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MINING THE TECHNOSPHERE

“Drivers and Barriers, Challenges and Opportunities”



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MINING THE TECHNOSPHERE “Drivers and Barriers, Challenges and Opportunities”

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1 Presentations





1.1 Session A:
**Resource Evaluation and Classification – Analogies and
Differences between Natural and Anthropogenic Resources**





EVALUATION AND CLASSIFICATION OF NATURAL RESOURCE DEPOSITS

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Need for a harmonized classification system

The successful and profitable operation of a deposit depends on sufficient resources (quantity and grade) and a high bonity (factors, controlling the production costs). Therefore reliable and reproducible reserve and resource estimates are of utmost concern for mining companies and investors as well.

Tools for company-reports

The very first minerals resources evaluation systems have been compiled more than hundred years ago by the Institution of Mining and Metallurgy. In the 1920-ies the Soviet Union developed its own classification system, which was revised in the 1960-ies. Important mining countries compiled own and differing classification systems, so that it became nearly impossible to compare reserve or resource figures. According to BGR-Hannover there exist more than 150 different reporting systems.

Amongst others in the 1990-ies South Africa created a modern reporting code for companies SAMREC [South African Mineral Codes] etc). Similar methodologies have been developed in Australasia (JORC [Joint Ore Reserves Committee]). In order to work against this adverse development in 1994 the „Committee for Mineral Reserves International Reporting Standards“ (CRIRSCO) has been founded. CRIRSCO is a grouping of representatives of organisations that are responsible for developing comparable mineral reporting codes and guidelines in Australasia (JORC), Canada (NI43-101, CIM), Chile (National Committee), Europe (National Committee PERC), Mongolia (MPIGM), Russia (NAEN), South Africa (SAMREC) and the USA (SME). According CRIRSCO the combined value of mining companies listed on the stock exchanges of these countries accounts for more than 80% of the listed capital of the mining industry (<http://www.criresco.com/background.asp>).

For mining companies quoted on stock exchanges the use of reporting codes is mandatory. Therefore reporting codes should guarantee reliable and transparent information for investors. The advantages and disadvantages of these different reporting codes for companies are discussed in detail by L. WEBER (2013).

Tools for governmental or global reporting

In 1976 both U.S. Bureau of Mines and the U.S. Geological Survey developed a common classification and nomenclature for national or global reporting, which was published as U.S. Geological Survey Bulletin 1450 A (Geological Survey Circular 831) with the well known “McKelvey Box”. However, the McKelvey classification should not be used in assessment studies except for descriptions of identified resources.

In the 1990-ies the „Committee of Natural Resources of the United Nations Economic and Social Council“ (ECOSOC) started a motion to compare the different terms and definitions in order to provide a tool for governmental, inter-governmental, or NGO reporting of mineral resource estimates and forecasts. The United Nations Framework Classification (UNFC) is a flexible system that is capable of meeting the requirements for application at national, industrial and institutional level, as well as to be successfully used for international communication and global assessments. United Nations provides international legitimacy and a platform where all stakeholders can meet in a neutral setting. The UNFC provides a mechanism for companies to use a standardised internal classification beyond the publicly reported CRIRSCO categories (exploration results; discovered, but not economic; recovered, but unrecoverable)

http://www.criresco.com/news_items/CRIRSCO_and_UNFC_2013.pdf

Table 1 **examples of different definitions for reserves and resources**

USGS	CRIRSCO
<p>Resource: A concentration of naturally occurring solid, liquid, or gaseous material in or on the Earth's crust in such form and amount that economic extraction of a commodity from the concentration is currently or potentially feasible</p>	<p>Mineral Resource: A Mineral Resource is a concentration or occurrence of solid material of economic interest in or on the Earth's crust in such form, grade or quality and quantity that there are reasonable prospects for eventual economic extraction. The location, quantity, grade or quality, continuity and other geological characteristics of a Mineral Resource are known, estimated or interpreted from specific geological evidence and knowledge, including sampling.</p> <p><i>Remark: Mineral resources are subclassified to Inferred, Indicated and Measured Mineral Resource.</i></p>
<p>Reserve: That part of the reserve base which could be economically extracted or produced at the time of determination. The term reserves need not signify that extraction facilities are in place and operative. Reserves include only recoverable materials; thus, terms such as “extractable reserves” and “recoverable reserves” are redundant and are not a part of this classification system.</p>	<p>Mineral Reserve: A Mineral Reserve is the economically mineable part of a <i>Measured and/or Indicated Mineral Resource</i>. It includes diluting materials and allowances for losses, which may occur when the material is mined or extracted and is defined by studies at Pre-Feasibility or Feasibility level as appropriate that include application of Modifying Factors. Such studies demonstrate that, at the time of reporting, extraction could reasonably be justified. The reference point at which Reserves are defined, usually the point where the ore is delivered to the processing plant, must be stated. It is important that, in all situations where the reference point is different, such as for a saleable product, a clarifying statement is included to ensure that the reader is fully informed as to what is being reported.</p> <p><i>Remark: Mineral reserves are subclassified into Probable Mineral Reserves and Proved Mineral reserves.</i></p>

In the **United Nations Framework Classification (UNFC)** the terms “reserves” and “resources” are not defined, because they both have specific, but different, definitions in the solid minerals and petroleum sectors. The terms are used purely in a generic sense to encompass all possible Classes and Sub-classes that are valid in UNFC-2009.

Improper use of reserve and resource figures

Reserve and resource figures are often used improperly by non-experts, e.g. to calculate the lifetime of a particular mineral commodity (lifetime = resources / annual production). Resources figures may vary even in short time due to rapidly changing modifying factors (e.g. commodity price). On the other hand mining companies are not interested in providing reserves or resources for some decades, as this means frozen capital. However it may not be overseen, that mineral resources are not renewable and limited. Therefore it is extremely important to keep an eye on the changing development of the global exploration to identify supply risks.

Added value of mineral resources figures

The combination of reserves and resources figures with fundamental geological information (e.g. tectonic position, deposit type, age of mineralisation) of the respective deposit provide important statements concerning the prospectivity of particular regions. Similarly the combination of resources data with geopolitical information (e.g. political stability of the producer countries) allows the identification of potential supply risks (WEBER. L., 2015).

According L. WEBER & J. LIU 2015 the published CRIRSCO conform resources estimates of more than 240 mining sites of Rare Earth Elements (REE) deposits allow to allocate 53,39% of the global resources to Asian countries. Surprisingly Europe contributes with 17,74% and ranks before North America (13,34%), Africa (8,89%), Oceania (5,99%) and Latin America (1,64%).

Combining the REE-resources figures with the “Political Risk Factor” of World Governance Index (Kaufmann, D. et al 2010) show, that 54,84% of the global REE resources are located in political unstable countries, 6,31% in extreme unstable countries. However, 24,13% of the global resources are linked to countries with a quite fair political stability, 13,72% in political stable countries (see also Fig. 1).

Merging the resource figures with the different type of REE deposits shows clearly the major importance of carbonatite associated deposits (72,44% both primary and lateritic), alkali-associated deposits (21,16%). Surprisingly the Ion absorbing clay-type can be characterized as single small deposits with extreme poor REE grades only (Fig. 2).

Those combinations of reserves and resources with geopolitical or geological information allow precise analyses for potential supply risks.

Figure 1 Resources vs Grade of REE deposits by political stability of the country (source: L. WEBER & J. LIU)

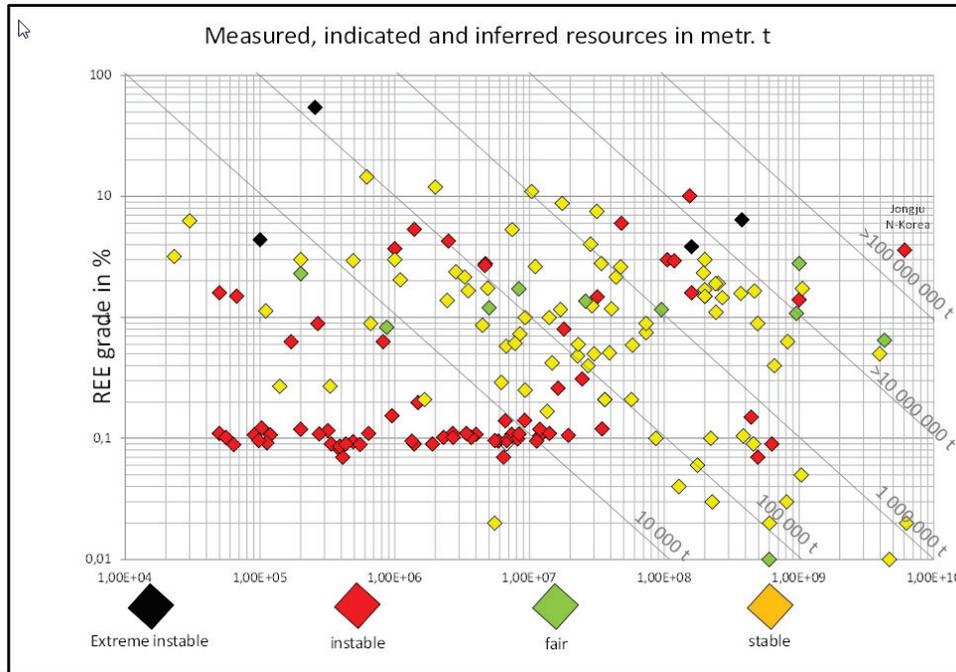
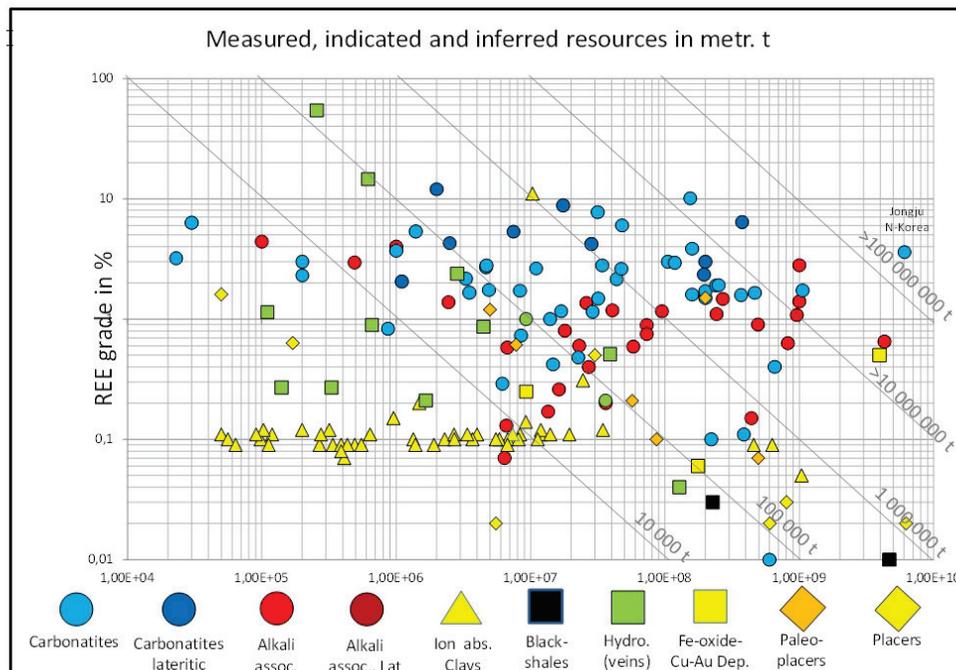


Figure 2 Resources vs Grade of REE deposits by type (source: L. WEBER & J. LIU)



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Kaufmann, D., Kraay, A. & Mastruzzi, M. 2010, The Worldwide Governance Indicators: Methodology and Analytical Issues (September 2010). World Bank Policy Research Working Paper No. 5430. Available at SSRN: <http://ssrn.com/abstract=1682130>

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<http://info.worldbank.org/governance/wgi/index.aspx#home> (retrieved Sept. 7th 2015)



EVALUATION AND CLASSIFICATION OF ANTHROPOGENIC RESOURCE STOCKS AND FLOWS – THE CASE OF PHOSPHORUS AND ZINC

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INTRODUCTION

The last two centuries have shown a significant increase in consumption of material resources. One of the predicted consequences of this growth is that humankind is facing shortage of resources, leading to initiatives to overcome expected problems with resource supply (UNEP, 2013). Corresponding concepts have been proposed by numerous authors with so-called anthropogenic mining concepts, i.e. landfill or urban mining. Though particularly the latter term has become popular, there is some critical discussion with respect to the definition and applicability of urban mining concepts (Johansson et al., 2012).

The aim of this contribution is to present a consistent terminology of anthropogenic resources and their utilization for the purpose of extracting secondary raw materials for production, derived from natural resource terminology, evaluation and classification (Ciriacy-Wantrup, 1944; Hartman and Mutmanský, 2002; USGS, 1980). Furthermore, two case studies on evaluating two different anthropogenic resources (Phosphorus recovery from anthropogenic stock resources and Zinc recovery from MSWI fly ashes) are presented to demonstrate the application of an evaluation concept for anthropogenic resources.

CONCEPTUAL FRAMEWORK FOR ANTHROPOGENIC RESOURCE EVALUATION

The critics of Johansson et al. (2012) on current concepts under the popular term *urban mining* can be first of all summarized as one of lacking definitions. This means that a conceptualization of activities that aim to evaluate and classify anthropogenic resources should start with a clear and consistent terminology. In order to be consistent with existing terminologies and concepts of natural resource evaluation while not losing the context to popular metaphors (i.e. urban mining), a conceptual framework of definitions in analogy to natural resources by Ciriacy-Wantrup (1944) is presented, generally distinguishing between anthropogenic stock resources (e.g. landfills, buildings) and anthropogenic flow resources (e.g. waste streams) (Lederer et al., 2014). Considering this framework, the term *urban mining* better fits to anthropogenic stock rather than flow resources.

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By having defined the basic terminology, procedures for evaluating these different types of anthropogenic resources are required in order to give the information necessary to decide whether a secondary raw material can be produced from them. Figure 2 shows one possible procedure that derives from primary raw materials mining of natural resource stocks and considers technological and economic factors (Hartman and Mutmansky, 2002).

Evaluation step	Method	Result
1. Prospection	Identification of stocks/flows based on macro-level MFA	Relevant anthropogenic stocks/flow identified and estimated
2. Exploration	Detailed stock/flow characterization based on micro-level studies	Grade, size of stock/flows, incl. uncertainties
3. Evaluation	Selection of technologies and economic analysis of costs & revenues (incl. avoided costs)	Costs/revenues ratio
4. Classification	McKelvey cross classification	Reserves, resources, & other occurrences of anthropogenic stocks/flows

Figure 1 Evaluation procedure for anthropogenic resources (Fellner et al., 2015; Lederer et al., 2014)

The result of this economic and technological feasibility evaluation of anthropogenic resource recovery is presented in a so-called McKelvey diagram, which is also used by the US Geological Survey (USGS) for the evaluation and classification of natural resource deposits in its Minerals Yearbook. Therein, natural resource deposits are classified according to their economic/technical extractability and their certainty/knowledge of existence.

The concept developed will be illustrated by two contrasting case studies, one that evaluates anthropogenic stock resource, and one that considers anthropogenic flow resource.

CASE STUDIES

Evaluation of anthropogenic phosphorus stock resources in Austria

In the ongoing discussion about natural resource scarcity and peak-everything, Phosphorus (P) receives particular attention, due to its essential features as nutrient for living organisms (Cordell et al., 2009). At the same time, research on the extraction of secondary raw materials namely phosphate fertilizer from anthropogenic phosphorus sources is increasing for both, anthropogenic flow resources (Adam et al., 2009; Tan and Lagerkvist, 2011) as well as anthropogenic stock resources such as MSW landfills (Wanka et al., 2013). What all of the mentioned studies have in common is that they assume a significant potential of anthropogenic phosphorus resources. In the light of these developments, the aim of this case study is to apply the concept developed to anthropogenic phosphorus stocks in Austria.

After determining the size of different anthropogenic P stocks in the prospection and exploration phases by material flow analysis (MFA) based on (Egle et al., 2014), a technology for recovering P from stocks where applicable (municipal solid waste (MSW) and municipal sewage sludge (MSS) landfills) is assumed (Adam et al., 2009). Other stocks where recovery in the sense of producing a product similar to mineral P-fertilizer from natural phosphate ores is not possible are considered too. Finally, an economic evaluation that also considers technological feasibility is

performed, leading to a classification of anthropogenic P stocks as economic resource, marginally economic resource, subeconomic resource, and low grade materials and materials not extractable (McKelvey, 1972). The results illustrated in Table 1 show that none of the considered anthropogenic P stocks in Austria can be classified as economic and marginally economic resource, thus being called a reserve under the USGS classification scheme.

Table 1 McKelvey diagram for anthropogenic phosphorus stocks in Austria formed between 1960 and 2009 (Lederer et al., 2014)).

	identified resources		<i>potentially undiscovered resources</i>
	demonstrated	inferred	
economic	0	0	0
marginally economic	0	0	0
subeconomic	70,000	32,000	32,000
other occurrences (low grade)	89,000	105,000	105,000
other occurrences (not extractable)	720,000		374,000
total	1,020,000		511,000

Only 10% (or 102,000 tons), are currently subeconomic (mainly landfills of mixed incineration ashes, MSW bottom ashes, and municipal sewage sludge landfills) and 20% are low-grade materials (194,000 tons), where recovery of P is in theory possible, but will not be performed due to the low P-concentration and high concentration of heavy metals (MSW landfills). Furthermore, the bulk of anthropogenic P-stocks are present in a form where no extraction is feasible. However, the results also suggest that if selected waste streams, i.e. ashes from municipal sewage sludge mono incineration, would not have been landfilled in mixed compartments with other ashes (in this case bottom ashes from MSW incineration), they would have been classified as marginal economic resources and thus reserves.

