manufacturing (50% area)	612 mm^2
IGBT 24x (sputtered)	229 mm^3	Silicon product	535 mg
IGBT 24x (sputtered)	24x99 mm^2	Wafer/semiconductor manufacturing (50% area)	1188 mm^2
Silver sintered paste (metallisation)	700 mm^3	Silver (50% due to nano–silver matrix)	3.67 g

13.4. Life Cycle Impact Assessment (LCIA)

The results of the life cycle impact assessment show that for all demonstrators the wafer manufacturing of the functional components has the highest environmental impact. The partial SPI area for fossil carbon use (Fossil–C) generates the largest footprint, followed by the area for emissions to water.

13.4.1. LCIA Results of the 500 W Power Module Demonstrator

The 500 W power module has a total SPI of 461 m^2.a/sample. Figure 13.5. shows the results in detail. The wafer manufacturing consumes about 6 times more area than the silver and the FR4 material, and 13 times more area than the copper parts. The silicon product without wafer manufacturing has a SPI share less than 1 ‰.

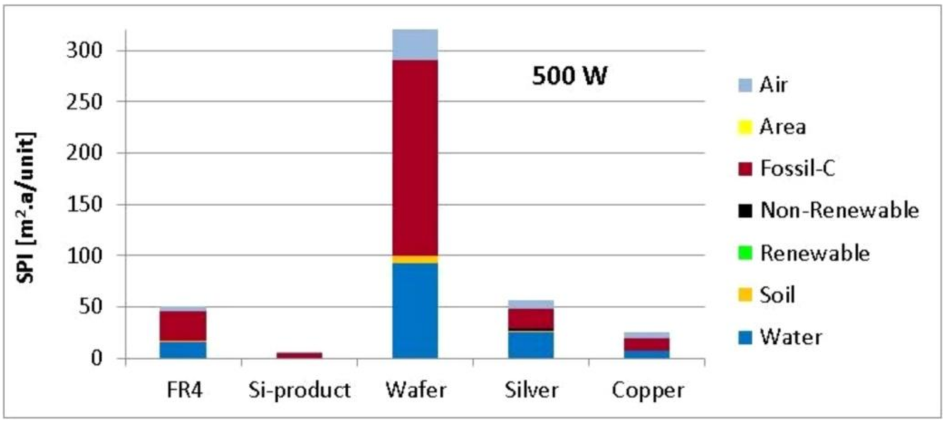

Figure 13.5. SPI results for the 500 W demonstrator.

13.4.2. LCIA Results of the 50 kW Power Module Demonstrator

The SPI results for the 50 kW demonstrator is shown in Figure 13.6. The total SPI is 18,817 m^2.a/sample. The wafer manufacturing consumes 5.6 times more area than silver. Copper has a total SPI share of 1.5 % and the FR4 material has a total SPI share of 0.6 %.

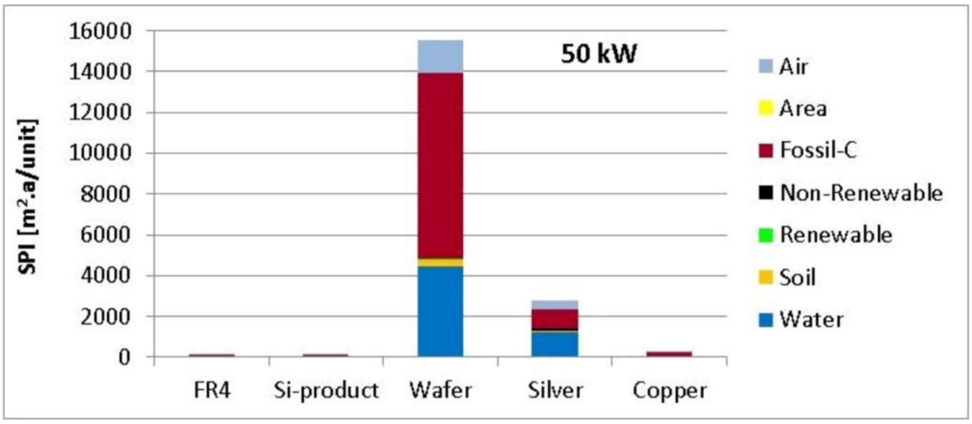

Figure 13.6. SPI results for the 50 kW demonstrator.

13.5. Interpretation

The SPI results are the first evaluation of the environmental impact of the embedding technology. All investigated samples show the highest environmental impact in the stage of wafer fabrication. Due to inapplicable electronic grade wafer data, the wafer manufacturing dataset is generated on basis of LED wafer production including data for raw materials, metals, chemicals, and electricity consumption (U.S. DOE, 2012). Within this dataset, the highest SPI impact with 70 % has the electical energy consumption followed by liquid nitrogen with 9 %. The applied dataset for the electical energy mix is “Net electricity USA_2009, low voltage” according to the country of origin of the LED-study.

These relations were the basis to use the data as a first approach for the wafer production of the demonstrators. Despite of data uncertainties it can be expected, that the tendency of the highest impact is correct. The often used dataset “silicon product” for semiconductor materials shows, that this dataset alone does not represent the real manufacturing efforts, as can be seen in Figure 13.5. and Figure 13.6.

A further important environmental pressure shows the FR4 prepreg and core material. The ecoinvent database includes a number of different datasets for PCBs with large impact variations. They are all related to standard boards with 1.6 mm thickness and a specific mass of 3.26 kg/m^2. There is a high data uncertainty for modelling the thin prepreg and core materials, metallisations, as well as structuring and lamination procedures. As evaluated in Chapter 14, Section 14.5., it is expected, that the electrical energy consumption for structuring the PCB layers have to be significantly increased.

Besides the wafer fabrication, the 500 W demonstrator shows high environmental pressures for the silver paste, the FR4 materials, and copper. In contrast, however, the 50 kW demonstrator shows a high environmental impact for silver. Since the silver sinter paste consists of silver nanoparticles, it can be expected that the actual manufacturing impact for nanomaterials will be much higher for all demonstrators.

13.6. Conclusion

New electronic components for harsh environment and high reliability usually base on a limited number of applicable materials due to technological reasons. The two new developed powermodules have a functional counterpart made in conventional technology including a comparable wafer fabricated component. Therefore, environmental assessment is indispensably to identify the environmental performance of a new product in an early stage as a basis for monitoring existing and controlling further technological developments.

This study shows that the electronic chips have by far the highest environmental impact due to the high impact of the energy intensive wafer fabrication. To reduce the environmental burden, the energy supply, that is the consumed energy mix, is of importance. The question, for example, to produce an electronic product in Europe or in an Asian country is not only a question of production costs but also a question of environmental costs. For example, the SPI values of the average electrical energy mixes of Europe, USA, and China have the relations 1:1.2:1.3.

There are following open research questions to get a better understanding of the environmental performance of the new demonstrators:

- New measurements of the energy consumption for wafer and FR4 material manufacturing and the related ancillary products
- Comparative assessments for the 500 W and 50 kW demonstrators with their relating conventional technologies to collect data for environmental trends in the course of the technology changes
- New manufacturing data for the silver paste and particularly nanoparticles
- Applying other assessment methods such as Ecoindicator 99 (Baumann & Tillman, 2004), ReCiPe (ILCD, 2010), and different midpoint impact categories to get a broader overview on the environmental performance for the market ready product

13.7. Acknowledgement

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13.8. References

Baumann, H.; Tillman, A. M. (2004). The Hitch Hiker's Guide to LCA: An Orientation in Life Cycle Assessment Methodology and Application. Studentlitteratur, Lund, 2004.

Boettcher, L.; Karaszkiewicz, S.; Manessis, D.; Ostmann, A. (2016). Embedded Die Packages and Modules for Power Electronics Applications. IMAPS 12th Int. Conf. and Exhibition on Device Packaging, Fountain Hills, Arizona USA, 14.-17. March, 2016.

Doering, R.; Nishi, Y. (2008). Handbook of Semiconductor Manufacturing Technology. 2nd Edition, CRC Press, 2008.