Evaluation of Zinc recovery from solid MSW incineration residues in Europe

In the second case study, an emerging technology of recovery of Zinc (Zn) from different solid residues of MSW incineration air pollution control (APC) systems (filter ash from wet APC, boiler and filter ash from wet APC, fly ash from dry and semi-dry APC, and bottom ash from grate incineration as well as fly ash from fluidized bed incineration) called FLUREC is evaluated with regards to the economic cost/revenues ratio of Zn recovery, considering the solid residues of all MSW incinerators in the EU-28 plus Switzerland and Norway. As the prospection phase is not necessary due to the preselection of the Zn-carrying material flow (solid MSW incineration residues), the first step is the exploration, performed by an MFA for MSW incinerated in general and flows of Zn in particular.

After the exploration, an economic evaluation is performed that considers all costs and revenues associated with the application of the FLUWA technology, and based on that, a classification according to the USGS (1980) classification scheme for natural resources is done. The results, illustrated in Figure 2, show that only filters ashes from wet APC systems can be classified as economic/marginal economic resources and thus represent reserves. All other solid residues

produced from MSW incineration in the European countries considered were classified as either subeconomic or other occurrences (Fellner et al., 2015).

When considering the size of the anthropogenic Zn-flow resources considered, it becomes clear that only a small portion of Zn in the MSW incinerated of 6% will be ready and suitable for extracting Zn as secondary raw material. The bulk of Zn is ends up in solid MSW incineration residues not economic suitable and/or technological feasible for recovery.

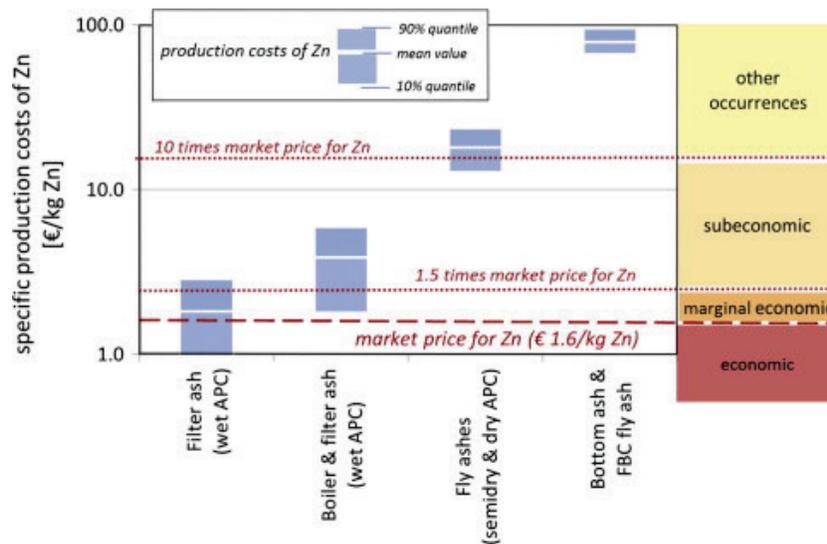


Figure 2 Specific recovery costs for Zn (in €/kg Zn) from different MSWI residues (Fellner et al., 2015).

DISCUSSION & CONCLUSIONS

The framework for the evaluation of anthropogenic resources based on a natural resource evaluation procedure for natural stock resource deposits has been applied to two different types of anthropogenic resources. In the first example of P-stocks in Austria, it was used to evaluate anthropogenic stock resource deposits in the sense of a commodity-specific search for an ore (Hartman and Mutmanský, 2002), which means that the commodity (P for mineral P-fertilizer) was predetermined, while the locations to look at (anthropogenic P-stocks) were not. Therefore, it proved to be quite applicable, which is not surprising due to the fact that a natural stock resource evaluation procedure for mining of raw materials from natural deposits was applied to another stock resource, in this case anthropogenic stocks. While the results are quite useful for an initial evaluation in order to eliminate anthropogenic stocks without any likely utilization to produce secondary raw materials (e.g. P in MSW landfills), the procedure is not applicable for an in-depth assessment of technological feasibility, as performed for instance in landfill mining assessments (Winterstetter et al., 2015). In the second example, an anthropogenic flow resource was investigated in the sense of a site- and commodity specific search for an ore, meaning that both the *locations* (solid residues from MSW incineration) as well as the

commodity (Zinc) were predetermined (Hartman and Mutmansky, 2002). Even though the approach derived from natural stock resource evaluation, it proved to be applicable to these flow resources too, and the results can be integrated into the evaluation of natural Zn-resource deposits.

Acknowledgements

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EVALUATION AND CLASSIFICATION OF HYDROCARBON RESOURCES

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Abstract

Oil and gas reserves and resources are explored, developed and produced from companies around the world. Exploration and production companies are in the jurisdictions of their home countries and, wherever they operate, in the jurisdictions of their host countries. In consequence, international E&P companies may have to apply the provisions of different national laws for the same operation, e.g. drilling a well Offshore North Sea or in the Gulf of Mexico.

They also may have to work with different units for the same medium, depending on the area of operation. Whereas oilfield units are dominant in the US, metric units are widely used in Central Europe. Some companies strictly apply SI units, which is even not identical with metric units.

All these E&P activities under various jurisdictions using different units and expressions for the same things have to be reported to various stakeholders. These may be (i) Governments to manage their resources, (ii) the industry to provide data and information for internal planning purposes and strategic planning, (iii) the Financial Community to provide information necessary to allocate capital accordingly and (iiii) creators of international energy and mineral studies.

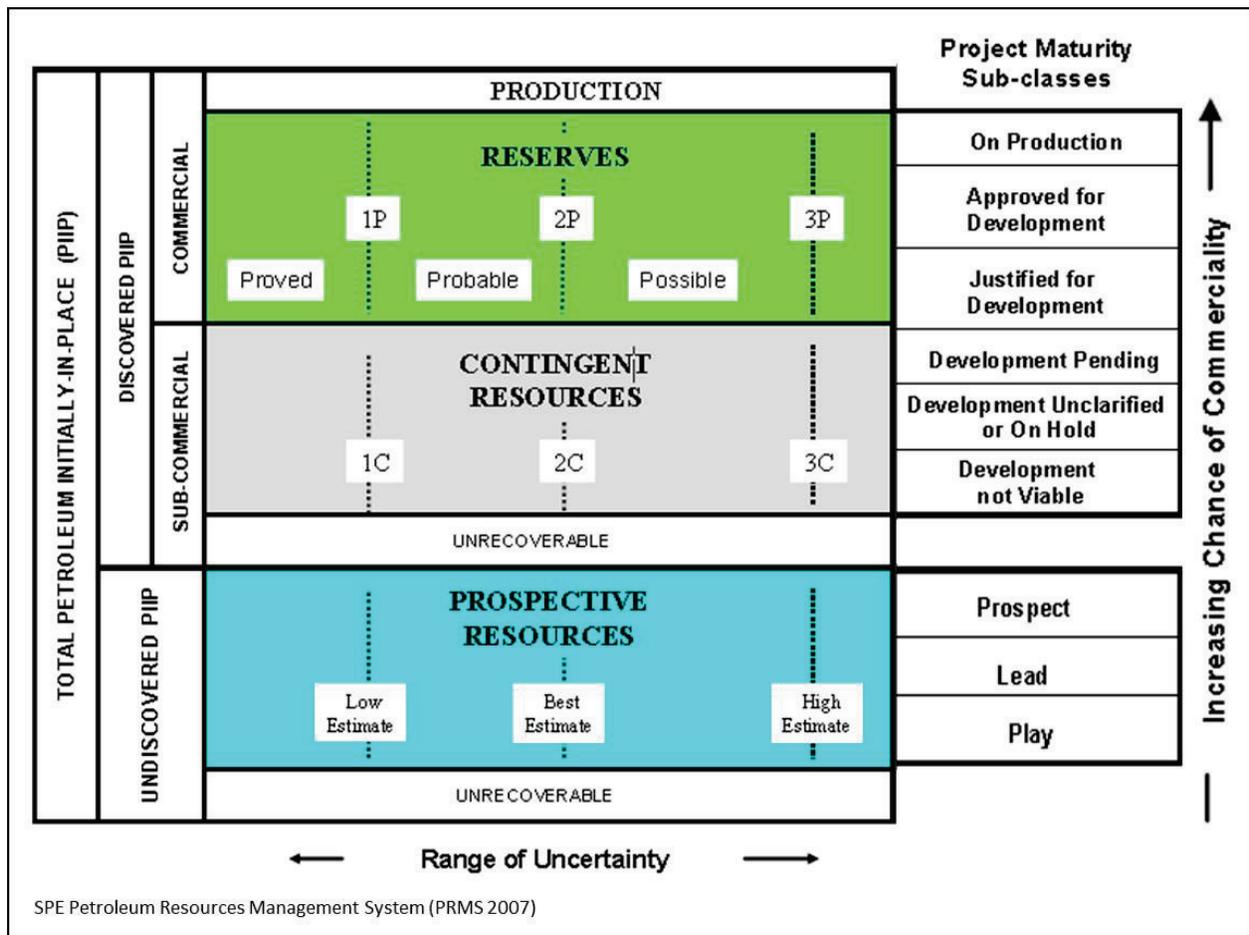
But any individual resource information, may it be for a field or for a company or country, is absolutely meaningless, if the effective date and the units are unknown and if the classification system, which was used to estimate and classify the volumes is unacquainted.

Apparently there are not only hydrocarbon reserves and resources but also mineral resources, uranium, renewables and anthropogenic resources. By nature all have individual systems used to classify and categorize them. Even for hydrocarbon resources, multiple systems exist worldwide.

This presentation refers to the SPE Petroleum Resources Management System (SPE PRMS 2007) (Fig. 1) which is the most commonly used systems and worldwide accepted. It highlights general challenges in reserves and resources evaluation and puts focus on

- Definitions and classification and
- The systematic explaining risk and uncertainties.

Table



The definitions and applied systems define the framework and give guidance for the resources evaluation. Each resource class as there are

- Prospective Resources
- Contingent Resources and
- Reserves

may be further subdivided into subclasses. The class (or subclass) is determined by the chance of maturation of a project. Economic criteria, contractual status and technical knowledge are key indicators for a resource class.

The categorization describes the uncertainty of a project. In the oil and gas industry, categories like proved, probable and possible or P90, P50, P10 or 1P, 2P, 3P or low, best high have been established. They are all used to express the uncertainty or a probability of a certain outcome, depending on the system and methodology that is applied.

Proved Reserves e. g. are quantities of petroleum, which by analysis of geoscience and engineering data can be estimated with reasonable certainty to be commercially recoverable from a given date forward, from known reservoirs and under defined economic conditions,



operating methods and government regulations. In a probabilistic approach the probability is at least 90% that quantities actually recovered will equal or exceed the P90 or 1P estimate.

Reasonable assumptions of uncertainties shall be consistently applied. But they highly depend on education, experience, and preference of the evaluators and on the availability and reliability of data and the interpretation of this data. Taking all this into account, it's therefore highly unlikely that different evaluators come to the same result. It is accepted practice to claim "material agreement" or "reasonable agreement" if the differences of independent external audits or reserves reviews and the companies evaluation are in a range of +/- 5% or in a range of +/- 10% respectively.

This paper concludes with a brief outlook on the United Nations Framework Classification UNFC 2009. This is a generic classification system which allows to compare hydrocarbon resources with mineral resources and even uranium. Projects to apply it to renewables and recipient reservoirs (CO₂ storage) are ongoing.

References:

SPE Petroleum Resources Management System (SPE PRMS 2007)

United Nations Framework Classification for Fossil Energy and Mineral Resources (UNFC 2009)



INTEGRATING ANTHROPOGENIC MATERIAL STOCKS AND FLOWS INTO MODERN RESOURCE CLASSIFICATION FRAMEWORKS

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Introduction

The growing global demand for raw materials over the past decades has fuelled concerns about rising prices and future availability of geogenic (primary) resources. To alleviate raw material criticality issues and, thus, the dependency on monopolistic supply structures, governments and institutions have been increasingly promoting improvements in resource efficiency as well as in the utilization of so-called anthropogenic (secondary) resources (e.g. recycling of waste) (e.g. EC, 2011). In this study anthropogenic resources are defined as “stocks and flows of materials created by humans or caused by human activity, which can be potentially drawn upon when needed” (Winterstetter et al., 2015b). However, while the exploration and classification of primary deposits is a well-established discipline, the knowledge on anthropogenic resource deposits and their availability for reuse and recycling is still very limited. To obtain a comprehensive overview of existing and potentially extractable anthropogenic resource inventories, it is therefore vital to provide a methodological framework for the evaluation and classification of anthropogenic materials. This would also facilitate decision-making for political and private business stakeholders.

Conceptual background

Starting in the early 18th century in Europe, first reflections on a more sustainable use of natural resources were primarily motivated by the perception of dwindling key raw material deposits, such as wood and coal (Carlowitz, 1713, Jevons, 1906). Considered as the precursors to modern resource classification systems, their common feature is managing scarce commodities by making potential resource extraction projects comparable for involved stakeholders. Since then, most major mining nations as well as economies strongly depending on resource imports have developed their own national classification codes. But when the mining industry has started becoming more and more of a global business from the 1990s on, increased efforts have been made to harmonize those codes to create transparency and comparability in reporting primary raw materials. After the Soviet Union’s collapse the German Government proposed a new classification system to the UNECE Working Party on Coal to compare the vast resources in the formerly centrally planned economies to those in the market economies (UNECE, 2013).

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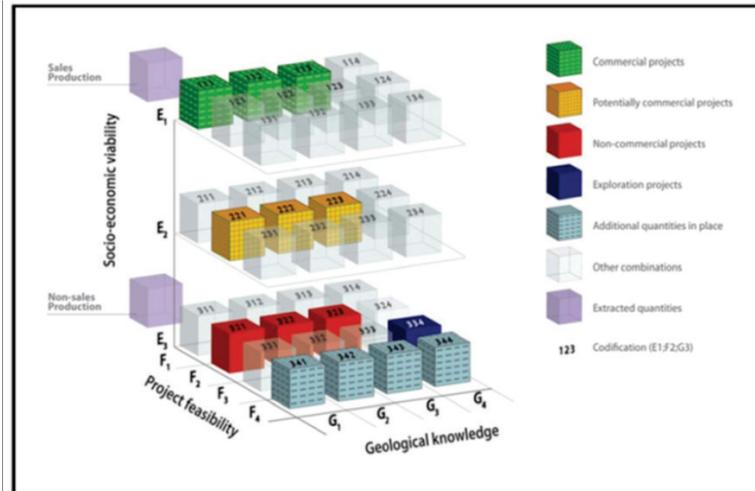


Figure 1: United Nations Framework Classification for Fossil Energy and Mineral Reserves and Resources 2009 (UNFC-2009). Reproduced courtesy of the United Nations Economic Commission for Europe (UNECE, 2010).

The United Nations Framework Classification for Fossil Energy and Mineral Reserves and Resources (UNFC) has been initiated by the UN Economic Commission for Europe, and was revised in 2009, today being known as UNFC-2009. Under UNFC-2009 quantities are classified on the basis of three fundamental criteria, namely “socioeconomic viability” (E1 – E3), “field project status and technical feasibility” (F1 – F4), and “knowledge on composition” (G1 – G4), with E1F1G1 being the best category (cf. Figure 1) (UNECE, 2010). The globally recognized United Nations Framework Classification for Fossil Energy and Mineral Reserves and Resources 2009 (UNFC-2009) offers a consistent classification system for various kinds of primary resources and related extractive activities, fulfilling both governmental as well as to a certain extent corporate stakeholders’ requirements. As the UNFC-2009 framework serves primarily classification purposes, it does not provide any criteria for the detailed evaluation steps of a mining project nor does it prescribe standardized methods to precisely distinguish between the different UNFC-2009 categories, e.g. “resources (E2)” and “materials without any prospects of future economic extraction (E3)” (UNECE, 2010).

Goals & Objectives

The goal of this study is to integrate anthropogenic resources into UNFC-2009, facilitating comparisons with geogenic resource inventories, and thus leading to better estimates of total stocks and recoverable fractions in consideration of various boundary conditions. This endeavour has been encouraged at the sixth session of the UNECE expert group on resource classification in April 2015 in Geneva (UNECE, 2015, Winterstetter et al. 2015c).

To show the UN framework’s applicability to anthropogenic resources in principle, an initial operative evaluation procedure has been developed for a case study on landfill mining (cf. Winterstetter et al., 2015a). To fine tune this first approach and to fit anthropogenic resources systematically into UNFC-2009, Winterstetter et al. (2015b) investigate factors influencing the classification of anthropogenic resources, on the one hand in comparison to geogenic

resources, and on the other hand, considering various types of anthropogenic resources. At present, we are working on applying this general methodology to mining anthropogenic materials from 1) an old landfill, 2) waste electrical and electronic equipment (WEEE) and 3) in-use wind turbines. Based on those three showcases, stringent criteria to distinguish between different UNFC-2009 categories for different kinds of anthropogenic resources are being developed.

Heterogeneity of anthropogenic resources: An attempted systemization (cf. Winterstetter et al. 2015b)

The human impact on production, consumption and disposal, combined with significantly shorter time spans of renewal were identified as major differences compared to the genesis of geogenic resources. To facilitate the classification of mining specific materials from various different and decentralized human-made sources, which is often linked to big technical and legal uncertainties, influencing factors can be systemized according to their role during the individual phases of resource classification. The prospection phase is determined by 1) the deposit's status of availability for mining, discriminating between "in-use stocks" vs. "obsolete stocks" and "waste flows", 2) by the specific handling and mining condition (e.g. mining a landfill for resource recovery purposes represents a "pull situation", while extracting materials from WEEE is regulated by laws and so a "push situation") and 3) the system variables. While the status of availability and the specific handling condition represent the preconditions for potential mining activities by defining the setting for the following classification, system variables (e.g. the set-up of E-waste collection systems or specific technical choices to extract materials from wind turbines) determine the amount of technically extractable materials.

When prospecting anthropogenic resource deposits, there can be two types of conditions: In a push situation, like in the case of WEEE flows, anthropogenic materials have to be treated (this may include material recovery to reduce costs) due to legal requirements, whereas in a pull situation the materials are mined only if the initial socioeconomic evaluation is positive and otherwise left untouched, like in the case of mining a landfill for resource recovery, which comes close to mining geogenic resources. In a push situation optimal solutions within the given legal framework are sought (Winterstetter et al., 2015b).

System variables also play a major role during the exploration phase. To account for different (possible) sets of system variable values scenario analysis can be used, to check, for instance, different technological treatment options. During the actual socioeconomic evaluation of resource extraction and valorization the modifying factors are investigated. Modifying factors, e.g. commodity prices, have an immediate impact on the project's socioeconomic viability and can potentially move the classification status of a given material deposit along the E-axis of UNFC-2009 from "non-commercial" to "potentially commercial" (resource) to "commercial" (reserve). They can hardly be influenced, but may change over time (Winterstetter et al., 2015b).

Table 1: Procedure for the classification of anthropogenic resources under UNFC-2009, based on Winterstetter et al. (2015b).

<u>Phases & UNFC-2009 axes</u>	<u>Influencing factors</u>	
Prospection	Availability	<ul style="list-style-type: none"> • In-use stock: Currently not available for mining (e.g. Nd in wind turbines), but at some point in the future • Obsolete stock: Potentially available for mining, sometimes even required (e.g. old landfill) • Waste flows: Treatment required (e.g. WEEE)
	Mining / handling condition	<ul style="list-style-type: none"> • Pull: Deposit can be mined • Push: Materials must be extracted from the deposit due to system constraints
Prospection (G-Axis) & Exploration (F-Axis)	System variables	System variables determine the amount of potentially extractable materials Different sets of system variables can be considered via alternative scenarios , but throughout a specific evaluation process, the system variables are exogenously given (e.g. composition, extraction technology, laws).
Socioeconomic evaluation (E-Axis)	Modifying factors	Modifying factors have a direct impact on the project's economics. They can hardly be influenced, but may change over time (e.g. commodity prices, treatment costs).
Classification	Combination of all criteria & classification under UNFC-2009	

Illustrating examples: E-Waste vs. old landfill vs. in-use wind turbines

Treating waste flows, such as WEEE, typically represents a push situation. The management of WEEE flows in the European Union is mainly regulated and driven by laws, in particular by the EU directive 2012/19/EU, determining the annual collection, reuse and recycling targets. (Directive, 2012). So here the question appears on how to treat WEEE in a socioeconomically optimal way within the given legal constraints. The amount of potentially extractable materials contained in a WEEE flow is influenced by system variables, such as the waste flow's volume, the product type and size, the share of usable materials and potential hazardous substances, as well as the recyclability of the specific product type. The technical and project feasibility of mining WEEE is mainly determined by the set-up of the collection and recycling system. Apart from collection, the recycling chain for WEEE consists of further succeeding steps, i.e. sorting, dismantling, pre-processing, and end-processing. Various different treatment technologies are available, which can (potentially) address the specific needs of each product group. Methods with higher recovery efficiencies are more likely to be selected if markets for the output fractions exist and if expected price levels are high enough to justify higher treatment costs or if alternative treatment and disposal costs can be avoided, i.e. if modifying factors with direct impact on the economics are positive.

Mining stocks, such as old landfills, can either represent a push or a pull situation. In a pull situation, mining an old landfill requires positive socioeconomic prospects either for a private investor or a public entity, since the alternative of mining a landfill is regulated aftercare, where

the closed landfill is simply left untouched. If the landfill represents an imminent threat to the environment, e.g. to groundwater, authorities will oblige the former landfill operator to act, i.e. the pull situation turns into a push situation, similar to mining a waste flow. When classifying a landfill-mining project in a pull situation, system variables, such as the landfill's location and size, its ash and water content, the share of valuables, combustibles, non-recyclables or even hazardous substances, and the contamination of the fine fraction, are considered as given for a certain scenario and the main focus is set on the modifying factors. Modifying factor with immediate impact on the economics differ, however, according to the chosen stakeholder perspective. A private investor is only interested in direct financial effects, while non-monetary effects tend to be ignored, if they are not internalized in form of subsidies. A public entity, on the other hand, is more interested in long-term effects, such as the elimination of a source of local soil and water pollution or the avoidance of long-term landfill emissions (Winterstetter et al. 2015a).

In-use stocks of NdFeB materials in wind turbines are currently not available for mining, but will become waste flows in the future. Most probably the recycling of wind turbines will represent a push situation, as permanent magnets and / or NdFeB materials will have to be extracted from wind turbines. To classify potential future mining projects of in-use stocks, gaining in-depth knowledge on a deposit's resource potential has priority over the following socioeconomic evaluation. The economics are obviously linked to high uncertainties, due to not (yet) existing commercially proven technologies. The in-use stock's composition and its potentially extractable share of materials within the defined boundary conditions is determined by type, size, location and the total number of the wind turbines and the contained permanent magnets, as well as the ease of dismantling wind turbines. Uncertainties arise from the technical feasibility of recycling permanent magnets, since manufacturers typically do not publish detailed reports or data on their individual recycling processes (Gattringer, 2012). Therefore, information on recovery efficiencies, investment and operating costs can practically not be found. The choice of specific methods and technologies for processing and separating rare earths from the magnets, influences the final amount of recovered materials, as well as investment and operating costs. Similar to WEEE, costly technologies are more likely to be chosen if modifying factors are positive, e.g. if expected price levels for output materials justify higher treatment costs.

Conclusions & Outlook

Factors influencing the classification of anthropogenic material deposits can be divided into the status of availability (in-use or obsolete stocks, waste flows), mining / handling condition (push vs. pull situation), system variables and modifying factors, according to their role during the individual phases of resource classification (prospection, exploration and socioeconomic evaluation). Exemplarily, the influencing factors of mining anthropogenic resources from an old landfill (obsolete stock), from E-waste (waste flow) and wind turbines (in-use stock) were analysed. In order to obtain a comprehensive overview of existing and potentially extractable anthropogenic resource inventories and to allow the full integration into UNFC-2009, specific guidelines are still to be defined and need to be demonstrated via case studies, to account for the heterogeneous nature of anthropogenic resources. This will facilitate decision-making for political and private business stakeholders and allow for meaningful comparisons between anthropogenic and geogenic mineral resources, promoting the efficient use of resources.

Acknowledgements

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1.2 Session B: Exploration of Anthropogenic Resource Flows and Stocks Using Material Flow Analysis



LESSONS FROM MATERIAL FLOW ANALYSIS

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Introduction

In the past 10-15 years, material flow analysis (MFA) has become a central tool in industrial ecology for understanding the ways in which resources flow, and the magnitudes of those flows, throughout human society. A few examples illustrate lessons that have been learned from MFA.

Methodology

The basic concept of MFA is to define the major nodes in the cycle and then to quantify the flows to and from those nodes. The generic form of the cycle often used is shown in Figure 1.

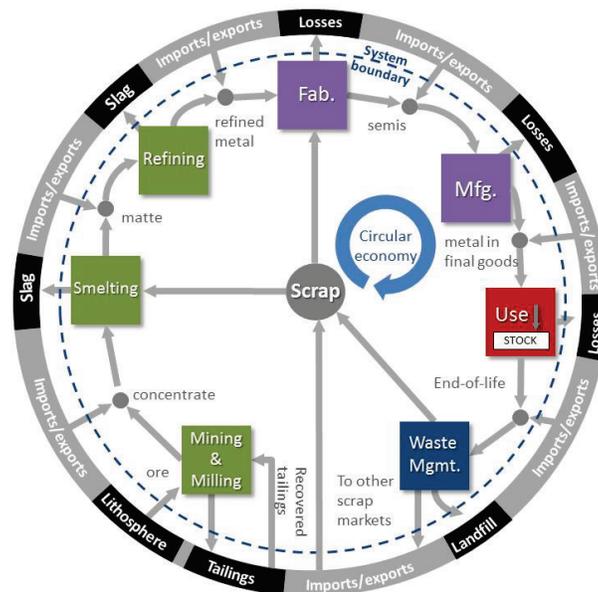


Figure 1 The generic material flow cycle for a metal

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Lessons from MFA Research

An example MFA result is shown in Figure 2. In year 2000 nearly 20 Tg of stainless steel were initially produced, of which nearly 15 Tg entered into use. The majority of the latter amount constituted in-use stock additions. Scrap was efficiently reused, so only 41% of stainless steel that was produced was from virgin stock.

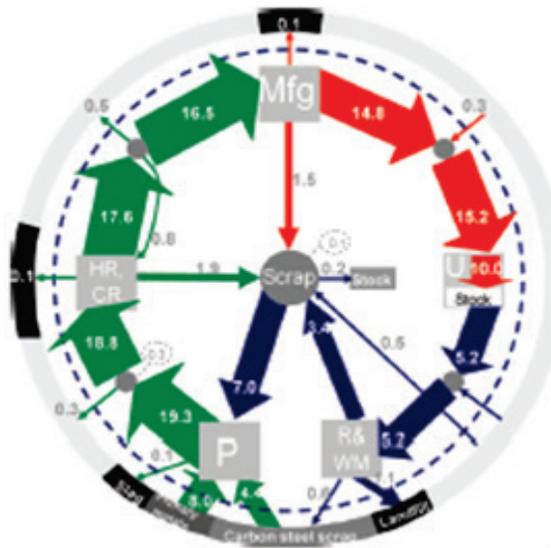


Figure 2 The global stainless steel cycle, year 2000. Flows are in million metric tons (Tg) (Reck et al., 2010).

When several related cycles are analysed, the results can sometimes provide information not derivable from an individual cycle. Figure 3 shows an example, in which the per capita use of seven metals in different countries was used to show that in most cases countries that rank above the global average in per capita use of any one metal are very like to be above the global average for all.

A time series of MFA cycles can generate estimates of in-use stock, as shown in Figure 4 for building construction materials in Japan. The case has often been made that it is the services provided by in-use stock that drive the consumption of resources, so stock determinations such as this have become a very important output of MFA analysis.

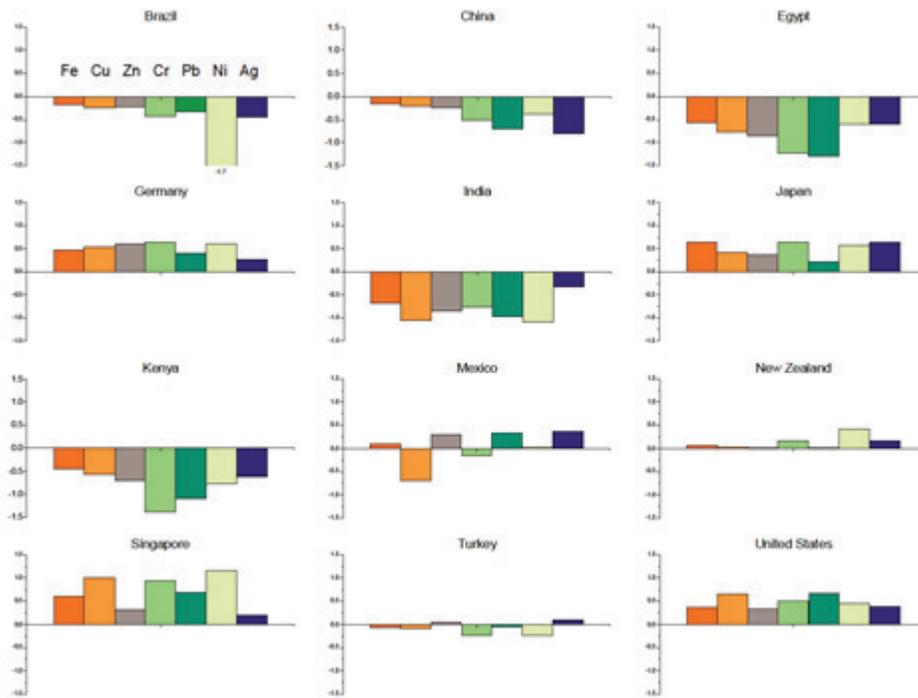


Figure 3 Metal use spectra for a selection of countries around the world. The dimensionless units are logarithm of per capita flow into use for each metal relative to the world average (Graedel and Cao, 2010).

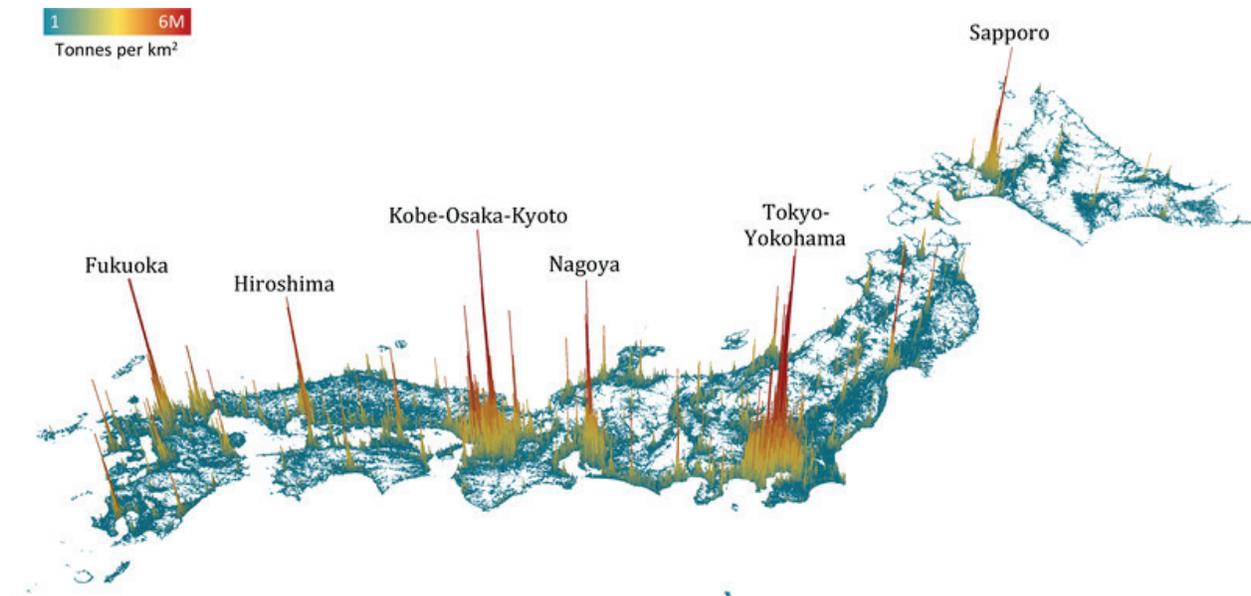


Figure 4 Country-wide spatial distribution of the material stocked in buildings, 2009. (Tanikawa et al., 2015)

Summary and Prospects

The examples above demonstrate the potential of MFA studies to inform our understanding of the human use of materials. Much has indeed been accomplished in this regard. However, as Figure 5 demonstrates, little or no MFA information exists for many elements of interest. Plenty of work remains to be done if we are to fully comprehend the story of human resource use.

Global Level

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
1	1 H		3					8										2 He
2	3 Li	4 Be											5 B	6 C	7 N	8 O	9 F	10 Ne
3	11 Na	12 Mg											13 Al	14 Si	15 P	16 S	17 Cl	18 Ar
4	19 K	20 Ca	21 Sc	22 Ti	23 V	24 Cr	25 Mn	26 Fe	27 Co	28 Ni	29 Cu	30 Zn	31 Ga	32 Ge	33 As	34 Se	35 Br	36 Kr
5	37 Rb	38 Sr	39 Y	40 Zr	41 Nb	42 Mo	43 Tc	44 Ru	45 Rh	46 Pd	47 Ag	48 Cd	49 In	50 Sn	51 Sb	52 Te	53 I	54 Xe
6	55 Cs	56 Ba	57-71	72 Hf	73 Ta	74 W	75 Re	76 Os	77 Ir	78 Pt	79 Au	80 Hg	81 Tl	82 Pb	83 Bi	84 Po	85 At	86 Rn
7	87 Fr	88 Ra	89-103	104 Rf	105 Db	106 Sg	107 Bh	108 Hs	109 Mt	110 Ds	111 Rg	112 Uub	113 Uut	114 Uuq	115 Uup	116 Uuh	117 Uus	118 Uuo
	Lanthanide Series			57 La	58 Ce	59 Pr	60 Nd	61 Pm	62 Sm	63 Eu	64 Gd	65 Tb	66 Dy	67 Ho	68 Er	69 Tm	70 Yb	71 Lu
	Actinide Series			89 Ac	90 Th	91 Pa	92 U	93 Np	94 Pu	95 Am	96 Cm	97 Bk	98 Cf	99 Es	100 Fm	101 Md	102 No	103 Lr

Figure 5 Elements for which global cycles have been derived (Chen and Graedel, 2012).

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ANTHROPOGENIC ALUMINIUM RESOURCES IN AUSTRIA AND SELF-SUPPLY POTENTIALS

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Introduction

Metals have ever played a vital role for technological and economic progress in society. In the past 60 years the global production of crude steel increased by factor 8, while the production of Al increased by factor 35. Especially due to light weight construction in vehicles, modern constructions in buildings as well as a flexible material in engineering and packaging, Al use increased considerably after 1960. Even though primary production is still the major supplier of Al on a global scale (45% China, 10% CIS, 9% Europe, 36% rest of the world) (EAA, 2012), secondary production has become more important, especially for countries with limited access to primary resources and high energy costs. After years of declining primary production in Europe, unwrought metal supply from intra Union primary production is now at a level of 14%, while secondary production and net-metal-imports have been increasing to 35% and 51% respectively. In Austria, a country without primary Al production, around 2/3 of unwrought metal demand is supplied by secondary production and around 1/3 from net-metal-imports (cf. Figure 1).