ILCD International Reference Life Cycle Data System Handbook (2010). Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment. European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy.

Manessis, D.; Boettcher, L.; Ostmann, A.; Lang, K.-D.; Whalley, S. (2015). Development of Advanced Power Modules for electric vehicle applications. EMPC 2015 European Micro-electronics Packaging Conference, September 2015, Friedrichshafen, Germany.

Morianz, M.; Stahr, H. (2015). Embedded Power Electronics on the way to be launched. EMPC 2011, Europ. Microelectronics Packaging Conf., Sept. 2015, Friedrichshafen, Germany.

Narodoslawsky, M.; Krotscheck, C. (1995). The sustainable process index (SPI): Evaluating processes according to environmental compatibility; J. of Hazardous Materials, 1995, 41 (2+3), 383-397.

Nicolics, J.; Unger, M.; Groß, S.; Morianz, M.; Stahr H. (2017). Thermal characterization of a Pedelec 500W motor driver module with embedded power transistors. IEEE Proc. of ISSE 2017, May 11-14, 2017, Sofia, ISBN 978-1-5386-0582-0.

Römpp Enzyklopädie Online (2019a), Stuttgart: Thieme Verl., URL: http://www.roempp.com [04/2019]. Abschnitt: Kupfer.

Römpp Enzyklopädie Online (2019b), Stuttgart: Thieme Verl., URL: http://www.roempp.com [04/2019]. Abschnitt: Silicium.

Römpp Enzyklopädie Online (2019c), Stuttgart: Thieme Verl., URL: http://www.roempp.com [04/2019]. Abschnitt: Silber.

Rotek Handels GmbH (2008). Glasfaser-Hartgewebe HGW2372.1 (FR4-HF). DE V1.1 Stand 08-2008.

SPI (2019). Sustainable Process Index. http://spionweb.tugraz.at/en/spi [04.2019].

Stahr, H.; Morianz, M.; Gross, S.; Unger, M.; Nicolics, J. (2016a). Investigation of a power module with double sided cooling using a new concept for chip embedding. ISBN 978-3-8007-4171-7, Proc. of 9th Int. Conf. on Integrated Power Electronics Systems, CIPS 2016, March 8-10, 2016, Nürnberg, Germany, pp. 1-10.

Stahr, H.; Morianz, M.; Gross, S.; Unger, M.; Nicolics, J.; Böttcher; L. (2016b). Investigation of a power module with double sided cooling using a new concept for chip embedding. 9th International Conference on Integrated Power Electronics Systems, CIPS 2016, Proceedings, March, 8 - 10, 2016, Nuremberg/Germany.

Stahr, H.; Morianz, M.; Gross, S.; Unger, M.; Nicolics, J.; Böttcher, L. (2016c). Thermal benchmark of a classic and novel embedded high-power 3-phase Inverter bridge. Proc. of 6th Electronics System-Integration Technology Conference, ESTC 2016, September 13 – 16, 2016, World Trade Center, Grenoble, France, pp. 1-6.

Unger, M.; Nicolics, J.; Stahr, H.; Morianz, M.; Dosseul, F. (2016). Thermal Performance of Novel Diode Packages. 39th Int. Spring Seminar on Electronics Technology, Pilsen, Czech Republic; 18.–22.05.2016; in: “Extended Abstracts ISSE 2016”, (2016), ISBN: 978-1-5090-1389-0; p. 65 - 66.

U.S. Department of Energy (2012). Life-Cycle Assessment of Energy and Environmental Impacts of LED Lighting Products. Part II, 2012.

14. Environmental Assessment for a Newly Developed Camera System for Automotive Application

Abstract: *Embedding technology is an advanced electronic packaging method which allows to assemble electronic components not only on the surface but also into the inner volume of the printed circuit board material. The resulting higher packing density of components leads to a miniaturisation of the package and reduces the expenditure on materials. This work compares the environmental performance of a conventional and a newly developed rear camera system for automotive application. The attributional life cycle assessment method is ReCiPe Midpoint (H), 2014, based on the ecoinvent 3.3 database. The considered characterisation categories are climate change (CC), human toxicity (HT), and terrestrial ecotoxicity (TE). The study addresses the application of midpoint life cycle assessment methods and discusses different environmental impact categories and process data uncertainties. The new camera design shows in all characterisation categories a reduction of the environmental impact between 55 % and 60 % within the defined system boundaries based on material composition data. Further study results show, that hitherto not considered additional process energy data due to miniaturisation must not be disregarded.*

14.1. Introduction

Advanced electronics are characterised by miniaturisation and simultaneously increased functionality and reliability. In particular, the automotive industry requires electronic packages which are suitable for harsh environment such as high temperatures, humidity, or vibrations.

An innovative technology allows the embedding of electronic components into the printed circuit board (PCB). In this way, the packing density can be significantly increased due to the assembly of active and passive electronic components not only on the surface of both sides of a PCB but also into the inner volume of the PCB itself. Figure 14.1 shows a typical cross–section of the embedded components.

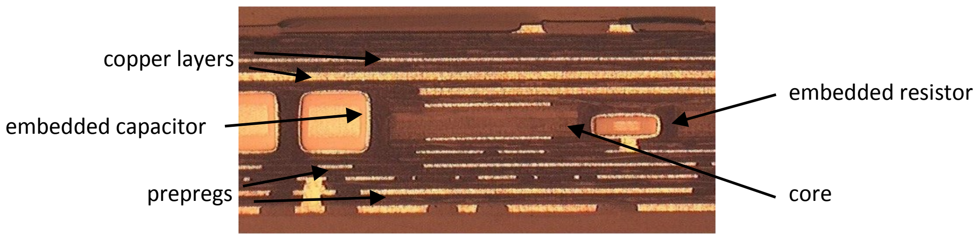

Figure 14.1. Cross section of embedded components in the PCB module of the camera.

The embedding of the components into the printed circuit board structure further enhances the electrical properties and the mechanical stability of the system.

In the course of this project, the described packaging concept is implemented in a new intelligent camera demonstrator system for automotive application. Based on a previous conventional model, the new demonstrator camera was manufactured in embedding technology following a similar camera design.

The aim of this work is to investigate the environmental performance of the new demonstrator camera compared to the conventional system. Figure 14.2 shows the images of the conventional and the new demonstrator camera systems, which are considered for the environmental assessment.

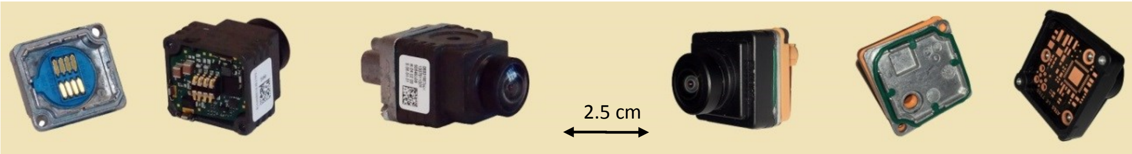

Figure 14.2. Image of the conventional (left) and new (right) demonstrator camera systems. In this case, the printed circuit board of the new system is not yet SMD assembled.

Basic life cycle inventory (LCI) data for electronic components are prepared by the ecoinvent database (Ecoinvent, 2018). There was no literature found about the life cycle assessment for embedding technology, but one previous study, conducted by the author of this work, which investigated the sustainable process index (SPI) for a power module based on embedding technology for automotive application (Franz et al., 2017).

In the frame of the thesis this work addresses the application of midpoint assessment methods and thereof three different characterisation categories, climate change, human toxicity and terrestrial ecotoxicity. Furthermore, data uncertainties and the limitation of material composition based life cycle inventory data are discussed.

14.2. Goal and Scope

A comparative environmental assessment for a conventional and a new demonstrator camera is carried out following the standard for life cycle assessment ISO 14040 and ISO 14044 (ISO, 2006a; ISO, 2006b). The system boundary is the cradle–to–gate manufacturing of the camera samples from raw material acquisition to the assembly of the final camera system and demonstrator, respectively. Samples of both variants are available in disassembled form. Life cycle inventory data are taken from weight and dimension measurements, and material labelling of the samples, data sheets, manufacturer data, literature data, simulation data, own estimations, and the life cycle inventory data of the ecoinvent database (Ecoinvent, 2018). The mass of all parts are weighed with a digital precision balance DENVER INSTRUMENT SI-234A.

The functional unit is one sample of the assembled camera of each type with the assumption of the same lifetime. The life cycle impact assessment method is ReCiPe Midpoint (H), v1.11, December 2014. ReCiPe is currently the most recommended LCIA method and builds on the Ecoindicator 99 and the CML method (Goedkoop, 2013). The considered characterisation categories include climate change (CC), human toxicity (HT), and terrestrial ecotoxicity (TE). The used software is openLCA 1.6.3. (openLCA, 2018). The used database is ecoinvent 3.3 (Ecoinvent, 2018). All flows and impacts are 100 % allocated to the camera systems. A sensitivity analysis is done by variation of material and process parameters of the electronic part.

The system model for the environmental assessment consists of three main parts, the housing, the electronic part, and the optics. Figure 14.3 shows the system model and the system boundary for the environmental assessment.

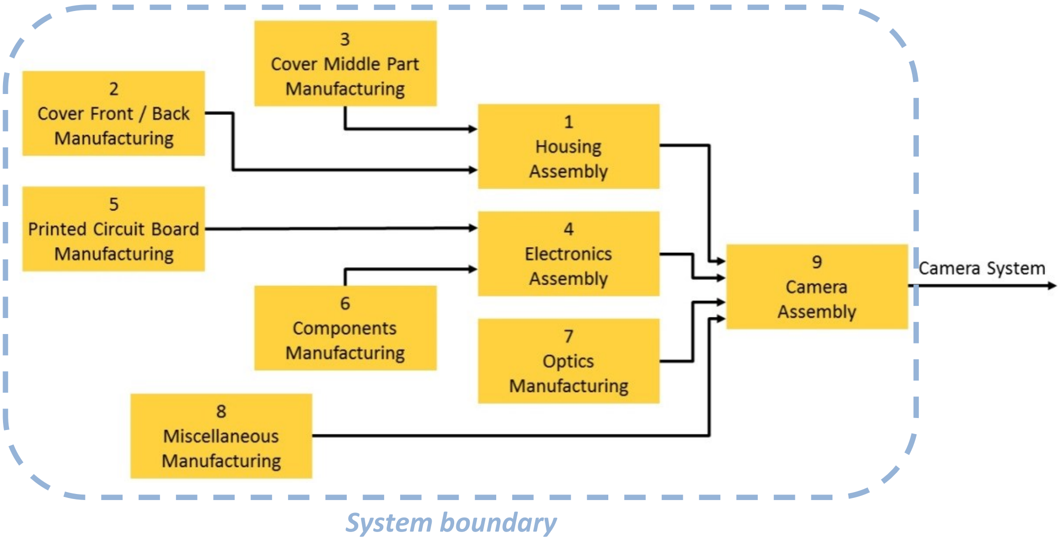

Figure 14.3. System model for both, the conventional camera and the new demonstrator camera.

The considered parts of the disassembled cameras are shown in figure 14.2, including all components for the activities 1 to 9 of figure 14.3.

14.3. Life Cycle Inventory

The two camera systems are similar in their design. In case of the conventional camera system, the electronic part consists of a stack of three multilayer FR4 PCBs populated with surface mounted devices (SMDs). The new demonstrator system consists of one multilayer printed circuit board with embedded and surface mounted components. Accordingly, the housing of the new camera system is smaller.

14.3.1. Life Cycle Inventory of the Conventional Camera System

14.3.1.1. Housing

The material of the front cover consists of an aluminium-silicon cast alloy AlSi12 with a mass ratio of 82 % aluminium and 12 % silicon. The back cover consists of an aluminium-silicon alloy AlSi12(Fe) and a plug connection made of brass with a galvanised gold surface, which is embedded in plastics PBT-GF10. The material composition of the alloy is estimated with 12 % silicon, 1 % iron, and 87 % aluminium (VAR, 2018).

The plastic part of the back cover consists of 90 % polybutylene terephthalate (PBT) and 10 % glass fibre. The plastic part is estimated with a volume of 0.25 cm^3 and an average density of 1.5 g/cm^3 (PBT: 1.4 g/cm^3 (Römpp, 2019a), glass fibre: 2.58 g/cm^3 (Wallenberger & Bingham, p. 211)). There is no dataset for PBT available, therefore, the dataset of PET granulate production is applied.

The connector pins are estimated to have a volume of 0.035 cm^3. With a brass density of 8.8 g/cm^3 (Deutsches Kupferinstitut, 2005) the mass amounts to 0.31 g. The gold surface has an estimated area of 1.5 cm^2 with a thickness of 0.1 µm. With a density of 19.32 g/cm^3 (Römpp, 2019b) the mass of gold amounts to 0.3 mg. Table 14.1. shows the applied LCI data for the housing.

Table 14.1. Life cycle inventory data for the housing of the conventional camera system.

Ecoinvent dataset name	Mass
market for aluminium, cast alloy – GLO	16.246 g
silicon production, metallurgical grade – RoW	2.226 g
cast iron production – RER	0.077 g
polyethylene terephthalate production, granulate, amorphous – RER	0.338 g
glass fibre production – RER	0.037 g
injection moulding – RER	0.375 g
market for brass – CH	0.310 g
market for gold – GLO	0.3 mg

14.3.1.2. Electronics

The electronic part of the conventional camera system consists of a stack of three double sided SMD mounted printed circuit boards (PCB). The components are considered as an average of SMD components according to the ecoinvent database.

Two scenarios are calculated and compared: firstly, the entire PCB, surface mounted, is calculated on kg-basis according to the average ecoinvent dataset.

Secondly, the un-mounted printed circuit boards and the components are calculated separately. Each PCB is assumed to consist of a 6 layer FR4 material with a measured height of 1.6 mm. The PCB areas are measured and the corresponding mass share is calculated according to the ecoinvent dataset with a density of 3.26 kg/m^2.

The components have a calculated mass of 4.845 g. The mass of the ICs and the electric connectors are calculated by volume measurement and the corresponding density of 2 g/cm^3, which was estimated by comparable volume and mass measurements of comparable SMD ICs and connectors. Table 14.2. shows the applied LCI data for the electronics.

Table 14.2. Life cycle inventory data for the electronics of the conventional camera system.

Ecoinvent dataset name	Mass / area
printed wiring board production, surface mounted, unspecified, Pb free – GLO	9.034 g
printed wiring board production, for surface mounting, Pb free surface – GLO	1285 mm^2
mounting, surface mount technology, Pb-free solder – GLO	1285 mm^2
market for capacitor, for surface-mounting – GLO	1.119 g
market for diode, glass-, for surface-mounting – GLO	0.135 g
market for electric connector, peripheral component interconnect buss – GLO	1.400 g
market for integrated circuit, logic type – GLO	1.000 g
market for light emitting diode – GLO	0.036 g
market for resistor, surface-mounted – GLO	0.803 g
market for transistor, surface-mounted – GLO	0.352 g

14.3.1.3. Optics

There is no information about the material composition of the optics. It is assumed, that the optical lens and housing have a mass ratio of 1:1. The housing material is assumed to be the same as the other housing materials AlSi12. The ecoinvent database does not include applicable data. Therefore, as a first approximation, data for coated flat glass, which is calculated with twice the mass to compensate expected higher production efforts, is applied. Table 14.3. shows the applied LCI data for the optics.

Table 14.3. Life cycle inventory data for the optics of the conventional camera system.