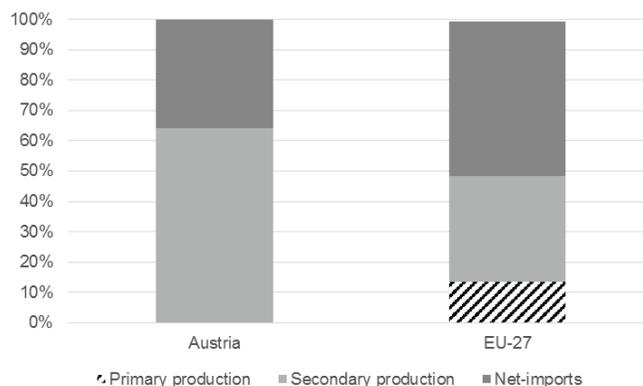


Figure 1 Sources of unwrought Al supply. Comparison between Austria (2012 data) and the EU-27 (2013 data)

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Since Al is predominantly used in products with long lifetimes, a considerable anthropogenic stock of Al has been built up as a consequence of the continuously increasing Al demand over the last decades. About 75% of all Al ever produced globally is expected being still in use (Bertram et al. , 2009). Therefore enhanced use of secondary raw materials is of common interest, from an industry perspective but also from an ecological and resource management perspective. In the present study, the quantitative opportunities of Al recycling in a national system are shown using a dynamic material flow model of Austrian Al flows from 1964-2050. Based on historical data current in-use stocks are calculated and forecasts of future in-use stock development and old scrap generation are produced based on assumptions about future final Al demand. Furthermore, the potential of Al self-supply (Al final demand can be satisfied solely based on domestic Al scrap supply) on the national level is evaluated. Finally, Al scrap flows are also discussed with respect to material quality, distinguishing between cast and wrought alloys. This allows for detecting future quality-related supply constraints and to evaluate the potential role of sorting technologies in this respect.

Methodology

An input driven dynamic material flow model is applied to calculate current in-use stocks based on historical production and trade data between 1964 and 2012. A special focus is put on the calibration and cross-checking of modelled stocks and flows with independent bottom-up estimates. Starting with data on the production of semi-finished products, all subsequent stages of the Al lifecycle are included in the model (cf. Figure 2). Considering national production, processing and manufacturing as well as the associated foreign trade flows, together with foreign trade of final products (e.g. cars etc.), the annual final Al demand with respect to the six most common sectors (Transport, Buildings and Infrastructure, Electrical Engineering, Mechanical Engineering, Electrical Engineering, Consumer products and Packaging) is calculated. Annual inputs into each sector are finally multiplied with sector-specific probability functions of discard in order to calculate old scrap generation in a given year from the inflows of the previous years. In-use stocks are finally derived from the annual delta between inflows and outflows of every sector. Cross-checks of the annual scrap generation with national secondary production in consideration of scrap trade and collection and processing losses are made to evaluate the plausibility of the model results in the light of independently derived estimates.

In order to estimate future development of in-use stocks and old scrap generation, current in-use stocks (derived from the historical model described above) are combined with forecasts on future final Al demand. For the sectors Transport, Buildings and Infrastructure and Electrical Engineering a stock-driven approach (i.e., the future stock development is pre-defined and thereby drives Al consumption) is used, for the remaining sectors Mechanical Engineering, Consumer and Packaging an input-driven approach (i.e. Al consumption of the different sectors is pre-defined in a scenario) is used in order to estimate future in-use stock and old scrap generation for each sector (Buchner et al. , 2015c).

Future self-supply is calculated considering two different perspectives. On the one hand, modelled old scrap generation is compared with expected future quantities of secondary Al

production in order to analyse self-supply at the industry level. On the other hand, modelled old scrap generation is compared with expected future final AI demand in order to analyse (theoretical) self-supply at the product consumption level.

In order to analyse self-supply with respect to wrought and cast alloy recycling (an excess of alloying elements could potentially limit direct recycling of cast alloys) a historical split of wrought and cast alloys in European product shipments (IAI, 2011) is applied to the modelled inflows. Future splits on wrought and cast alloys are kept at the level of 2012 values. Taking the current recycling practice of end-of-life vehicle management, where mostly no separation between cast and wrought alloys is conducted, a comparison between future final AI demand and old scrap generation regarding wrought and cast alloys is conducted. The comparison is based on theoretical inflows and outflows neglecting collection, processing and melting losses as well as product exports.

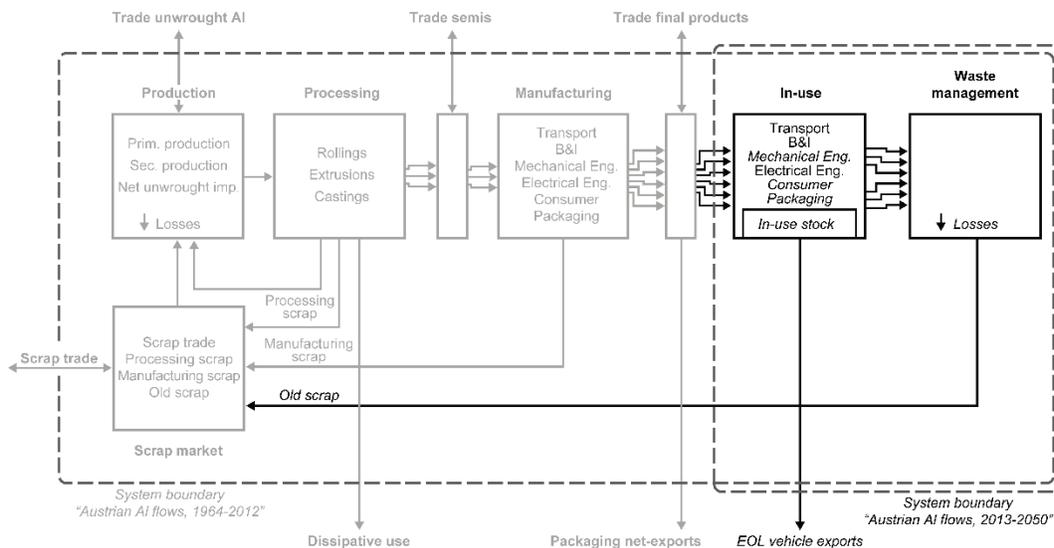


Figure 2 National dynamic material flow model. System definition after Buchner et al. (2015a)

Results

About two-thirds of all anthropogenic AI is currently (2012) stored in buildings and infrastructure (159 kg/cap) and transport applications (109 kg/cap). Lower AI amounts are retained in engineering applications (66 kg/cap), consumer products (14 kg/cap) as well as reusable packaging material (5 kg/cap.), which yields a total stock of nearly 360 kg/cap. Regarding final AI demand, flows have been slightly increasing over time (except for the period of the financial crisis) to a level of 25 kg/cap in 2012. Per capita final AI demand over time as well as in-use stock development and old scrap generation are shown for selected years in Table .

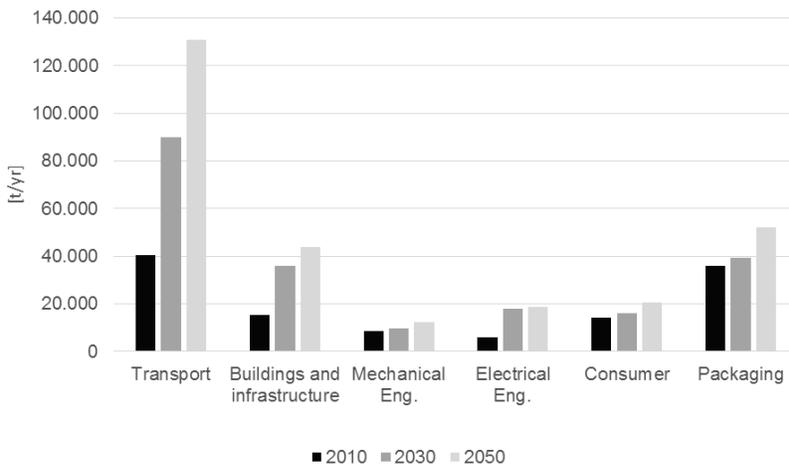


Figure 3 Current and future old scrap generation subdivided by in-use sectors (based on (Buchner et al. , 2015b))

Table 1 Historical and forecasted development of most important model outputs (Total in-use stock, final AI demand and old scrap)

	1982	1992	2002	2012	2030	2050
Total in-use stock [kg/cap.]	94	150	280	360	440	530
Final AI demand [kg/cap.]	6.9	17	25	25	28	35
Old scrap [kg/cap.]	3.5	6.1	8.8	12	24	31

Future self-supply regarding industrial AI demand (secondary production) is estimated assuming a +2% and a +3% compound annual growth rate (CAGR) for secondary production. The self-supply level is currently around 15% and is not expected to increase (substantially) in the future even in case of the lower growth (2%) scenario. Thus, in case of a CAGR of 3%, self-supply is expected to decrease. . These results highlight the crucial importance of CAGR with respect to the evaluation of future industrial AI self-supply. Enhanced recycling, which is modelled by recycling rates of at least 90% in all in-use sectors, could contribute about 3-4% to industrial self-supply (Figure 4a).

From a final demand perspective self-supply remains quite stable in the range of 40%, if no changes in system regarding scrap collection and final AI demand occur. Through enhanced national recycling an increase of self-supply by nearly 10% could be achieved. If final AI demand for the future is fixed to current levels (constant final AI demand) an increase of self-supply up to 70% is observed (Figure 4b).

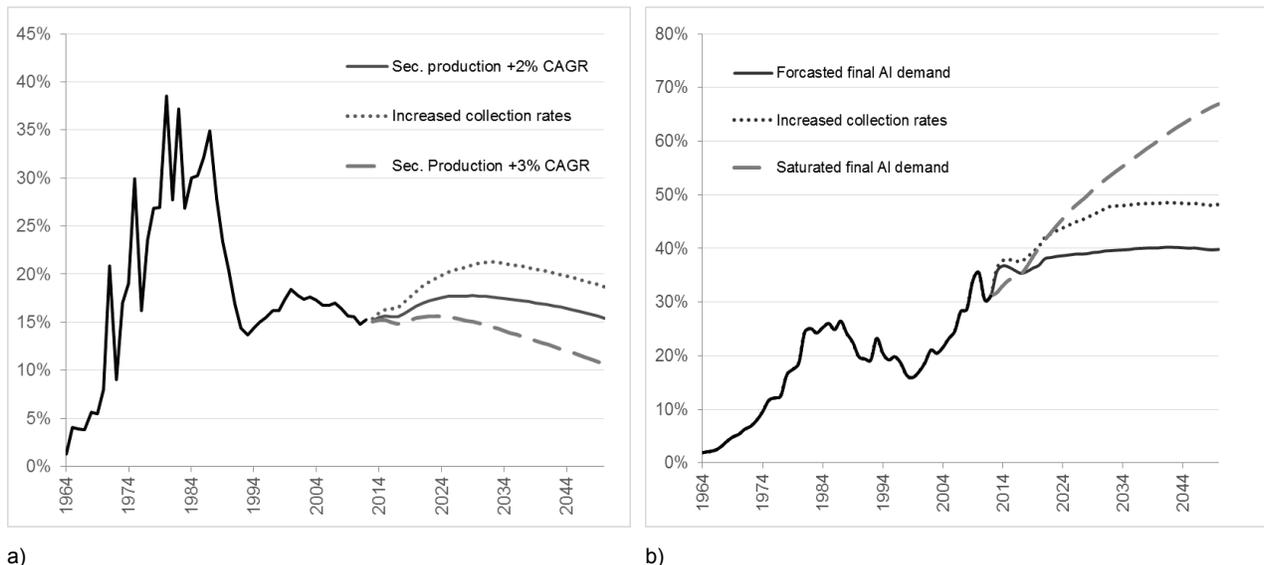


Figure 4 Calculated self-supply of Austrian AI industry (Figure 4a) and theoretical self-supply with respect to final AI demand (Figure 4b)

A comparison of future final AI demand and old scrap generation with respect to wrought and cast alloys is shown in Figure 5. Total old scrap generation and total final AI demand (all in-use sectors) is shown in Figure 5a, final demand and old scrap generation of the Transport sector is shown in Figure b. If no separation of wrought and cast alloys from vehicles is conducted, an excess of cast scrap (mixture of cast and wrought alloys) in the whole system is soon to be expected. Considering an increasing share of (wrought) AI components in cars, this situation might be further exacerbated, if no separation techniques are applied in vehicle recycling. However, it should be emphasized that currently scrap markets are not locally confined but open, which might decrease the pressure to separate cast and wrought alloys (either at the source or using processing technologies) and even lead to a higher level of scrap trade. Nevertheless, the use of low-quality scrap is ultimately dependent on primary AI production to dilute unwanted constituents. Hence, if secondary AI production shall continue to gain importance, scrap quality aspects will become a crucial issue, sooner or later.

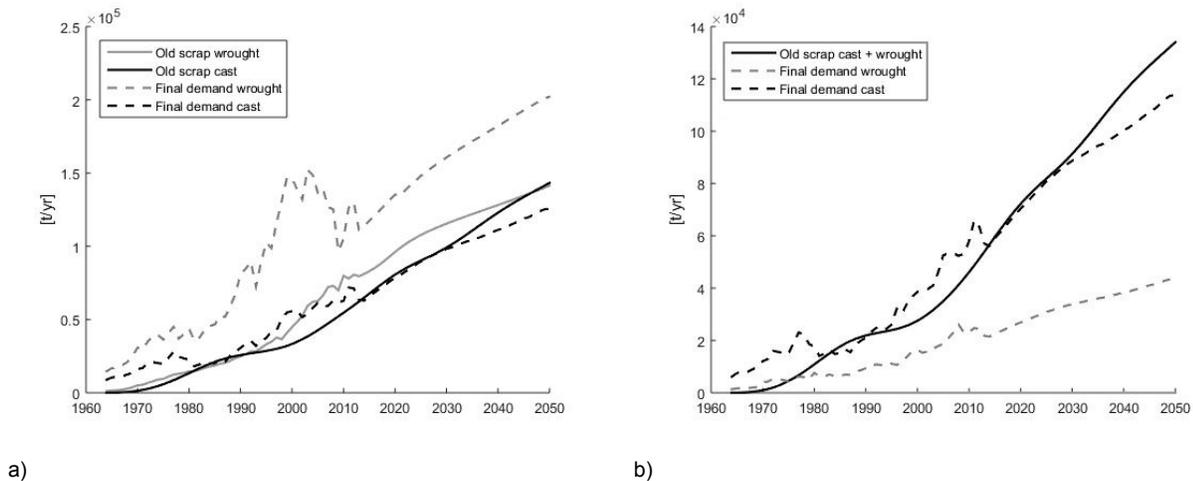


Figure 5 Old scrap generation with respect to cast and wrought alloys. Total final AI demand and old old scrap generation (Figure a) and final AI demand and old scrap generation of the Transport sector (Figure b)

Conclusions

Based on the analyses performed in this study, it could be shown that there are opportunities and limits for utilizing anthropogenic Al resources. Strategies for the sustainable management of Al resources need to consider quantitative as well as qualitative characteristics of the Al resource system, in particular with respect to the implementation of new recycling schemes and their potential effect on secondary raw material supply. The potential for Al self-supply of the Austrian industry is low given no extreme developments. Also from a final demand perspective, self-supply might hardly reach levels above 50%. Thus, high levels of self-supply can only go along with constant or decreasing Al consumption. This is also highlighted by the respective demand assumptions being the dominant factors for the estimated levels of self-supply. However, further analysis of the national material flow system, especially with respect to material quality aspects, is required in order to provide robust recommendations on how to optimize future recycling systems. In any case, an excess of certain old scrap qualities (cast alloys mixed with wrought alloys) might be unfavorable, in terms of raw material availability as well as in terms of resource efficiency.

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ADDRESSING THE ISSUE OF UNCERTAINTY IN MFA: AN APPLICATION TO RARE EARTH ELEMENTS IN THE EU 28

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Uncertainty in MFA

The most important (and time-consuming) step in material flow analysis (MFA) is data mining. In the case of, e.g., rare earth elements (REE), sources of data are multiple and of various natures. Sources may include import-export data (e.g., EUROSTAT, GTA, ...), specialized reports (Castilloux, 2014; ROSKILL, 2015; USGS, 2014, ...), measurements of concentrations in products (Westphal and Kuchta, 2013; Rotter et al., 2013, ...), literature data, expert information, etc. Different types of information have different associated uncertainties. While repeated measurements may be treated in a statistical framework to provide indicators of parameter variability (means, standard deviations, ...), other tools are better suited for addressing uncertainty with respect to, e.g., expert information or scarce data. Such tools include possibility theory (Dubois and Prade, 1988) or the belief functions of Dempster-Shafer (Shafer, 1976). As shown in Dubois et al. (2014), such tools often apply more naturally to the representation of the type of information that is available in typical MFA studies.

Consider for example the data of Westphal and Kuchta (2013), who measured the neodymium contents of magnets in WEEE (waste electric and electronic equipment); i.e., earphones from various manufacturers. Figure 1a (Guyonnet et al., 2016) shows these data in the form of a cumulative frequency plot. In a classical probabilistic (or Bayesian) approach, one option could be to fit a uniform probability distribution to this data, as illustrated in Figure 1b. But it can be argued that these data are scarce and therefore that had the authors obtained additional data, a different distribution might have emerged. In order to account for the relative paucity of these data, a different mode of information representation can be adopted. The possibility distribution (or “fuzzy set”; Dubois and Prade, 1988) of Figure 2a conveys the information that: (i) the neodymium content of earphones lies between 20 and 140 mg (this interval is called the “support” of the distribution) and (ii) the most “likely” value is 80 mg (this value is called the “core” of the distribution). In terms of probabilities, this information translates into a “family” of cumulative probability distributions depicted in Figure 2b, where the data of Westphal and Kuchta (2013) has been included. The idea conveyed here is that the “true” distribution lies somewhere between the upper and lower limit distributions in Figure 2b, but that available information does not allow the selection of a specific distribution.