Ecoinvent dataset name	Mass
market for aluminium, cast alloy – GLO	2.186 g
silicon production, metallurgical grade – RoW	0.243 g
flat glass production, coated – RER	4.858 g
adhesive production, for metal – DE	1.000 g

14.3.1.4. Miscellaneous

The miscellaneous parts are needed for the camera assembly and include screws for mounting the PCB–stack into the housing and screwing together the housing, clips to fix the PCB-stack, a sealing strip between the cover parts, and adhesive for fixing the electric socket and the optics.

The screws are estimated to be made of steel. The clip is made of CuNi18Zn27. The seal is estimated to be made of silicone. The mass of the required adhesive is estimated to 1 g. Table 14.4. shows the applied LCI data for the miscellaneous parts.

Table 14.4. Life cycle inventory data for the miscellaneous parts of the conventional camera system.

Ecoinvent dataset name	Mass
market for steel, chromium steel 18/8 – GLO	1.000 g
wire drawing, steel – RER	1.000 g
market for copper – GLO	0.505 g
market for nickel, 99.5% – GLO	0.165 g
market for zinc – GLO	0.248 g
sheet rolling, copper – RER	0.919 g
silicone product production – RER	0.131 g
adhesive production, for metal – DE	1.000 g

14.3.2. New Demonstrator Camera System

14.3.2.1. Housing

The front cover and the middle part consist of an aluminium-silicon cast alloy AlSi12. The back cover consists of plastics made of PA66 GF50 and it is manufactured by injection moulding. Table 14.5. shows the applied LCI data for the housing.

Table 14.5. Life cycle inventory data for the housing of the new camera system.

Ecoinvent dataset name	Mass
market for aluminium, cast alloy – GLO	8.553 g
silicon production, metallurgical grade – RoW	1.252 g
glass fibre reinforced plastic production, polyamide, injection moulded – RER	3.529 g

14.3.2.2. Electronics

The PCB consists of 23 layers of FR4 material and copper. The volume of the FR4 material is 312 mm^3, and that of the copper material is 103 mm^3. The masses are calculated by the corresponding densities of FR4 with 2 g/cm^3 (Rotek, 2008) and copper with 8.94 g/cm^3 (Römpp, 2019c).

To determine the life cycle inventory data, the manufactured surface of the PCB layer stack before cutting is considered, which is the rectangular outer area plus 1 mm cutting losses.

Three scenarios for manufacturing of the PCB are compared:

1. Original dataset for printed wiring board production, for surface mounting on m^2-basis
2. Original dataset and mass corrected area of the PCB due to the thinner PCB material
3. Original dataset and selected change of dataset input flows concerning copper, glass fibre, and epoxy resin

The mounting activity remains unaffected by the variations of the area. The life cycle inventory data are determined as following:

1. $A_{demo} = 6.76\ cm^2$
2. PCB mass dataset: 3.26 kg/m^2; demonstrator PCB: 3.21 kg/m^2 → mass corrected area for LCI: $A_c = 519\ mm^2 \times 0.985 = 511\ mm^2$. Compared to the height relation of 1:1.6 of the dataset value and demonstrator PCB, the density of the demonstrator PCB must be higher than the considered sample in the dataset. The area difference is less than 5 % and therefore this scenario is not further considered.
3. It is assumed that the FR4 material consists of epoxy resin and glass fibre in the mass ratio of 1:1. The original dataset for PCB for surface mounting includes amongst others the input flows per m^2 which are corrected by the calculated mass data for the demonstrator PCB according to table 14.6. The corrections were directly implemented in the newly created dataset "modified" and they are not further visible in table 14.7.

Table 14.6. Modified input flows of the dataset for printed wiring board production.

Material/Activity	***Original data***	***Corrected data (incl. 10% waste)***
Copper	2.875 kg/m^2	1.948 kg/m^2
Sheet rolling, copper	1.817 kg/m^2	1.948 kg/m^2
Glass fibre reinforced plastic, PE resin	1.890 kg/m^2	–
Epoxy resin, liquid	–	0.833 kg/m^2
Glass fibre	–	0.661 kg/m^2

Table 14.7. Life cycle inventory data for the PCB of the new camera system. Version 1 includes the original dataset and version 3 the new created modified dataset according to table 14.6.

Ecoinvent dataset name	**Area**
1. printed wiring board production, for surface mounting, Pb free surface – GLO	6.76 cm^2
mounting, surface mount technology, Pb-free solder – GLO	6.76 cm^2
3. printed wiring board production, for surface mounting, Pb free surface, ***modified*** – GLO	6.76 cm^2
mounting, surface mount technology, Pb-free solder – GLO	6.76 cm^2

The mounted PCB includes 139 components and thereof 70 are assembled as embedded components. The dimensions of the components are taken from the respective material datasheets. To calculate the mass of the components, the following first assumptions for the density are made: passive components are based on aluminium oxide and therefore, their density is assumed with 4 g/cm^3 (Römpp, 2019d). Inductors are composed of ferrites and (less) copper. Including the plastic packaging, their density is assumed with 5.5 g/cm^3 (Amidon, 2019; Römpp, 2019c). The integrated circuit components are based on silicon and therefore, their density is assumed with 2.33 g/cm^3 (Römpp, 2019e). The SMD ICs have a density of 2 g/cm^3 according to section 14.3.1.2. Table 14.8. shows the applied LCI data for the electronic components.

Table 14.8. Life cycle inventory data for the components of the new camera system.

Ecoinvent dataset name	Mass
market for capacitor, for surface-mounting – GLO	247 mg
market for diode, glass-, for surface-mounting – GLO	135 mg
inductor production, miniature radio frequency chip – GLO	391 mg
market for resistor, surface-mounted – GLO	13 mg
market for integrated circuit, logic type – GLO	385 mg
market for electric connector, peripheral component interconnect buss – GLO	1.400 g

14.3.2.3. Optics

The camera optics of the new demonstrator system is approximately the same as for the conventional camera and therefore, the same LCI data are applied. Table 14.9. shows the applied LCI data.

Table 14.9. Life cycle inventory data for the optics of the new camera system.

Ecoinvent dataset name	Mass
market for aluminium, cast alloy – GLO	2.186 g
silicon production, metallurgical grade – RoW	0.243 g
flat glass production, coated – RER	4.858 g
adhesive production, for metal – DE	1.000 g

14.3.2.4. Miscellaneous

The masses of the miscellaneous parts are weighted and the materials are estimated similar to the conventional camera. Table 14.10. shows the applied LCI data for the miscellaneous parts.

Table 14.10. Life cycle inventory data for the miscellaneous parts of the new camera system.

Ecoinvent dataset name	Mass
market for steel, chromium steel 18/8 – GLO	0.800 g
wire drawing, steel – RER	0.800 g
silicone product production – RER	0.132 g
adhesive production, for metal – DE	0.500 g

14.4. Life Cycle Impact Assessment

Three characterisation categories are considered, Climate change (CC), Human toxicity (HT), and Terrestrial ecotoxicity (TE). The following sections present the comparative life cycle impact assessment results of the two camera systems in numerical and relative values. At first, the total results of the assembled camera are presented, followed by detailed results for the activities 1 to 9 according to figure 14.3.

14.4.1. Total LCIA Results of the Conventional and the New Demonstrator Camera

The total environmental impact results for the conventional and the new demonstrator camera systems are presented in table 14.11 and in figure 14.4, respectively.

Table 14.11. Comparison of the total results of the conventional and the new camera system in numerical values.

Σ Activities 1 – 9	**CC**	**HT**	**TE**
	kg CO_2 eq	kg 1,4-DB eq	kg 1,4-DB eq
Conventional camera	1.850	18.50	5.18E-04
New camera	0.848	7.38	2.34E-04

The new demonstrator camera system shows in all characterisation categories a reduction of the environmental impact between 55 % and 60 %. In comparison, the mass reduction of the new camera system amounts to 31 % (conventional camera: 35 g; new camera: 24 g).

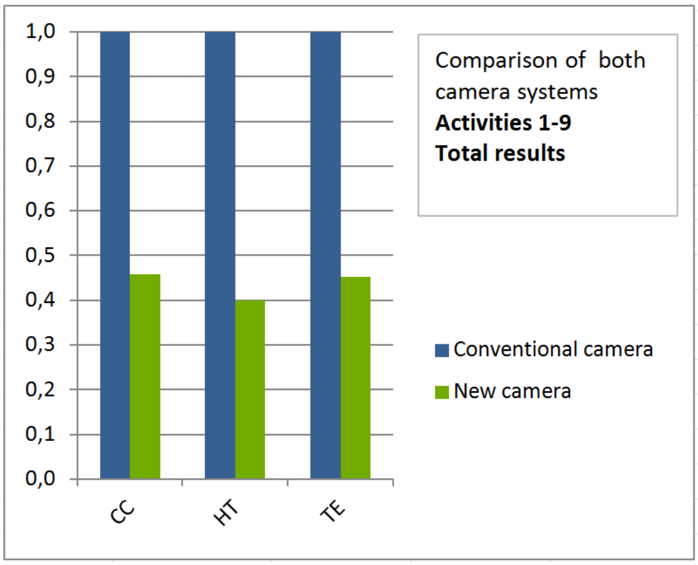

Figure 14.4. Comparison of the relative results of the conventional and the new camera system.

Figure 14.5 shows the proportional shares of the main camera parts. Both cameras show a similar distribution between the characterisation categories, but their respective shares differ significantly. It is valid for all categories, that the housing, the optics, and the miscellaneous parts play in sum a minor role in the range of 2.5 % (HT_{old}) to 8 % (CC_{old}) and 0.5 % (HT_{new}) to 13.5 % (CC_{new}), respectively.

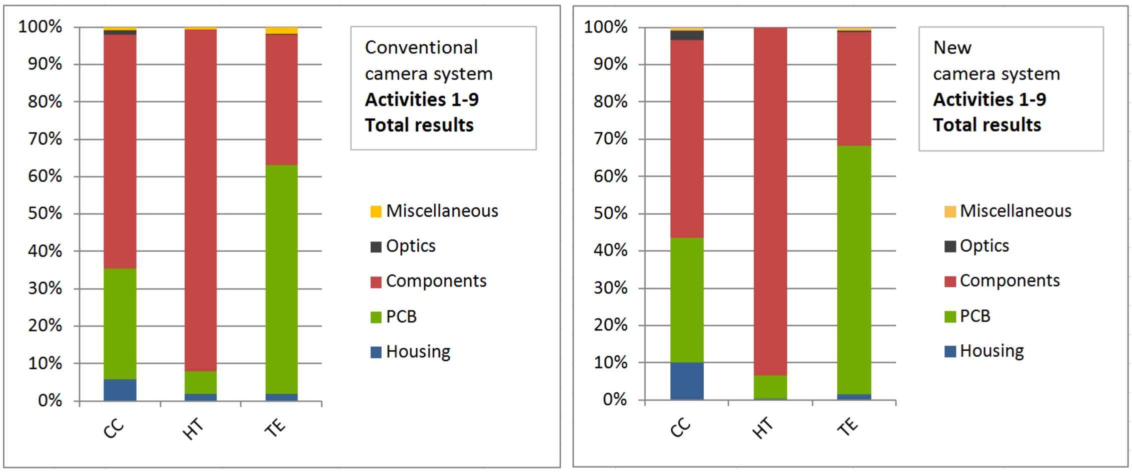

Figure 14.5. Relative LCA results for the assembled conventional camera system (left) and the new demonstrator camera (right) divided into the main parts.

The main impact contributions are split into the printed circuit board manufacturing and the electronic components manufacturing. The Human toxicity is caused by the components manufacturing for more than 90 % for both cameras. Apart from that, the Terrestrial ecotoxicity is caused by the printed circuit board production between 61 % (TE_{old}) and 67 % (TE_{new}). The components manufacturing dominates the contribution to the Climate change with 62.5 % (CC_{old}) and 53 % (CC_{new}).

14.4.2. LCIA Results of the Housing

The results of the characterisation categories of the activities 1–3 for both camera systems are presented in table 14.12 and figure 14.6, respectively.

Table 14.12. Comparison of the total results of the housing for both camera systems.

Σ Activities 1 – 3	CC	HT	TE
	kg CO_2 eq	kg 1,4-DB eq	kg 1,4-DB eq
Conventional camera	0.108	0.340	1.01E-05
New camera	0.085	0.025	3.35E-06

The assessment is done on basis of available material data, which are different for the conventional and the new camera system in terms of the connectors. The conventional camera has an integrated gold plated connector and there was no equivalent available for the new camera. Figure 6 shows that the material gold is highly significant for the Human toxicity with 75 % and for the Terrestrial ecotoxicity with 20 % share. The contribution to the Climate change with 4.5 % is not relevant. Although the housings are not comparable due to the missing connector, this does not affect the overall results according to figure 14.5., since in the relevant characterisation categories HT and TE the housing is negligible compared to the other parts.

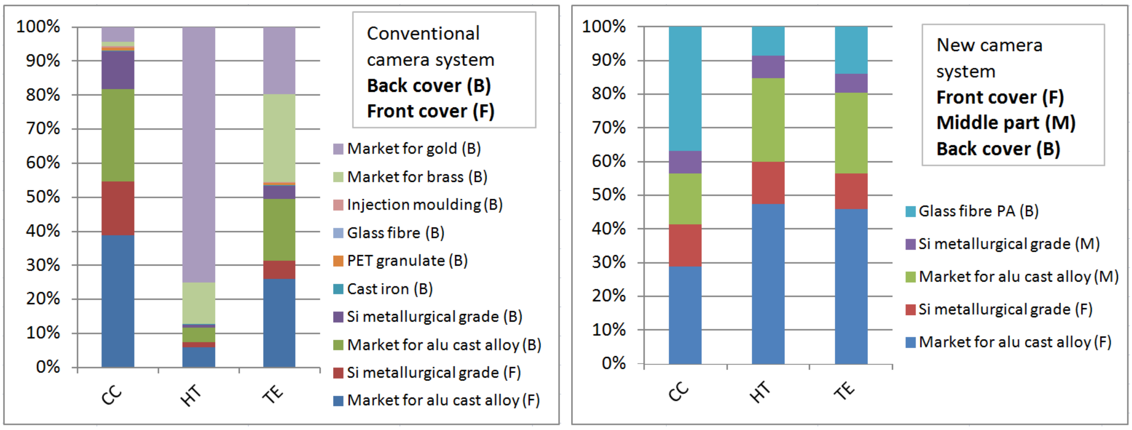

Figure 14.6. Relative LCA results for the housing of the conventional camera system (left) and the new demonstrator camera (right).

Besides gold, the aluminium production shows a large contribution, which is characterised by a share of 41 % related to the electrical energy consumption. Associated with the connector of the conventional camera, the material brass has a significant environmental impact in the category of Terrestrial ecotoxicity with 26 %. The new camera includes glass fibre reinforced polyamide which has a relevant impact on the Climate change with 37 % total share.

14.4.3. LCIA Results of the Electronic Parts

The numerical results of the characterisation categories for the printed circuit board and components manufacturing (activities 4–6) are presented in table 14.13. The unspecified results relate to the ecoinvent dataset for average populated PCBs. The separate calculated results include the modifications according to Section 14.3.2.2., which are consistently lower.

Table 14.13. Comparison of the total results of the electronic part for both camera systems, including the printed wiring board and components manufacturing, and the mounting.

Σ Activities 4 – 6	CC	HT	TE
	kg CO_2 eq	kg 1,4-DB eq	kg 1,4-DB eq
Conv. unspecified	2.180	18.30	6.90E-04
Conv. sep. calc.	1.700	18.00	4.97E-04
New camera	0.733	7.34	2.28E-04

Figure 14.7 shows the comparative results of the PCBs and eight groups of electronic components. The relations between PCB and components are the same as described in figure 14.5. Within the components group the integrated circuit production holds the main share for all categories: The detailed results of the contribution tree of the openLCA calculations show that the IC manufacturing has the highest impact in the category of Climate change with 66 % (CC_{old}) and 65 % (CC_{new}). It if followed by the Human toxicity with 56 % (HT_{old}) and 53 % (HT_{new}), and the Terrestrial ecotoxicity with 54 % (TE_{old}) and 53 % (TE_{new}). The connector shows a high Human toxicity impact for the new camera, therefore, it is needed to investigate the material composition in more detail.