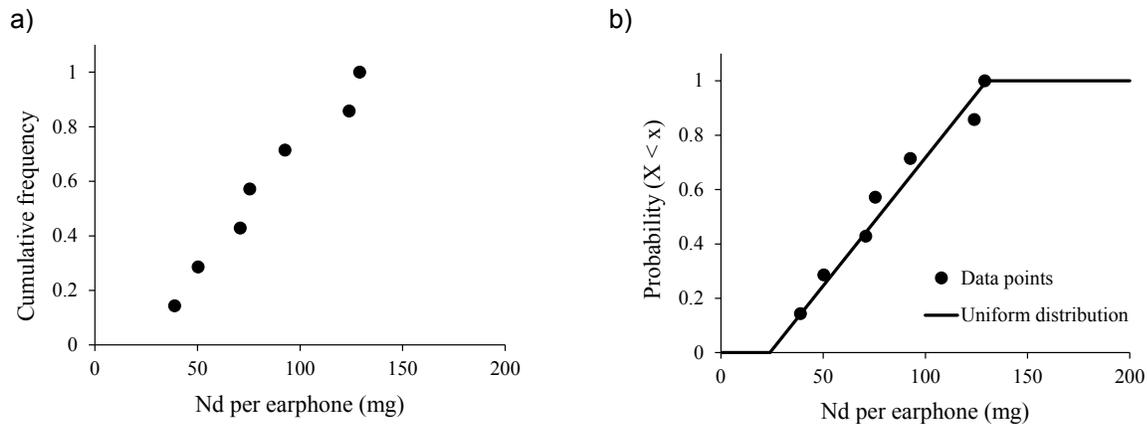


Figure 1 Cumulative frequency distribution of Neodymium content of earphones (a) and best-fit uniform distribution (b). Data from Westphal and Kuchta (2013)

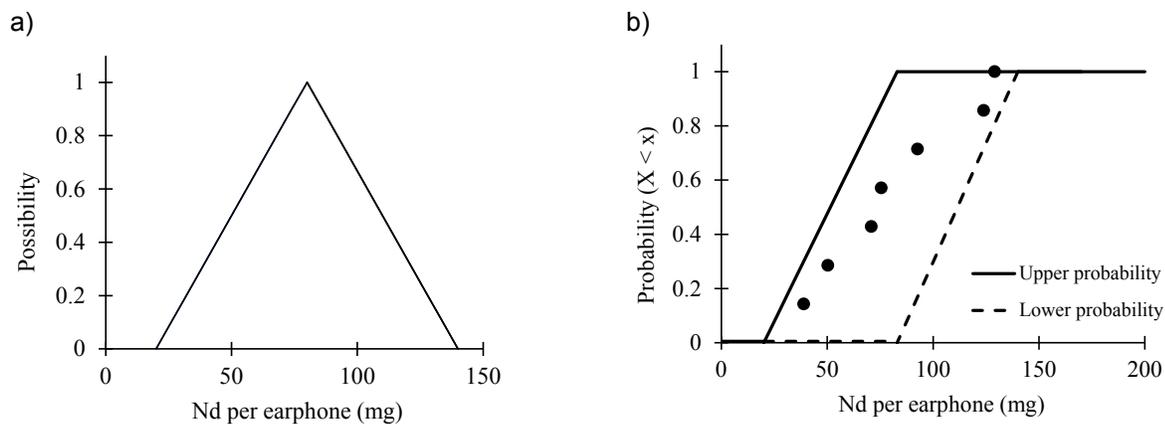


Figure 2 Possibility distribution for Neodymium content of earphones (a) and corresponding family of cumulative probability distributions (b). Data from Westphal and Kuchta (2013)

Dubois et al. (2014) propose a methodology for performing data reconciliation in MFA under fuzzy constraints such as those depicted in Figure 2a. The general principle is illustrated below using the trivial (and theoretical) case of a single process with a flow in and a flow out (and no stock). Figure 3 shows the fuzzy sets used to convey available information relative to both flows. The reconciliation methodology consists in identifying the maximum level of possibility (noted α^*) such that the mass balance equation (here: flow in = flow out) is satisfied, while constraining the reconciled flows within the boundaries of the initial data. As seen in Figure 4, these conditions are satisfied for flow values that fall within the area defined by the intersection between the inflow and outflow. The resulting reconciled flow has a core (most likely value) of 106 tons and a support (outside which flows are considered very unlikely) on the interval [90; 120]. The maximum level of possibility (α^*) is 0.4; i.e., an indicator of the reliability of the reconciliation. A value $\alpha^* = 0$ indicates that it is not possible to reconcile the data and therefore that the information on flows (or the model itself) need to be reassessed. This is in contrast with the

least-squares reconciliation method, performed under the assumption of Gaussian distributions for the flows, since these distributions are defined over the interval $[-\infty; +\infty]$ and therefore a solution can generally be found, albeit in areas of very low “probability”.

In real MFA studies, the number of constraints to be satisfied is much larger than in this simple illustration. Maximum likelihood must therefore be found considering the mass balance equations for each process while constraining all reconciled flows to the boundaries of the input information. Dubois et al. (2014) present an application to copper flows in the Australian economy, which may serve as a benchmark for testing reconciliation algorithms.

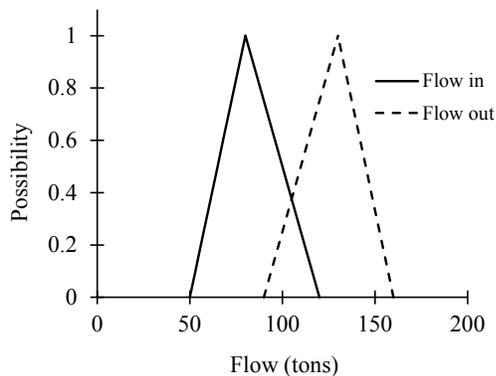


Figure 3 Schematic illustration of the principle of fuzzy reconciliation for the case of one process with an inflow and an outflow

Rare earth elements in the EU-28 in 2010

Guyonnet et al. (2015) used MFA to obtain estimates of flows and stocks of certain REE in the EU-28 in 2010. They considered rare earths that are used in applications that are important for a low-carbon energy transition and/or have a high recycling potential: NdFeB magnets (Pr, Nd, Dy), NiMH batteries (Pr, Nd) and fluorescent lamp phosphors (Eu, Tb, Y). The data mining phase, using multiple sources of data, provided initial estimates of flows that were organized as lower, best (or “preferred”) and upper estimates. The methodology of Dubois et al. (2014) for data reconciliation was then applied. Figure 4 shows the Sankey diagram of best estimate flows for Nd in permanent NdFeB magnets. Applications that use such magnets and that were considered in the analysis include hybrid, electric and non-electric vehicles, hard disk drives, wind turbine generators, headphones and earphones, etc. (see Guyonnet et al., 2015). In this example, in-use stock of Nd in 2010 in Europe was estimated as 16000 tons, a number that can be compared to the global in-use stock of Nd in permanent magnets estimated by Du & Graedel (2011) for year 2007: 62600 tons. The flow into use of Nd is around 1230 tons, a number that is three times larger than the estimate of Rademaker et al. (2013) for year 2011 (400 tons). The discrepancy is explained by the fact that Rademaker et al. (2013) did not consider non-electric vehicles in their analysis. As indicated in Figure 4, there was no recycling of Nd from NdFeB magnets in 2010.

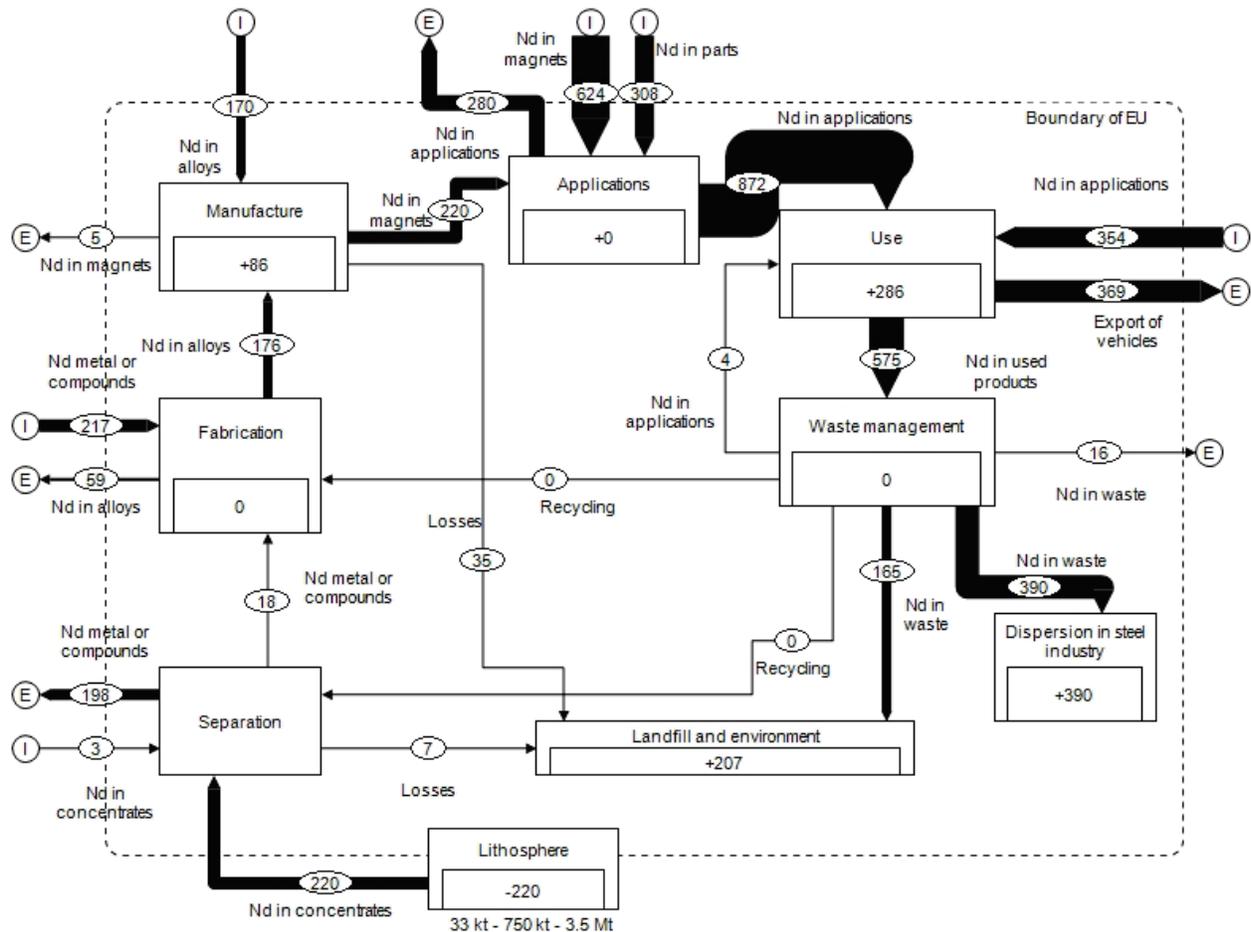


Figure 4 Sankey diagram for Nd in permanent magnet applications in 2010 in the EU. Values in tons of Nd metal unless indicated otherwise (Guyonnet et al., 2015)

Table 1 shows the uncertainty ranges obtained from applying the data reconciliation methodology of Dubois et al. (2014) for several applications. Unlike previous MFA studies where uncertainties are usually reported as mean values and standard deviations, uncertainties in Table 1 are expressed as best estimates and ranges (min-max) around these estimates.

Table 1 Estimated uncertainty ranges for several REE flows

	Manufacture to Use			Imports to Use			Use to Waste			Waste to L&E		
	Min	BE	Max	Min	BE	Max	Min	BE	Max	Min	BE	Max
Nd magnets	780	872	960	320	354	390	520	576	630	150	165	180
Nd batteries	28	35	46	75	84	95	64	69	71	52	55	57
Tb phosphors	8	14	16	10	21	30	9	11	13	9	11	13

Notes: BE = best estimate; L&E = Landfill & Environment; Values in tons metal

Conclusions

A methodology has been proposed for representing uncertainty in MFA and for performing data reconciliation. With respect to the example application presented above, it should be recognized that similar results would have been obtained if the “preferred” estimates had been considered as “mean values” and standard deviations had somehow been computed within the intervals defined by the lower and upper estimates. The principal merit of the proposed methodology is to provide a framework whereby the information that is typically available in MFA can be represented in a manner that is more consistent with that information than by using means and standard deviations, the identification of which requests some form of prior statistical analysis

Acknowledgments

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CONSIDERATIONS OF RESOURCE QUALITY IN MFA

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Abstract

This contribution focuses on resource quality of metals used by humans. Metals are rarely found in a pure or elemental form in nature, and also in final products, metals are mainly used in alloyed form. The amounts and forms of metals change throughout their life cycle, which in turn shapes future primary and secondary resource qualities and quantities: While ore grades tend to decline, often resulting in higher energy amounts needed to extract the metal, the availability of secondary resources tends to increase, offering a potential to save energy through recycling. However, this potential can only be exploited if the old scrap can be used to produce new goods. In practice, old scrap often includes a variety of mixed alloys and impurities, which can lead to challenges in recycling. In order to anticipate potential problems and identify or evaluate strategies related to resource quality, it is essential to understand the linkages between different metal cycles. These linkages can be described in quantitative terms using multi-layer MFAs. Here, we use case studies for aluminium to illustrate how MFA can be used for this purpose. The case studies demonstrate that the aluminium cycle needs to fundamentally change from a cascading use towards a closed loop in order to make use of the increasing amount of old scrap.

1 Introduction

Resource quality is a relative term that needs to be defined by the resource's usefulness for specific applications. Humans use metals for their distinct and specific properties, such as hardness, ductility, tensile strength, density, melting point, heat conductivity, or electrical conductivity, all of which are essential to build the diverse products we use in our daily lives. The properties of these resources can be changed to fit the specific needs by cleaning (refining) or by mixing (alloying/blending or mechanically joining) them.

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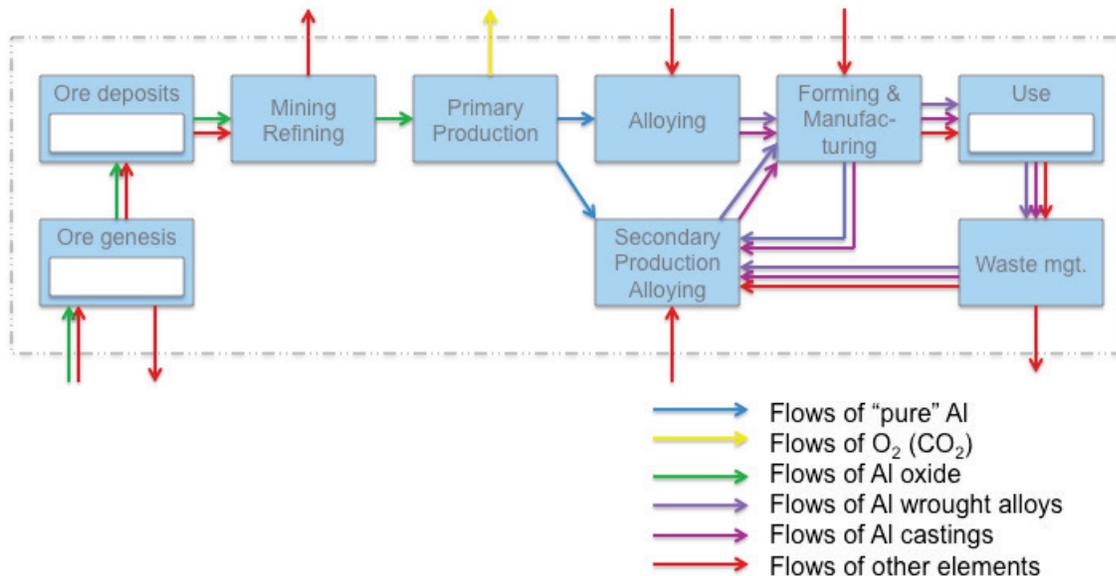


Figure 1 Linkages of elements in the aluminium cycle

The metal cycles are linked with each other throughout the life cycle, as illustrated in Figure 1 for the case of aluminium. A first concentration process is taking place in the genesis of ores. In the case of bauxite, the main aluminium ore, progressive dissolution of silica from clays in wet soils, mainly in tropics, eventually turn the kaolinite clay, $\text{Al}_2\text{Si}_2\text{O}_5(\text{OH})_4$, into gibbsite $\text{Al}(\text{OH})_3$. These soils eventually become bauxite. The bauxite mined contains about 45-60% aluminium oxide, along with various impurities such as sand, iron, and other metals, which are removed in a refining process. The refining process may also be a source for additional useful resources: in the case of aluminium, gallium may be recovered as a by-product of bauxite refining. The resulting alumina, Al_2O_3 , is used in as raw material in primary aluminium production, where the oxygen is removed by binding it with carbon to form carbon dioxide.

Aluminium from primary production includes only very small amounts of impurities, and is alloyed with other elements, such as copper, manganese, silicon, magnesium, or zinc in order to achieve the specific properties required. Two different types of alloy types are produced: (i) wrought alloys, used for rolled and extruded products, include a wide variety of combinations of alloying elements, but these are generally added in relatively small amounts, and (ii) casting alloys, used to produce castings, generally include large amounts of all kinds of alloying elements. Aluminium semi products, often in combination with other materials, are used in manufacturing to produce final products, such as cars, buildings, or cans. At the end of their service life, aluminium old scrap is recovered from most applications. The collected scrap fractions often include a mix of different alloys and may also include impurities from imperfect separation in waste management. Secondary aluminium production uses a mix of old scrap, new scrap, primary aluminium, and alloying elements to produce wrought products and castings. While new scrap of wrought aluminium is often used to produce wrought products (closed alloy loops), old scrap of wrought aluminium is often used to produce castings, which are more tolerant for additional materials (cascading use). Castings scrap is usually exclusively used to produce castings. Aluminium refining is very limited due to its thermodynamic properties.