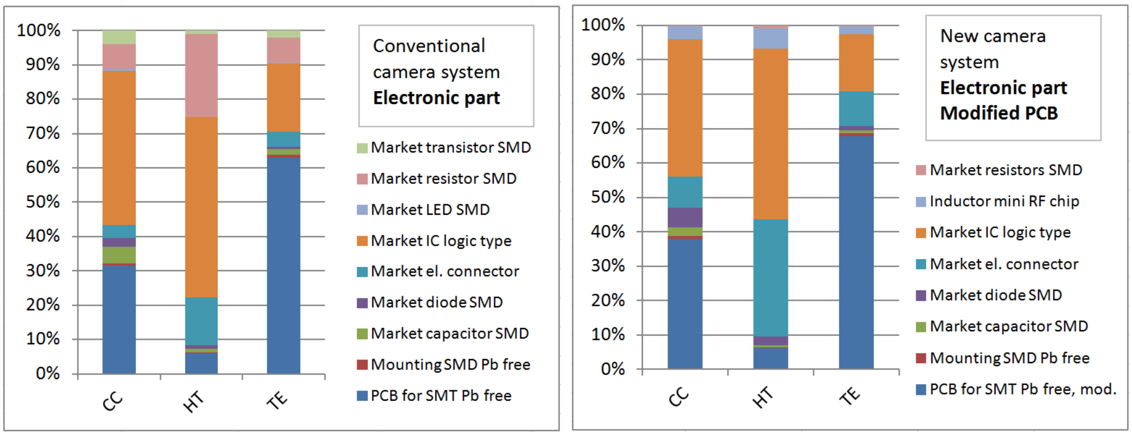

Figure 14.7. Relative LCA results for the electronic part of the conventional camera system (left) and the new camera system (right) including the separately calculated printed circuit board and the electronic components.

The main impact contributions of the IC manufacturing itself in the category Climate change are 66 % electrical energy consumption and 23 % gold production according to the contribution tree of the openLCA LCIA results. The main contribution to the category Human toxicity is the gold production with 97 %. Finally, the main contributions to the Terrestrial ecotoxicity are 74 % gold production and 21 % electrical energy consumption. The IC dataset is equal for both cameras and differs only by total mass.

Figure 14.8 shows a comparison of all considered scenarios for PCB and components manufacturing. The differences between the standard datasets and the modified LCI data for the new camera are insignificant for all characterisation categories. Concerning the conventional camera, there are large differences between 20 % and 30 % for Climate change and Terrestrial exotoxicity. That means, the unspecified dataset for surface mounted PCBs of the ecoinvent database is not applicable for all characterisation categories.

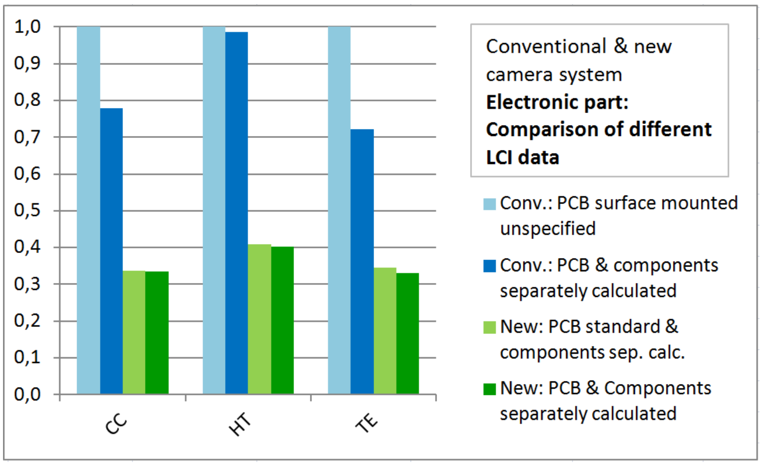

Figure 14.8. Comparison of the relative LCA results for all life cycle inventory scenarios of the mounted printed circuit boards for both camera systems.

14.4.4. LCIA Results of the Optics

The life cycle inventory data for the optics have a large uncertainty. Table 14.14 and figure 14.9 show the results for the assumption, that the glass and the housing part have a mass ratio of 1:1. In this case, the aluminium production has the highest impact share for all characterisation categories.

Table 14.14. Comparison of the total results of the optics, equal for both camera systems.

Activity 7	CC	HT	TE
	kg CO_2 eq	kg 1,4-DB eq	kg 1,4-DB eq
Both cameras	0.023	0.0076	1.11E-06

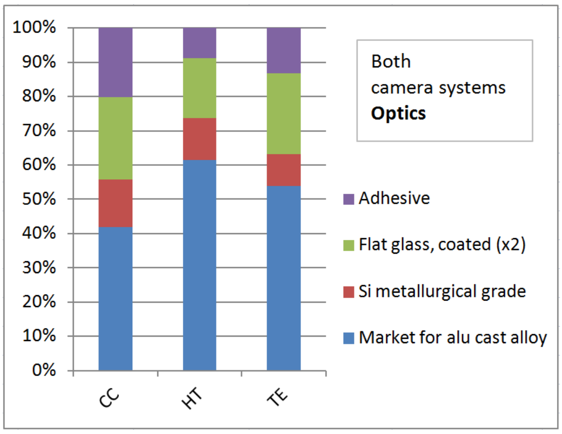

Figure 14.9. Relative LCA results of the optics applied for both camera systems.

14.4.5. LCIA Results of the Miscellaneous Parts

The miscellaneous parts include screws, clips, seals, and adhesives. The comparative results of the characterisation categories for the activity 8 are presented in table 14.15 and figure 14.10, respectively.

Table 14.15. Comparison of the total results of the miscellaneous parts for both camera systems.

Activity 8	**CC**	**HT**	**TE**
	kg CO_2 eq	kg 1,4-DB eq	kg 1,4-DB eq
Conventional camera	0.0153	0.1140	9.47E-06
New camera	0.0066	0.0028	1.97E-06

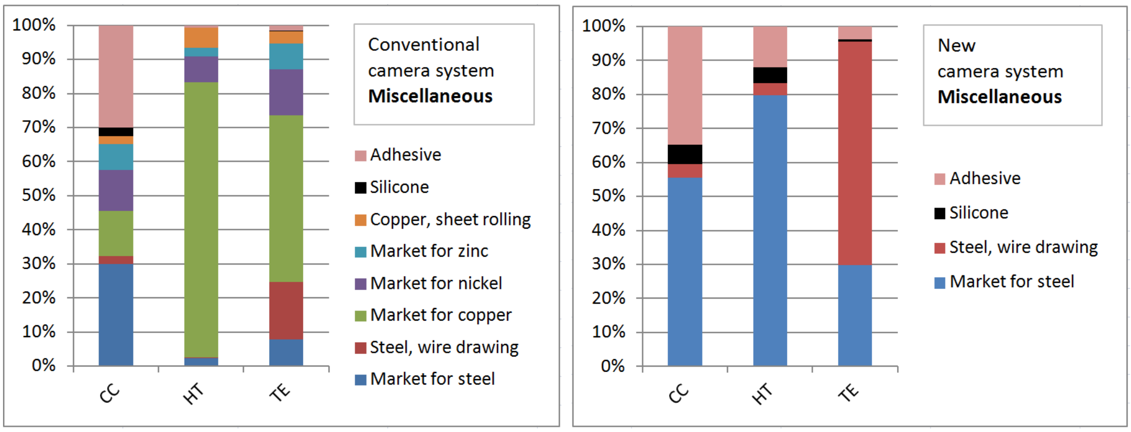

Figure 14.10. Relative LCA results of the miscellaneous parts of the conventional camera system (left) and the new demonstrator camera (right).