2 Resource quality in primary production

For most metallic minerals there are long-term declines in average ore grades processed (Mudd 2010). This ore grade decline can be explained by increasing mine production and the fact that mining tends to take place in deposits with highest ore grades. The ore grade decline, however, varies a lot depending on the mineral. While the ore grades of gold, copper, nickel, and uranium have declined orders of magnitude over the past century, ore grade decline of bauxite has been moderate. As a result of the declining ore grade, the extraction of the metals from the ore requires higher energy and water consumption, and results in larger amounts of greenhouse gas emissions and waste being produced per unit of metal recovered (Norgate and Haque 2010).

3 Resource quality in secondary production

The large energy investments into the production of aluminium make this metal highly attractive for recycling. Aluminium recycling currently occurs in a cascading fashion, where secondary castings, used in a limited number of applications in vehicle engine parts, absorb most of the end-of-life scrap. Vehicles therefore form the bottom reservoir – and thus the bottleneck – of the current aluminium cascade.

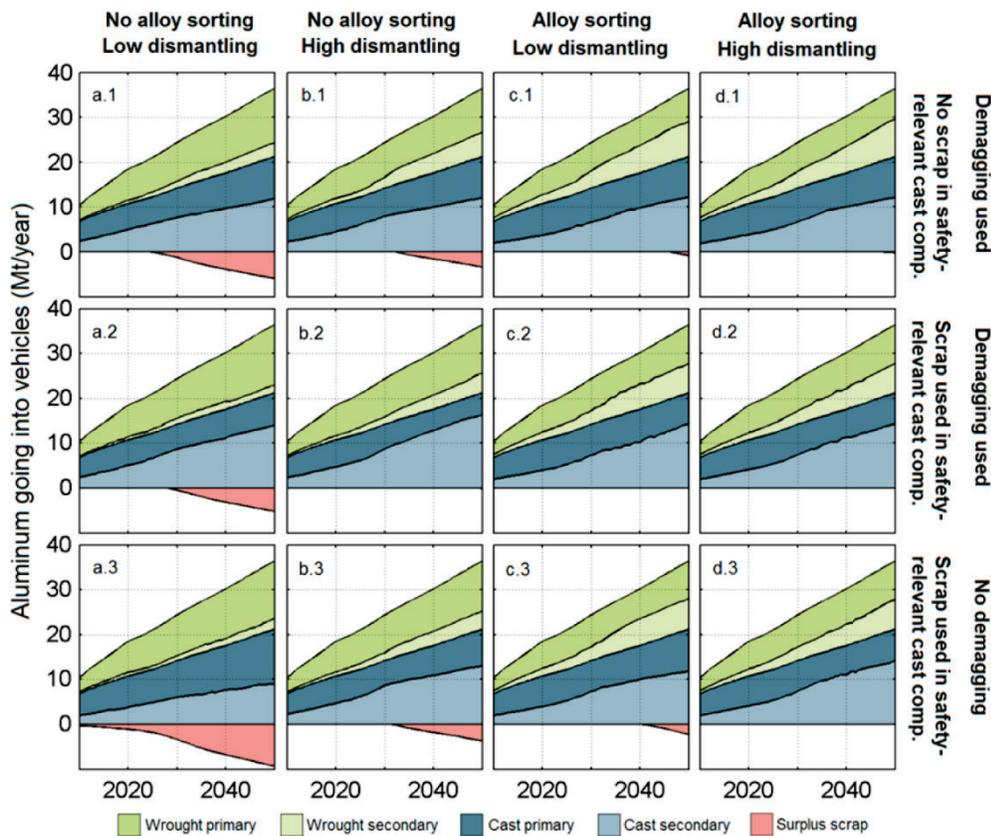


Figure 2 Simulated future production of wrought and cast aluminium for vehicles, and the relative share covered by primary and secondary sources (Løvik et al 2014)

Figure 2 shows simulation results of a dynamic multi-layer MFA model of aluminium and its alloying elements used to explore various interventions in end-of-life management and recycling of automotive aluminium (Løvik et al. 2014). It was found that increased component dismantling before vehicle shredding can be an effective, so far underestimated, intervention in the medium term, especially if combined with development of safety-relevant components such as wheels from secondary material. In the long term, automatic alloy sorting technologies are most likely required, but could at the same time reduce the need for magnesium removal in refining. A combination of better scrap segregation and recycling into safety-relevant cast components is necessary to avoid surplus scrap until 2050 (b.2, c.2, d.2–3). Increased dismantling combined with alloy sorting eliminated the need for magnesium removal during refining (d.2).

4 Conclusions

Resource quality is mainly an issue of energy use and associated greenhouse gas emissions to produce metals with required properties. Secondary resources tend to have higher quality than primary resources, which makes them attractive for recycling. However, the growing amounts of old scrap can, in certain cases, form increasing challenges for recycling.

In the case of aluminium, this necessitates restructuring of the aluminum cycle to open up new recycling paths for alloys and avoid a potential scrap surplus. Cooperation between the primary and secondary aluminum industries, the automotive industry, and end-of-life vehicle dismantlers is essential to ensure continued recycling of automotive aluminum and its alloying elements.

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1.3 Session C: Poster session

(see part 2)





1.4 Session D: Integrated Assessment of Technospheric Mining – The Role of Ecological Indicators





POTENTIALS AND LIMITATIONS OF LCA IN RESOURCE MANAGEMENT

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Introduction

Some environmental problems cannot be solved by one environmental assessment tool alone. To support decision in resource management, Material Flow Analysis (MFA) is useful to recognize resource limitations and to quantify stocks and flows (Brunner & Rechberger, 2004). However, MFA does not provide information on the environmental impacts associated with various resource management options. Comparative assertions of different products and processes can be done with Life Cycle Assessment (LCA), but typically LCA is performed related to a so-called “functional unit”, neglecting overall resource availabilities and limitations. Although there are already some links between MFA and LCA, the two methods are rarely used in a fully integrated way (Vadenbo et al. 2014a, b). In the presentation, we will show how LCA can be combined with MFA to manage resources and minimize environmental impacts. We will illustrate the capabilities and limitations of this approach using several case studies. Finally, we will give an outlook of further research needs.

Methods

The combination of MFA and LCA allows for the consideration of resource availability and restrictions, while simultaneously assessing the environmental impacts of the system. The MFA-LCA model can be further combined with scenario analysis or with optimization algorithms to derive environmentally preferable solutions and develop resource management strategies. For instance, the objective function could be the minimization of environmental impacts (assessed with LCA methodology) for the provision of industrial production and treatment of waste with a given (modifiable) infrastructure. The availability of resources is defined and resources are distributed to those uses which minimize overall impacts. For this, the various uses of (primary and secondary) resources need to be known as well as potential substitution options and restrictions. The outcome would be a set of optimal material flows (and, if dynamically implemented, also stocks), which minimizes the environmental impact while satisfying the defined needs (here industrial production and treatment of waste). Thus, the MFA is the starting point, providing key process- and system-constraints, as well as the final output of the model, in the form of optimal physical exchanges between activities while respecting resource limitations and leading to minimized environmental impacts.

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Case studies

Several simplified case studies will illustrate the application of MFA-LCA in resource management. The first case study deals with the optimal life cycle of the resource wood, i.e. addressing the question which (cascading) uses are environmentally optimal to substitute other materials with higher impacts. A particular challenge for this case is the definition of functionally equivalent substitution options. Another challenge is the longevity of many wooden products, which calls for a dynamic approach. Another challenge is the various quality requirements of different wood uses. The results of the simplified model show that cascading often (but not always) is environmentally beneficial, but only if the final use of wood as fuel is not hampered.

A second example will illustrate the application in waste management. While the final results are not yet available for this case, we will discuss the concept and also some limitations. Using the example of steel recycling, we will discuss the role of secondary resource quality and the influence on the quality of product as well as electricity demand of the recycling process and resulting environmental impact. A merit of the combined MFA-LCA model is that such aspects can be considered, which is not possible by the single application of either LCA or MFA. However, knowledge on the causal relationships between waste quality and all implications on the recycling process is still scarce, which makes it difficult to build comprehensive models.

A final case study is an existing study on regional optimization of sewage sludge in the canton of Zurich (Vadenbo et al. 2014b). This case study illustrates the (efficient) tradeoffs of various treatment strategies in terms of environmental impacts. It also illustrates some limitations of LCIA methods, which are appropriate to assess impacts such as climate change, but show large deficiencies in evaluating long-term impacts to resource depletion (Rørbech et al. 2014), e.g. of phosphorous as a resource in the case of sewage sludge management.

Conclusions and Outlook

The combination of MFA and LCA allows for the consideration of resource availability and restrictions, while simultaneously assessing the environmental impacts of the system. First examples of resource management cases exist where this combined method has been successfully applied. However, setting up such models can be complex and often there are data gaps, e.g. on the quality of resources and the influence on recyclability (and environmental impacts of the recycling process) as well as substitution options. Assuming that these data gaps can be abridged, the potential of this approach is rather large, as it allows for comprehensive assessments of resource management options, revealing tradeoffs and providing a basis to derive resource management strategies.



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QUALITY INDICATORS FOR ANTHROPOGENIC RESOURCES

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Introduction

Within the recent decade, increasing focus has been placed on recovery of resources from waste materials in society. Through initiatives such as "Roadmap to a resource efficient Europe", the European Commission has supported a transition towards increased recycling of secondary raw materials in society and increased utilization of anthropogenic resources. Traditionally, waste management in Europe – as well as recycling of resources – follows the so-called waste hierarchy indicating that reuse and recycling is preferred over other approaches for utilization of resources in waste (e.g. energy utilization). For specific waste fractions, a deviation from the waste hierarchy has to be argued based on life cycle considerations, i.e. the environmental benefits from an alternative solution should be larger than simply following the waste hierarchy.

A vast number of life cycle assessments (LCA) have been carried out focusing on recovery of resources from waste and subsequent recycling of these materials for substitution of virgin raw materials. Generally, these LCAs demonstrate that recycling indeed provides environmental savings – also larger environmental savings than the alternative management solutions for the same waste materials. Very few of these LCAs, however, take into account the quality of the recovered resources and to which extent these resources can actually substitute the intended raw materials. Two aspects are critical here: i) less than the resource potential may actually end up being recovered and recycled, and ii) the presence of contaminants in the recovered materials may cause the materials to have lower quality during recycling. Without accounting for these aspects, the LCAs may not adequately quantify the environmental benefits associated with the resource recovery.

To consistently assess resource recovery and recycling of anthropogenic resources as raw materials, systematic evaluation of the quality of the involved resources is necessary. However, in many cases information about final recycling and substitution, and thereby essentially about the quality of the recovered resources, is scarce. Evaluation of resource quality, therefore, can be challenging. Selected examples of resource quality evaluations are provided and the implications of resource quality for prioritization of resource recovery from waste is discussed.

Example A: Aluminium recovery from waste incineration bottom ashes

Metal recovery from incineration bottom ashes is well-established and in many countries an integrated part of ash management. Because of the substantial amount of energy required for primary Al production, there is economic and environmental interest in recovering aluminium. However, the Al scrap quality is critical with respect to the potential environmental benefits associated with the recycling. The quality of Al scrap is related to oxidation level, contamination and the inherent content of alloying elements in the Al scrap. In particular, contents of alloying elements in the scrap determine the potential utilisation and needs for addition of primary aluminium or high-grade aluminium alloys to obtain the desired cast alloy quality. Al scrap recovered from waste incineration scrap includes a mixture of aluminium alloys that are subsequently treated by refiners for producing cast alloy and/or applied as de-oxidation agent in the steel industry. As such, the quality of the recovered aluminium is critical for the subsequent substitution of primary aluminium.

For a range of recovery scenarios, Allegrini et al. (2015) evaluated the potential for aluminium recycling and substitution of primary aluminium based on the quality of Al scrap in waste incineration bottom ashes, see Figure 1 (Al recovery efficiencies ranged from 0 to 97 % of the potential in five consecutive scenarios).

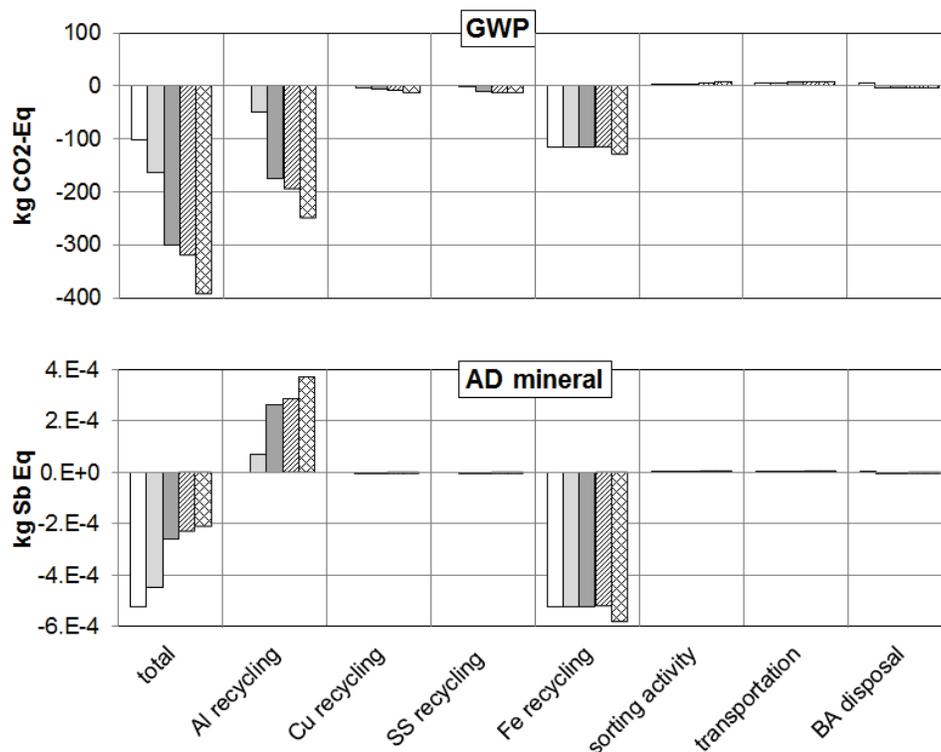


Figure 1 Life cycle assessment (LCA) results representing recovery and recycling of metals from waste incineration ashes (Allegrini et al., 2015).

While increasing recovery efficiencies and substitution resulted in increased savings within global warming potential (GWP), the same recovery caused increasing consumption of alloying elements such as Zn, which caused significant increases in abiotic resource depletion (AD mineral). As Zn has a higher characterization factor than Al in relation to abiotic resource depletion (Rørbech et al., 2014), potential Zn consumption during the recycling process outbalances the benefits of Al savings in this context. This illustrates that not only the recovered quantities of aluminium are important, but also the quality and level of impurities are critical for the subsequent recycling process in industry.

Example B: Contaminants in waste paper for recycling

Today, paper recycling is an integral part of paper and pulp production with estimated recycling rates above 70 % in Europe (CEPI, 2013). Several LCA studies have demonstrated that paper recycling offers environmental benefits. However, increasing concerns related to the presence of potential harmful chemical substances in paper have been voiced within recent years (e.g. see Pivnenko et al., 2015a), for example in relation to migration of chemicals from packaging materials into food. Bisphenol A (BPA, 4,40-isopropylidenediphenol) is a chemical produced in large volumes for a variety of applications in consumer and industrial products. Within recent years, BPA has received significant attention in relation to the quality of paper packaging materials. As an additive, BPA has been used in thermal paper (e.g. used for receipts, admission and lottery tickets) a "developer" triggering colour formation in the paper when exposed to heat, but may also be present in non-carbon copy (NCR) paper for office use.

Pivnenko et al. (2015b) analysed samples of a wide range of waste paper materials and products, both from source-segregated paper fractions as well as from paper in the residual fraction of municipal solid waste from households, see Figure 2.

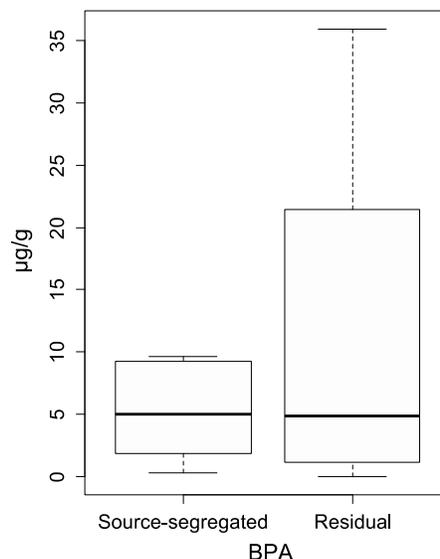


Figure 2 Boxplot of source-segregated versus residual waste paper for BPA. Outliers such as receipts at >8100 µg/g were excluded in the figure (Pivnenko et al., 2015b).

Although the mean values in Figure 2 were similar for paper found in the two fractions, significant variations between paper materials were observed. Relative high contents of BPA were observed in receipts ($> 8100 \mu\text{g/g}$) and office paper ($> 280 \mu\text{g/g}$). The latter possibly due to the presence of NCR paper in the office paper fraction. This illustrates that specific data for the quality and composition of recyclable material fractions, such as paper, is needed in order to evaluate the potential recyclability. The above results suggest that some paper fractions (with receipts as an extreme example) do not have the same quality as the virgin fibres they may be intended to displace. As such, not only the technical properties of recovered materials (including presence of material impurities) are critical for recycling, but also the presence of chemical contaminants embedded in the products themselves.

Resource quality: Implications and priority of resource recovery

As indicated by the above examples, the composition and properties of recyclable materials are critical with respect to the subsequent substitution of virgin resources. Currently, very limited information is available about the detailed quality of resources in waste. With pressure on increasing the recycling rates in society, a concern may be that this may also result in increasing levels of contaminants within the material cycles. As indicated for aluminium, resource depletion indicators may be limited for representation of resource quality, but at the same time contribute to an illustration of consumption of auxiliary materials during recycling. Prioritization resource recovery from waste has to be based on detailed understanding of i) composition and material properties of the recovered resources, ii) downstream use of these resources as secondary raw materials in industry and potential consequences for consumption of auxiliary materials, iii) resulting substitution of virgin resources, and potentials for spreading of un-wanted chemicals in society. As such, simple life cycle assessment of resource recovery scenarios solely focusing on global warming aspects are not sufficient.

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RESOURCES AND RAW MATERIALS: MEASUREMENT OF THE EFFICIENT USE AND THE BENEFITS OF CLOSING THE LOOPS

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Introduction

In this contribution, we discuss the many different approaches of resource efficiency and propose a framework to structure the many different approaches. From a technical point of view, we discuss the potential of exergy as a base to quantify resource efficiency, both in abiotic and biotic production chains.

Further on, there are quite different interpretations of the terminologies ‘natural resources’ and primary and secondary ‘raw materials’. We bring proposals for coherent definitions of ‘natural resources’ and ‘raw materials’. We introduce a systematic classification for both natural resources and raw materials.

Finally, we present how the benefits in terms of natural resource savings of closed loop and open loop recycling can be quantified, illustrated with some case studies.

What can be understood by Resource Efficiency?

Resource efficiency indicators have been developed for systems situated at different levels of economic activity: from the micro-scale of specific processes and products to the meso- and macro-scale of sectors and countries. Some indicators evaluate resource efficiency in a national or regional perspective, while others consider a more global perspective by including resources that are embodied in imported products.

It is essential to make a distinction among different resource efficiency indicators. We propose a framework that provides five types of resource efficiency indicators, organized in a two level approach (Huysman et al., 2015b). Level 1 indicators define an efficiency that originates from

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engineering: the ratio between the useful outputs (or benefits) and the inventoried flows. The inventoried flows can be both inputs (resources) and outputs (emissions):

(1) In case the inventoried flows are inputs, we obtain 'resource efficiency at flow level'.

(2) In case the inventoried flows are emissions, we obtain 'emission efficiency at flow level'.

Level 2 is linked to the eco-efficiency concept and defines efficiency as the ratio between the intended effects (or benefits) and environmental impacts, assessed through specific impact assessment models. There are three possibilities:

(3) When the environmental impact in the denominator is derived from resource flows, the resulting efficiency is called 'resource efficiency at impact level'.

(4) When the environmental impact in the denominator is derived from emission flows, the resulting efficiency is called 'emission efficiency at impact level'.

(5) When the denominator represents an overall environmental impact, derived from both resource flows and emission flows, the resulting efficiency is called 'overall efficiency at impact level'.

Exergy as a base to quantify Resource Efficiency

From an engineering perspective, exergy has been used since decades to optimize processes technically, especially in the processing of abiotic raw materials (fossil fuels, minerals and metals). More recently, exergy has been used in a broader sense, e.g. along life cycles and as indicator in Life Cycle Analysis (Dewulf et al., 2008). In this contribution, we show some of the recent developments of the use of exergy in resource efficiency measurement, in particular in biobased production systems (Huysveld et al., 2015).

Natural Resources versus Raw Materials

Different stages can be identified along the life cycle of a product, starting from the asset of natural resources in the environment, through to the production of products and services bringing functionality to fulfill human needs, followed by end-of-life waste management (Dewulf et al., 2015a,b). We differentiate the following assets of natural resources: land area; sea area; flow energy resources (solar irradiation, water, wind and tidal currents); water; metallic ores; minerals (for industrial and construction applications: so called industrial minerals and construction materials); fossil fuels; nuclear ores; atmosphere/air; and natural biomass (natural flora and fauna).

In the primary production sector, natural resources at the cradle are transformed into base products, typically the first market commodities. We have named these (primary) (non-energy) raw materials or primary energy carriers (or (primary) energy raw materials), depending on their further applications. Raw materials will have in the end mainly material functions (e.g. refined metal). Primary energy carriers (e.g. natural gas) are mainly used as a utility for heating, cooling, pressurizing, transportation etc. Primary energy carriers are basically the first marketable

element in energy supply chains. They are the first traded energy form as the primary production sector typically transforms the natural resources into a form that can be supplied to further use elsewhere after trade. We propose a coherent structure of 7 raw materials groups and 5 types of primary energy carriers.

Quantification of benefits of open and closed loop recycling

In a last section, we analyse how open and closed loop recycling can be beneficial in saving of natural resources, relying on the ReaPro methodology (Ardente and Mathieux, 2014) and the exergy-based quantification of resource use (Dewulf et al., 2007). We illustrate the work with some specific cases of waste flows from the Belgian take-back scheme (Huysman et al., 2015a).

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1.5 Session E: Drivers and Barriers for Resource-Efficient Nutrient Management – The Case of Phosphors



TOWARDS SUSTAINABLE PHOSPHORUS MANAGEMENT IN EUROPE



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Summary

Phosphorus (P) is a non-renewable resource, irreplaceable for food production. L surface waters exceed Planetary Boundaries by $8x^{(i)}$ and P is the main reason for failure to achieve European water quality objectives. This offers major opportunities for improving management, recycling, the bio-nutrients circular economy and rural/farm job creation. The EC Communication on Sustainable Phosphorus 2013 identified stakeholder concerns and possible actions. Phosphate rock was added to the EU List of Critical Raw Materials in 2014. Bio-nutrients were identified as a priority in the European Commission public consultation on Circular Economy policy in 2015. The EU FP7 P-REX project published proposed policy orientations in 2015. The European Sustainable Phosphorus Platform (ESPP) brings together stakeholders (industry, users and market, regulators, innovation, NGOs) in an active network to address phosphorus management, use efficiency, recycling (reuse, recovery). ESPP ensures stakeholder dialogue and input to decision makers, e.g. on R&D and innovation policies, data on nutrients to support stewardship (DONUTSS), EU Fertiliser Regulation recast and criteria for struvite and biomass ashes, REACH, BAT documents, BEMPs (EMAS), standards (CEN, ISO ...). ESPP maintains an inventory of P-recycling R&D and facilitates innovation dissemination, experience transfer, value chain communication to market.

Phosphorus use in Europe

Information on phosphorus flows in Europe (van Dijk et al. 2015)⁽ⁱⁱ⁾ shows that (in 2005) the EU-27 imported nearly 2.4 million tonnes of phosphorus (P, i.e. 2 400 Gg P), of which somewhat less than half accumulated in agricultural soils and just over half was lost as waste. Phosphorus use efficiency (PUE) was around 70% in crops but retention in animal products was only 24%, so that in total 28% of net P input finally reached supplied human food products. Around 4% of net phosphorus input was lost to surface waters, potentially contributing to eutrophication when reaching lakes, rivers or estuaries.

The authors conclude that there are wide variations between different European countries and regions, but that generally phosphorus use in EU-27 was characterized by relatively (1) large dependency on (primary) imports, (2) long-term accumulation in agricultural soils, especially in west European countries, (3) large losses throughout entire society, especially emissions to the environment and sequestered waste, (4) little recycling with the exception of manure, and (5) low use efficiencies, because of aforementioned issues, providing ample opportunities for improving sustainability of phosphorus management.

EU Critical Raw Material

In 2014, the European Commission added phosphate rock to its list of (now 20) Critical Raw Materials⁽ⁱⁱⁱ⁾. These materials are defined by high economic importance and a high risk for supply security. Phosphate rock is considered by the EU Commission report to be subject to “high supply risk”; the principal use of phosphate rock is in fertilisers and global demand is expected to increase (because of growing world population and so need for food), and there are no alternatives to phosphorus in fertiliser and animal feeds to produce food. European policies to address raw materials criticality include improving efficiency of materials use and recycling, waste policy and international cooperation to address supply security.

An update of the EU’s assessment of phosphate rock criticality is currently underway. The stakeholder meeting underlined issues with the methodology used (MSA = Raw Materials System Analysis) and the lack of adequate data on P flows and stocks (see below). ESPP input underlines that the assessment should consider not only “phosphate rock”, but also phosphorus in all forms (P) in order to **address bio-nutrients and recycling**, and also specific forms of phosphorus, in particular **White Phosphorus (P4)** which is critical for strategic industries (chemicals, fire safety, electronics) and is no longer produced in Europe (100% technological import dependency).

European Commission Communication on Sustainable Use of Phosphorus

In 2014, the European Commission published conclusions (SWD(2014)263) to the Consultative Communication on the Sustainable Use of Phosphorus (COM(2013)517, see ESPP’s SCOPE Newsletter n° 95) underlining that “closing the phosphorus cycle is both possible and desirable” and identifying the following needs for action:

- Increasing knowledge of P supply (resources) and demand (flows)
- Security of supply
- Risk assessment of contaminants in mineral or in recycled fertilisers as used
- Recovery of P from wastewaters, food waste and other biodegradable wastes
- Improving P use in agriculture
- Innovation and research in P use, mining & processing, environmental impacts of P, agronomic P-efficiency, P-recovery and recycling
- Policy instruments and integration of P into existing legislation
- Awareness raising, including through nutrient Platforms

Need for better information on bio-nutrients

Adequate, up-to-date, useable **data on flows of N, P, K, Mg and other nutrients is essential to define stewardship objectives, targets and actions**, and to monitor their effectiveness, for regulators, industry and users (market). An interesting initiative in this area is the national Observatory of Mineral and Organic Fertilisation established by the French fertiliser industry (ANPEA, Unifa, CAS, AFCOME)^(iv). ESPP is launching work with stakeholders to define how to improve nutrient data for decision making and recycling industry development (DONUTSS)^(v).

Van Dijk et al. (above) have estimated the **potential for phosphorus recovery and recycling in Europe**, concluding that phosphorus recycling from waste streams other than manure could represent 274-396 ktP/year, i.e. around one quarter of current EU mineral phosphorus fertiliser consumption (1 448 ktP/year). Additionally, large amounts of P in manures are already recycled, including through on-farm use (1 736 ktP/year).

There is also a clear **need for better information concerning recycled nutrient products**, many of which are new or are evolving: agronomic value, use and application, quality. With support of 60+ organisations, ESPP submitted to EIP-AGRI (European Innovation Partnership for Agriculture, Horizon 2020 / DG AGRI) a “Focus Group” proposal on agronomic use of recycled nutrients, covering fertiliser value; nutrient loss mitigation; contaminants; impacts on soil, BMPs for handling and application; circular economy and farm added-value^(vi).

Policy recommendations

The **EC published conclusions of the phosphorus recycling workshop** (organised parallel to the 2nd European Sustainable Phosphorus Conference, 2015)^(vii) noted that some P-recycling processes are already at the commercial production scale and underlined the need for policy support for phosphorus recycling. The workshop identified R&D needs including: flow studies, social science, value-chain actions, risk assessment of organic contaminants, full-scale demonstration projects, R&D to support standards and BAT/BEMP, regional approaches niche markets, clustering of projects and networking.

The **EU-funded FP7 project P-REX “Policy Brief”** 2015^(viii), coordinated by Christian Kabbe, recommends the following to enable widespread implementation of P recycling from wastewater

- EU phosphorus recycling target and roadmap for implementing this target
- Regional phosphorus policies
- Adaptation of existing legislation to recycled nutrients, level playing field to mineral-based fertilisers, and implementation guideline
- Mechanisms to share costs of phosphorus recycling: e.g. quotas or recovery obligations
- Funding of full-scale demonstration installations

Platforms in action

The **European Sustainable Phosphorus Platform (ESPP)** launched 2013, addresses P recycling (recovery, reuse), P supply, P management and use efficiency in a sustainable and safe food chain, from farm to diet. National nutrient platforms are operational in Netherlands, Flanders, Germany, the Baltic (BSAG), with projects in UK, Norway, Czech Republic. ESPP also cooperates with the North America (NAPPS) and Japanese platforms, and with the Global Partnership on Nutrient Management (GPNM). The platforms bring together industry, regulators, technology suppliers, R&D, farmers organisations, NGOs and enable:

- **Networking** of expertise, projects, success stories, value chains
- **Stakeholder dialogue with decision makers** and input to policy and regulation
- **Innovation, knowledge and R&D dissemination**, from production through to market
- Development of a **vision for P sustainability** in Europe, communications, awareness raising

The platforms strongly promote **innovation** and its implementation in nutrient sustainability

- inventory of P-recycling technology assessments (WETSUS)^(ix)
- inventory of current and recent P-recycling R&D projects^(x)
- R&D needs (see EC P-recycling workshop, above)
- update of farm Best Nutrient Management Practice (BMP) fact sheets^(xi)

ESPP and its partners are directly involved in a range of **current EU regulatory and policy processes**:

- EU Fertiliser Regulation recast: taking recovered nutrient products into account and EU criteria development (JRC) for struvite, biomass ashes
- Organic Farming Regulation: proposed validation of recycled P products
- REACH (EU chemicals regulation): Art 2(7)d “recovered” substances, by-products
- BAT BREFs (Industrial Emissions Directive): waste incineration, pig & poultry production
- BEMPs: EMAS (EU Eco-Management and Audit Scheme Regulation) “agriculture”
- EIP-AGRI Focus Group proposal: agronomic use of recovered nutrient products
- EU Critical Raw Materials list: MSA update
- standards: CEN SABE, ISO 275 ...

A key current development is the **proposed EU Circular Economy policy**^(xii). The Ellen MacArthur Foundation^(xiii) estimates that a circular economy system for food production could cut Europe’s food cost per person by 30%. Bio-nutrients were identified as a priority sector for EU Circular Economy policy by a quarter of respondents to the EU’s 2015 public consultation. ESPP is actively involved in this process, fostering stakeholder input to policy development.

- (i) “Taking planetary nutrient boundaries seriously: Can we feed the people?”, *Global Food Security* 3 (2014) 16-21 <http://www.sciencedirect.com/science/article/pii/S2211912413000540> (Open Access)
- (ii) Van Dijk, K. C., Lesschen, Jan Peter and O. Oenema. "Phosphorus flows and balances of the European Union Member States." *Science of the Total Environment*, accepted 2015.
- (iii) See ESPP’s SCOPE Newsletter n° 104 on www.phosphorusplatform.eu
- (iv) see ESPP’s SCOPE Newsletter n° 109
<http://www.afcome.org/wpcontent/uploads/2014/10/Création-Observatoire-national-de-laferilisation-minérale-et-organique.pdf>
- (v) DONUTSS workshop (Data on Nutrients to Support Stewardship), Ghent, 3-4 September 2015, with support of the EU Commission (DG GROW) and BioRefine
www.phosphorusplatform.eu
- (vi) See ESPP’s SCOPE Newsletter n° 114 on www.phosphorusplatform.eu
- (vii) See ESPP’s SCOPE Newsletter n° 115 and 111: “Circular approaches to phosphorus: from research to deployment”, 48 pages, European Commission Directorate-General for Research and Innovation 2015, ISBN 978-92-79-46827-8 <http://bookshop.europa.eu/en/circular-approaches-to-phosphorus-pbKI0115204/> conclusions of the workshop on phosphorus recycling, 4th march 2015, Berlin, organised by the European Commission, P-REX and ESPP parallel to ESPP2 (2nd European Sustainable Phosphorus Conference)
- (viii) P-REX policy brief, March 2015: see ESPP’s SCOPE Newsletter n° 111 and 115
http://prex.eu/uploads/media/P-REX_Policy_Brief_final.pdf
- (ix) WETSUS « Inventory and summary of technology assessments of phosphorus recycling technologies » regularly updated, see ESPP’s SCOPE Newsletter n° 109 and 105, www.phosphorusplatform.eu under ‘Downloads
- (x) see ESPP’s SCOPE Newsletter n° 111 www.phosphorusplatform.eu
- (xi) see ESPP’s SCOPE Newsletter n° 115 www.phosphorusplatform.eu
- (xii) European Commission public consultation on the Circular Economy (closed 2th August 2015) http://ec.europa.eu/environment/consultations/closing_the_loop_en.htm based on proposed ‘Roadmap’ (April 2015)
- (xiii) “Growth Within: a circular economy vision for a competitive Europe”, 100 pages, 25th June 2015, Ellen MacArthur Foundation, McKinsey Center for Business and Environment, SUN (Stiftungsfonds für Umweltökonomie und Nachhaltigkeit)
<http://www.ellenmacarthurfoundation.org/news/latest-research-reveals-more-growth-jobs-and-competitiveness-with-a-circular-economy>



HISTORICAL ANALYSIS OF PHOSPHORUS FLOWS IN AUSTRIA

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The construction of multi-year MFAs

Based on the analysis of the Austrian phosphorus budget by Egle et al. (2014) this analysis was repeated for the years 1990 to 2013 forming a time series for this system (Zoboli et al. 2015). The analysis was carried out applying material flow analysis according to Brunner and Rechberger (2004) and using the MFA software STAN, a product by Technische Universität Wien (free download under www.stan2web.net). The application of STAN allows for the consideration of data reconciliation and error propagation (Fellner et al. 2011). For this purpose data uncertainty was determined by the approach of Laner et al. (2015). The goal of this analysis was to find out how the system changed over time (small or large changes; steadily or abruptly) and how such a multi-year time series enhances the system understanding compared to classical MFA, where usually a single year budget is analysed and interpreted.

Methodological approach to analyse the time series

The Austrian phosphorus budget consists of 56 processes, 8 stocks, and 122 flows. As almost all flows are determined by data sources the system is overdetermined and data reconciliation is performed. The extent of the reconciliation step indicates if the database is plausible. In a next step all flows were analysed with respect to changes over the years. This was done in two ways: First, each single flow of the budgets for 1991 to 2013 was compared to the initial year 1990. Second, flows of neighbouring years were compared. The first comparisons gives some indication about the overall evolution of the system while the second one provides information if changes appeared more steadily or even abruptly. For the flow comparisons different tolerance levels were applied ($\pm 0\%$, $\pm 5\%$, $\pm 10\%$, $\pm 15\%$, $\pm 20\%$, $\pm \sigma$, $\pm 2\sigma$). A flow is considered constant if the ranges of two years, given by their mean values and tolerance levels, do not overlap. An extreme change is given when the flow more than doubled or halved, all other changes in between are considered as moderate, see Figure 1.