The new camera requires less miscellaneous parts and therefore, less material. The adhesives have a large share in the category Climate change and therefore, their mass should be evaluated more precisely. The impacts of the conventional camera are dominated by copper and steel. In particular, copper has a share of 81 % in the category Human toxicity, followed by 49 % in the category Terrestrial exotoxicity. The steel production and processing of the new camera system dominate all impact categories, as the screws are the main part. The process wire drawing, which was used due to the lack of a suitable dataset for the screws manufacturing, shows a high impact in the category of Terrestrial ecotoxicity.

14.5. Conclusion

The manufacturing of one sample of the new demonstrator camera system causes climate change emissions of 0.85 kg CO_2 eq, Human toxicity emissions of 7.38 kg 1,4-DB eq, and Terrestrial ecotoxicity emissions of 0.23 mg 1,4-DB eq. The embedding technology shows a clear reduction of the environmental impact of the new demonstrator camera system within the system boundary. The mass of the new camera is reduced by one third of the conventional system. This compares with the results of the environmental assessment which shows, that the new demonstrator camera system with the highly miniaturized PCB module with embedded components has less than the half of the environmental impact of the former camera type fabricated in conventional SMD technology in all impact categories.

The new printed circuit board production has a large impact share in the category Terrestrial ecotoxicity with 68 %. The manufacturing of the electronic components affects the Human toxicity with a share of 94 %.

Printed circuit boards are available with different layer compositions and processing stages, which influence the environmental impact. The current work is a first assessment for the embedding technology PCB production, based on the available standard ecoinvent datasets with different modifications of the material input flows. An actual published case study shows that the global warming potential based on the electrical energy consumption for embedding technology PCB processing significantly increases by the factor 3 compared to the conventional PCB manufacturing (Franz et al., 2019). This result is an important indication that it is not sufficient to solely obtain accurate LCI data on a material composition and geometric form.

The application of the midpoint assessment method shows that each characterisation category identifies different environmental hot spots within the camera parts. The method is useful for problem oriented evaluation of a device. Apart from that, the characterisation categories cannot be compared to each other and therefore, a weighting and identification of environmental hot spots of the product design is not possible. As long as the product is still in the stage of development and demonstration, highly aggregated environmental assessment methods should be applied to identify necessary design and process corrections before entering the market.

14.6. References

Amidon-Deutschland (2019). Profi-Electronic Deutschland. Übersicht weichmagnetische Kernmaterialien. http://www.amidon.de/contents/de/d641.html [04/2019].

Deutsches Kupferinstitut (2005). CuZn5. Werkstoff-Datenblätter.

Ecoinvent (2018). http://www.ecoinvent.org/ [02/2018].

Franz, M.; Unger, M.; Stahr, H.; Morianz, M.; Kollmann, R.; Nicolics, J. (2017). Environmental aspects of new embedded power modules with 500 W and 50 kW. in: IEEE 2017 40th International Spring Seminar on Electronics Technology (ISSE), IEEE Xplore, 2017, ISBN: 978-1-5386-0582-0, S. 1 - 6.

Franz, Manuela; Kupka, Tobias; Schmid, Gerhard; Nicolics, Johann (2019). Greenhouse Gas Emissions of Printed Circuit Board Manufacturing of Different Technologies. IEEE 2019 42th International Spring Seminar on Electronics Technology (ISSE), Book of Abstracts, May 15-19, 2019, Wroclaw, Poland; ISBN: 978-83-7493-083-3, pp. 151-152.

Goedkoop, Mark; Heijungs, Reinout; Huijbregts, Mark; De Schryver, An; Struijs, Jaap; van Zelm, Rosalie (2013). ReCiPe 2008 – A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First Edition (revised). Ruimte en Milieu. Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer, 2013.

ISO 14040 (2006a). Environmental management – Life cycle assessment – Principles and framework.

ISO 14044 (2006b). Environmental management – Life cycle assessment – Requirements and guidelines.

openLCA (2018). http://www.openlca.org/ [01/2018].

Römpp Enzyklopädie Online (2019a), Stuttgart: Thieme Verl., URL: http://www.roempp.com [04/2019]. Abschnitt: Polybutylenterephthalate.

Römpp Enzyklopädie Online (2019b), Stuttgart: Thieme Verl., URL: http://www.roempp.com [04/2019]. Abschnitt: Gold.

Römpp Enzyklopädie Online (2019c), Stuttgart: Thieme Verl., URL: http://www.roempp.com [04/2019]. Abschnitt: Kupfer.

Römpp Enzyklopädie Online (2019d), Stuttgart: Thieme Verl., URL: http://www.roempp.com [04/2019]. Abschnitt: Aluminiumoxide.

Römpp Enzyklopädie Online (2019e), Stuttgart: Thieme Verl., URL: http://www.roempp.com [04/2019]. Abschnitt: Silicium.

Rotek Handels GmbH (2008). Glasfaser-Hartgewebe HGW2372.1 (FR4-HF). DE V1.1 Stand 08-2008.

VAR (2018). Aluminium-Gusslegierungen. Verband der Aluminiumrecycling-Industrie e.V. http://www.schoett-druckguss.de/assets/Uploads/pdf/legierungen.pdf [01/2018].

Wallenberger, Frederick T.; Bingham, Paul A.; (Eds.) (2010). Fiberglass and Glass Technology. Springer, 2010.

15. General Conclusion

The thesis investigates the extent to which the environmental performance of electrical and electronic products and renewable energy generation systems can be described using existing commonly used methods and life cycle inventory data. Moreover it explores what modifications are needed to comprehensively understand the environmental and human impacts of the dynamic global technology development of the electrical sector.

In ten case studies the market situation and environmental performance of selected examples from the sectors of materials for electrical engineering, electronic components, modules and devices as well as systems of electrical power generation are investigated. The results show that, according to the theses formulated in Chapter 2, the use of streamlined life cycle assessment methods on the basis of existing life cycle inventory data is not sufficient to fully describe the environmental performance of the continually changing global electrotechnical sector.

The following summarises the results for the three key research questions of Chapter 2. Answering the ***first key research question*** about the ***dimension of the global electrical sector***, according to Chapter 4, 11 % of the value of global industrial production is assigned to the electrical sector, of which slightly less than half is assigned to the electronics sector. The total market volume of the electrical sector is about 4 trillion Euro per year. At least 63 % of electrical products are manufactured in Asia and 12 % in Europe. It is notable that about half of the German production of electronic components, modules, and semiconductors is related to the automotive industry. These relations apply for economic data and they do not have to be identical to mass flows as shown by the example of the global lighting market in Chapter 7. Furthermore, it is not specified in which region the products of European companies, to which the revenues are assigned, are actually produced. Therefore, it could be possible that the mass–based total production in Asia as well as the import / export trade in goods and thus the environmental pressure of local manufacturing and transportation in Asia are much higher according to Chapter 4 Table 4.7.

The annual global e–waste volume is 42 million tons, whereby only 15 % are treated according to the state of the art. There are major research gaps in this field, such as the actual extent of the environmental pressure of informal e–waste treatment and its allocation to the end–of–life management in LCA case studies. The comparison of regional e–waste flows with the production flows of electrical products shows that in Asia 39 %, and in Europe and the Americas 28 % each of the global e–waste is generated and thus there are significant regional differences to the global production share.

The illegal intercontinental transfer of e–waste is known but not further quantified and assigned to life cycle inventory data. The legal export of functional WEEE from Europe to Africa is also problematic, because only a few years later it becomes hazardous waste, which will be unregulatedly disposed in the respective region. Therefore, it is recommended to amend the WEEE directive in such a way that a formal state of the art disposal shall be ensured on site for each exported device.

The recorded material flows of collected and treated electrical products are limited to metals, plastics, glass, selected pollutants, such as cooling liquid, as well as some valuable metals such as gold, silver, and copper. There is a research gap of material flows of a large number of other valuable substances, such as indium and rare earths, as well as toxic substances and substances of very high concern, such as polycyclic aromatic hydrocarbons and bisphenol A. Some of the substances are included in very small quantities in individual electrical appliances, but in case of rare earths, strategic metals, or flame retardants, the total global amount of devices generates a significant environmental pressure. There is a need for a regional differentiation of hazardous material flows, since, for example, the European Union's Restriction of Hazardous Substances differs from the Chinese one in that there is only a declaration requirement in China and no ban.

The ***second key research question*** concerns the ***appropriate evaluation method*** for electrical products. The electrical energy consumption in the use phase is a unique characteristic and a significant environmental factor in the life cycle of electrical products. As a result, a number of environmental assessments are limited to midpoint indicators such as the carbon footprint or the aggregated global warming potential, while other environmental indicators are omitted. This can be useful for devices that are in use for decades. Since electrical devices became more energy efficient and the lifetime tends to decrease, the environmental pressure of the manufacturing phase gets more significant. Further, the time period of the development phase of an electronic component or module is in many cases longer than the final mass product on the market. That means, LCA studies in the development phase have to continuously consider ongoing market changes as well as technical and environmental frameworks and the possible change of standards. Forecasted technology developments and a possible change of the consumer behaviour should be reviewed within short intervals.

Lessons can be learned by the example of LED lamps, where streamlined LCA studies with high data uncertainties were not able to comprehensively describe the environmental pressure, the well–being of the user, and the rebound effect as investigated in the Chapters 7 and 10. The commonly used functional unit of one device, with the respective technical parameters and lifetime, excludes spatial and temporal conditions of manufacturing and use. A more suitable functional unit could be in case of lighting a specific illuminated distance and/or area in case of outdoor lighting or the illumination of a household per year, which includes on the one hand possible parallel operating devices and on the other hand the annual operation time. The change of the functional unit is therefore accompanied with the change of the system boundary.

The mass related functional unit of the databases is an important indicator for the system boundary, accurate life cycle inventory data, and the informative value of the LCIA results. In case of LCI data, electronic devices show in many cases geometric dependent processes. This concerns printed circuit board and wafer manufacturing as well as any component made of thin structured layers. In this cases the number of layers and the area are more important than the mass of the material. There are some conflicts with LCI databases where electronic components are normalised to mass–units such as integrated circuits and mounted PCBs (see Chapter 13 and 14). Furthermore, comparative studies could lead to opposite results whether the functional unit is defined on mass or on area basis such as ceramic thin layers (see Chapter 9).

The product development phase and the stage of the product put on the market have different requirements concerning the objectives of the environmental assessment. In the development phase, highly aggregated assessment methods are appropriate to clearly show the environmental hot spots of the manufacturing process from cradle–to–gate. In this stage it is possible to identify and optimise the material composition and the process design. The functional unit should be one piece or a batch, but it is important to generate data for the mass production as it is shown in Chapter 9 and 14 to avoid misleading results concerning the amount of the electrical energy consumption and the applied electrical energy mix of the location of the mass production, respectively. For example, the Chinese electrical energy mix for mass production produces a multiple of CO_2 emissions than the eco–certified electrical energy mix for the product design phase in Austria.

The development of highly sophisticated new materials and their processing for electrical application have less degrees of freedom for variations of materials and process parameters, since the optimal point of physical properties and reliability shows small tolerances. That means, the environmental optimisation has to be made by indirect measures such as using renewable energy sources for the electrical energy consumption, design for repair and recycling, long lifetime, or sustainable raw material acquisition.

Electronic materials, components, and modules show environmental hot spots for ceramic firing processes and laser processing (see Chapter 9), the manufacturing of copper and epoxy resin (see Chapter 12) as well as the wafer manufacturing of integrated circuits and the production of printed circuit boards and silver (see Chapters 12, 13, and 14). It is notable, that in the conducted case studies, the environmental impact of gold plays a minor role compared to the other materials.

Midpoint assessment methods are problem oriented and therefore, they are appropriate to investigate single environmental indicators of the mass production and the product put on the market on local level such as CO_2 emissions, particulate matter, human and eco–toxicity, or agricultural land occupation. The respective results are not comparable with each other and therefore the single indicators do not reflect the environmental performance of the entire product. The midpoint indicator Global warming potential, for example, is useful to investigate the use phase of an electronic product, but not the entire environmental performance of the device. The indicator agricultural land occupation is appropriate to investigate the land conflict potential of renewable energy sources (see Chapter 11).

All the above described assessment methods relate only to materials. A holistic life cycle assessment has to extend the system boundary to the consumer behaviour, the well–being of the affected user or animals, and to the global market context besides individual product assessments.

The ***third key research question*** concerns the ***future challenges*** arising from the high dynamic of the electrical sector. The future development of the electrical sector concerns different aspects of the life cycle. The total amount of electrical products of all sectors will continuously increase which will be reflected in the increasing global volume of e–waste, the energy consumption in manufacturing and use phase, and resource scarcity of rare earths and strategic metals. Nevertheless, also the local relations of the environmental pressures will change.

The environmental assessment of the production of new electronic materials and sophisticated components should not be determined directly from the existing LCI data of the ecoinvent database. This was demonstrated by the examples in Chapters 9-10 and 12-14. It has been shown in all studies, that even accurate determination of the material composition and geometry, which were provided by manufacturing companies, is insufficient for the environmental evaluation because miniaturised devices have a significantly higher demand for electrical energy. Still unknown is the amount and type of doping materials and nanomaterials included in metal compounds, ceramics, and plastics, which are not considered in the literature or in the studies carried out here. The commonly used cutting rules in LCA studies in terms of omitting small amounts of materials less than 1 % is not applicable for the electronics sector, because these material are expected to generate a high environmental pressure. Their consideration leads to an increase of the mass–specific environmental impact and it must be assumed that the environmental impact of the global component sector is significantly underestimated. Furthermore, the fabrication of silicon substrates within integrated circuits should be evaluated based on area rather than mass, since miniaturisation primarily concerns the reduction of the substrate thickness.

The ecoinvent database uses the electrical energy mix "GLO" for all electronic components and printed circuit boards. According to Chapter 4, since the major part of electronic devices is manufactured in Asia, where the environmental pressure from energy generation is higher than the global energy mix, the life cycle inventory data should be adapted to the actual manufacturing countries.

The investigation of the environmental pressure of the use phase of computers and telecommunication equipment should consider, additionally to the power consumption of the device, a share of the energy consumption of global data networks and the related equipment, which is currently completely omitted.

In the field of energy generation, new challenges arise due to the high direct land use of power plants based on renewable energy sources, as shown in the Chapters 5 and 11. This additional land consumption is not reflected in the databases and should be considered in new generated LCI data. The environmental assessment of renewable energy sources needs a broader system boundary and holistic assessment methods. Due to the emerging area competition with food and feed production, industrial raw materials, and the preservation of natural habitats, interdisciplinary research approaches are needed, which also include energy spatial planning on a local and a global level.

The future increase in the global demand for energy efficient electrical appliances, such as photovoltaic modules and storage batteries, will create high environmental pressure for the manufacturing countries that are located in the Asia and Pacific region according to the Chapters 4 and 11, which generate higher manufacturing emissions than it would be in industrial locations in Europe. It is recommended to define environmental requirements for the manufacturing countries in the European Ecodesign Directive.

In the life cycle stage of e–waste management, the environmental pressure will increase as a result of the expected high mass of end–of–life PV modules (see Chapters 5 and 6), vehicle batteries, and wind turbines.

A sustainable circular economy depends on improved recycling technologies. The development of better recycling technologies for electrical devices, such as the recovery of rare earths, strategic metals, and renewable base materials, requires more accurate declaration and separation of e–waste. From this follows the necessity of the development of new collection and logistics systems and a higher differentiation of treatment steps of end–of–life devices. This is not possible in the framework of the current European system, since only the recycling targets of the WEEE Directive are applied, which do not cover these materials. Further differentiation is associated with higher costs and requires qualification of the working staff all over Europe.