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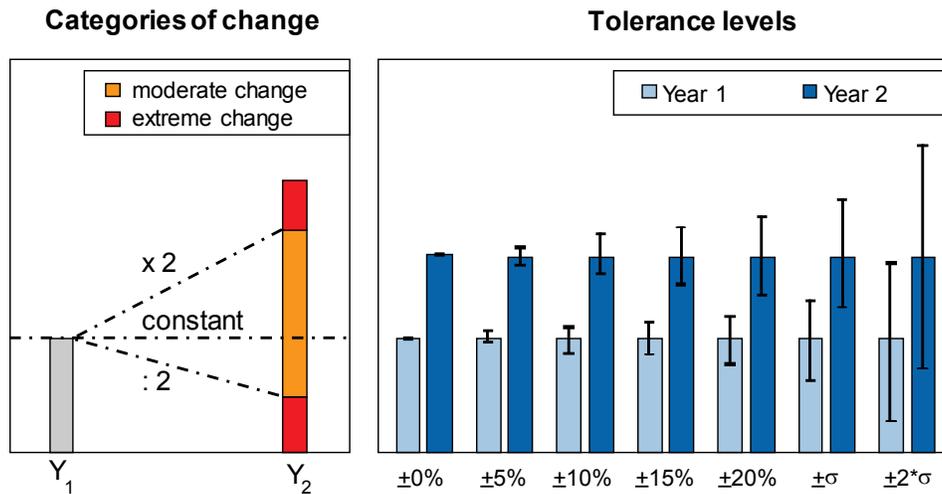


Figure 1 Classification of flow changes (Zoboli et al. 2015)

Results

Figure 2 left side shows that more than 50% of all flows (122) and net stock changes (8) experienced extreme changes since 1990. This is in so far considerable as results of MFAs are often taken as actual and as a basis for decision-making for times spans up to 10-15 years. At least for the example of phosphorus and Austria this would not be justifiable. The right side shows that even some 30% of all flows change extremely within one year, therefore steadiness of the system is rather low, which has to be kept in mind if extrapolations into the future are performed. Analysing the total system by focusing on the development of single flows is a straight forward approach to detect areas of required political action. For example, Figure 3 displays the trends for phosphorus flows into the waste management subsystem (left) and the recovered phosphorus amounts (right).

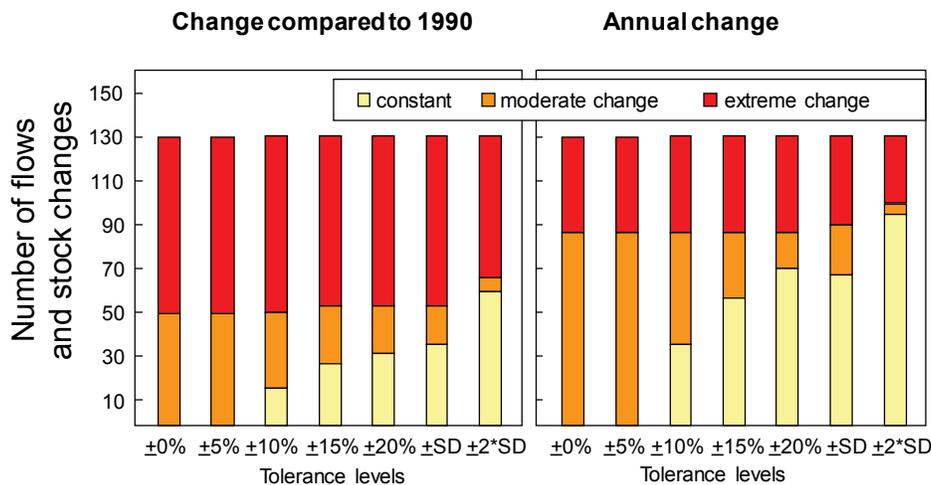


Figure 2 Result of the classification process for flow changes (Zoboli et al. 2015)

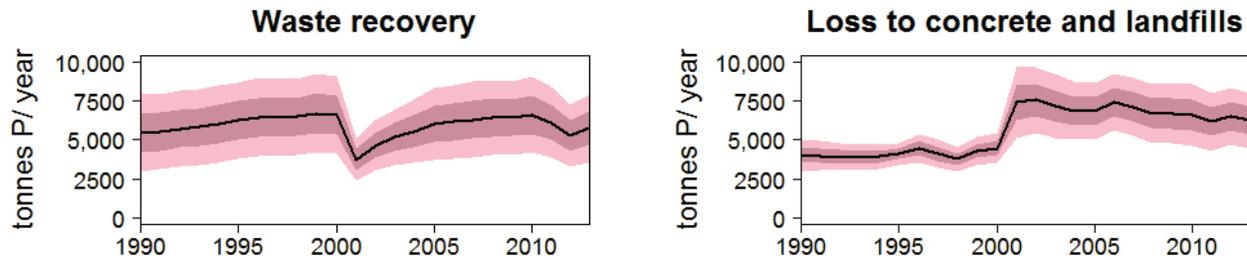


Figure 3 Time series 1990 to 2013 for the input of phosphorus to the waste management system (left) and the recovered phosphorus amounts (right).

It becomes clear that the “responsibility” of waste management has increased since 1990 but the recovered amount stayed rather constant resulting in an overall decline of the recycling rate: a clear indication for the need to take counter measures. Figure 4 shows the development of the three relevant net stock additions in the system. It can be seen that the extent of over-fertilization with phosphorus could be significantly reduced over the years. The high prices for phosphate rock in 2008/09 even made a balanced phosphorus soil budget possible. The largest loss in today’s (2013) system occurs to landfills (disposal of sewage sludge ashes) and concrete (combustion of sewage sludge and meat and bone meal in cement kilns). Again a clear indication for required policy action. Another loss happens in the process private households, which comprises also private and public gardens. There, a rather constant amount of 2,200 t/yr are “potted”. The number is a result of the input/output balance of this process.

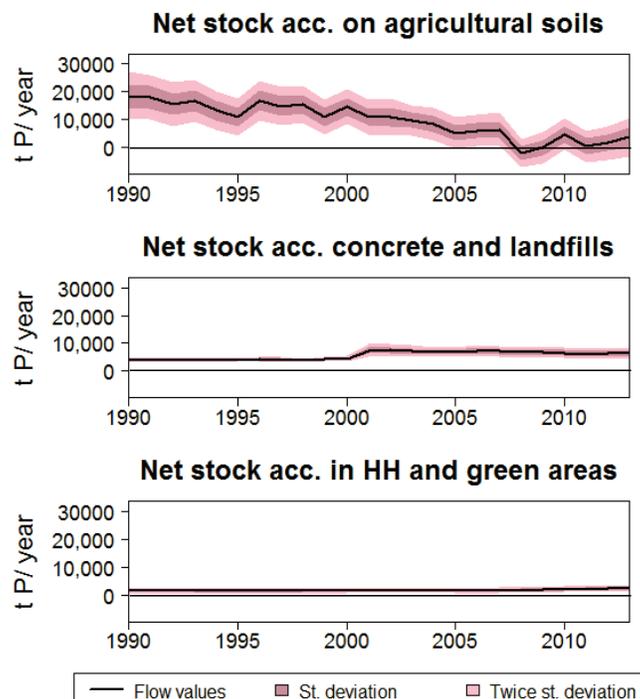


Figure 4 Time series 1990 to 2013 of the three major stock changes in the Austrian phosphorus budget.

To date it is not quite clear where this loss stems from. However, the miss-balance appears fairly constant through all the years making data imperfections rather implausible. This is a typical case where the time series approach points to further and deeper investigation.

Conclusions and outlook

MFA time series increase the system understanding considerably as system dynamics become visible. The results are beneficial with regard to

- Detection of points where the system could be optimized (support in decision-making)
- Monitoring of flows and control of success of policy actions
- Control of data and detection of data imperfections (conflicting data)
- Optimization of data generation with regard to quantity and quality

In our next steps, we want to identify measures of optimization and construct a hypothetical optimized system. Then we want to apply statistical entropy analysis to the time series and test it for its usefulness as a quality indicator. If this is successful, the identified measures will then be monetarised and ranked with respect to efficiency.

Phosphorus here also serves as a role model as we think that national MFA time series should be produced on a routinely basis by national authorities, e.g. the EPAs. The required tools are available now to establish sound time series, which are a necessary requirement for efficient resource management and the establishment of a high-level operating circular economy.

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PHOSPHORUS FLOWS IN GERMAN SEWAGE SLUDGE ASHES AND POTENTIAL RECOVERY TECHNOLOGIES

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Introduction

Phosphorus (P) is essential for all forms of life and cannot be substituted. It is one of the macro nutrients applied in form of mineral or organic fertilizers in agricultural crop production. Mineral P fertilizers are generally based on phosphate rock, a limited resource. Western Europe completely depends on imports as it has no own relevant phosphate mines. The most abundant phosphate deposits are located only in a few countries such as Morocco, China, South Africa and the United States of America. Furthermore, phosphate rock and as a consequence mineral P fertilizers are often contaminated with the toxic elements cadmium and uranium. In order to substitute phosphate rock P-bearing waste streams are investigated for their potential to be recycled as fertilizers. Wastewater is one of the important P-bearing waste streams that were in the focus of research in the last 10 years. German wastewater is e.g. a carrier of approx. 70,000 t of phosphorus per year. This is a relevant potential for partly substituting or complementing the annual German P fertilizer consumption of approx. 140,000 t phosphorus. Several technologies were developed to extract P from waste water, concentrated side streams, sewage sludge or sewage sludge ashes. Techniques to recover P from the aqueous phase of waste water treatment plants (WWTP) with enhanced biological P-removal (EBPR) in a concentrated side stream are already well established on industrial scale at large WWTP in several countries. These technologies are successfully marketed by different companies and can have benefits for the whole WWTP-system in terms of improved sludge dewatering, lower P-backload, higher total P-elimination and lower maintenance in the periphery of the anaerobic digesters (pipes, pumps, reactors). These systems produce struvite ($\text{MgNH}_4\text{PO}_4 \cdot 6 \text{H}_2\text{O}$) and are even economic without selling the struvite. However, the existing industrial installations recover approx. 10% of the total P-stream of a WWTP and are only applicable in combination with an EBPR-plant (only 10% of German WWTP). High P-recovery can be achieved by processing dewatered sewage sludge or sewage sludge ash (theoretical potential ~90%). Extracting P from sludge comes along with high consumption of chemicals such as sulphuric acid and citric acid. A demonstration plant is operating since 2007 at the WWTP Gifhorn with a recovery rate of approx. 50% of the total P-load. The highest P-mass fractions can be found in the ashes after mono-incineration of the sludge with up to 13 w-% P. Principally, technologies for P recovery from sewage sludge ash (SSA) are based on two different approaches: wet-chemical and thermochemical treatment. Some of these technologies are currently considered for large-scale applications. More than 90% of phosphorus can be leached from SSA by mineral acids at pH values <2 (Petzet et al.,

2012). The level of P leaching strongly depends on the ratio of acid to ash and on the chemical composition of the ash (e.g. Ca, Al and Fe mass fractions). Petzet et al. (2012) summarized the data from literature and found that 0.3-0.68 kg acid / kg SSA are required to dissolve 66.5-99.4 % P. Phosphorus is subsequently precipitated in form of calcium phosphate or upgraded to phosphoric acid. Thermochemical approaches focus on transformation of phosphates into plant available forms and separation of undesired heavy metals. A thermochemical approach jointly developed and patented by BAM and Outotec (Adam et al., 2014) is described more in detail below.

Monitoring of German Sewage Sludge Ashes

In order to determine the P-recovery potential of German sewage sludge ashes a complete survey was undertaken by BAM in 2011-2012. Aim of the project was to analyze all SSA that arise in Germany regarding their chemical composition. The results of the chemical composition are given in Table 1 (main elements) and Table 2 (trace elements).

Table 1 Main element contents of all SSA samples [mass-%]

Element	Min	Max	Mean value	Median	Number of samples
Al	0.7	20.2	5.2	4.8	252
Ca	6.1	37.8	13.8	10.5	252
Fe	1.8	20.3	9.9	9.5	252
K	<0.006	1.7	0.9	0.9	227
Mg	0.3	3.9	1.4	1.3	252
Na	0.2	2.6	0.7	0.6	252
P	1.5	13.1	7.3	7.9	252
S	0.3	6.9	1.5	1.0	252
Si	2.4	23.7	12.1	12.1	252
Ti	0.1	1.5	0.4	0.4	252

26 mono-incineration facilities are operated in Germany, annually producing approx. 300,000 t of SSA containing 19,000 t P. This amount is approx.12% of the amount of P applied in form of mineral fertilizers in German agriculture and represents a relevant potential for substitution of conventional fertilizers based on phosphate rock. At present, 29% of the SSA are landfilled, 37% filled in underground mines, 29% used for construction purposes and only 5% are used as fertilizer. Comparing

the results of the chemical composition of the German sewage sludge ashes with the limit values of the German fertilizer ordinance it turns out that 12,000 t of the 19,000 t of phosphorus cannot be used as fertilizer directly due to heavy metal contents exceeding the limit values. Furthermore, the bioavailability of P in the ashes is poor (P-solubility in neutral ammonium citrate: 25.6% [median of 35 samples]) and not sufficient for fertilizer applications. The results of the survey can be found in (Krüger et al., 2014; Krüger and Adam 2015).

Thermochemical treatment of sewage sludge ash

A thermochemical treatment for SSA was developed following the two main goals: i) increasing the P bioavailability and ii) removal of toxic trace elements. SSA is treated with sodium compounds under reducing conditions at approx. 900-1000°C in a rotary kiln (Stemann et al., 2015; Herzel et al., 2015). The mineral P phases present in SSA are transformed to CaNaPO_4

(buchwaldite) which is known as a well bioavailable P species. The P bioavailability can be evaluated by determination of the P solubility in neutral ammonium citrate solution (P_{NAC}). Thermochemical experiments were carried out in crucibles in order to determine the efficiency of different alkali compounds that could be alternatively used for this process.

Table 2 Summary of trace element content of all SSA samples [mg/kg]

Element	Min	Max	Ø	Median	samples	Element	Min	Max	Ø	Median	samples
Sc	0.7	48.5	6.1	4.2	252	La	9.8	269	32.0	25.5	252
V	9.3	1206	136	54.0	252	Ce	15.7	169	49.2	42.8	252
Cr	58	1502	267	159.7	252	Pr	1.37	18.4	4.9	4.2	252
Mn	334	6488	1914	1307	252	Nd	4.71	60.9	17.2	15.6	252
Co	7.3	83.5	28.1	20.7	164	Sm	0.9	20.8	3.3	2.9	252
Ni	8.2	501	105.8	74.8	251	Eu	0.18	2.1	0.7	0.6	252
Cu	162	3467	916	785	252	Gd	0.8	59.1	4.6	2.8	252
Zn	552	5515	2535	2534	252	Tb	0.1	1.8	0.5	0.4	252
Ga	3.2	24.7	11.8	11.6	252	Dy	0.5	11.1	2.4	1.9	252
Ge	2.4	12.8	5.9	5.5	252	Ho	0.1	1.9	0.5	0.4	252
As	4.2	124.0	17.5	13.6	252	Er	0.3	6.1	1.3	1.0	252
Se	0.8	9.9	2.5	2.0	102	Tm	0.0	0.7	0.2	0.2	252
Sr	67	2340	578	493	252	Yb	0.3	5.7	1.3	1.0	252
Y	1.1	53.0	11.6	9.2	252	Lu	0.0	0.9	0.2	0.2	252
Zr	23.9	984	167	106	250	Hf	0.8	25.2	4.4	3.2	252
Nb	0.6	293	19.5	11.0	252	Ta	<0.15	98.7	4.3	1.2	252
Mo	7.5	112	25.3	20.0	252	W	<6.0	336	52.8	41.1	252
Ag	0.5	133.1	12.5	9.1	248	Au	<0.15	8.4	1.2	0.9	252
Cd	<0.1	80.3	3.3	2.7	252	Hg	0.1	3.6	0.8	0.5	143
Sn	7.3	8706	194	76.6	252	Pb	<3.5	1112	151	117	252
Sb	1.5	454	23.0	12.4	252	Th	1.0	19.3	5.2	4.9	252
Ba	128	13650	2173	1057	229	U	1.58	25.5	5.8	4.9	252

The P_{NAC} -solubility is displayed over the ratio alkali/P of the input material mixture in Figure 1. Generally, the P_{NAC} -solubility of SSA (35 %) was significantly increased for alkali/P ratios > 1. A P_{NAC} -solubility ≥ 90 % was achieved by treatment with alkali/P ratios ≥ 2 with the Na additives Na_2SO_4 , Na_2CO_3 and NaOH. Generally, higher amounts of potassium (K_2CO_3 and KOH) were required to achieve comparable results. This was due to partly evaporation of potassium during the experiment resulting in lower alkali/P-ratios in the product after thermochemical treatment. The lower P_{NAC} -solubility for K_2SO_4 (maximum 70 %) is linked to the high melting point of K_2SO_4 (1070 °C) which is above the temperature of the thermochemical treatment that requires melt phases for the desired reactions.

The described thermochemical process with Na_2SO_4 was successfully tested in a demonstration trial with a total output of 2 t recycling fertilizer. The product had a P content of 7.7 % and a high P_{NAC} -solubility of ~82 %. Additionally, the mass fractions of the elements As, Cd, Hg and Pb were significantly decreased by evaporation under the reducing conditions in the kiln. The removal rate calculated on the mass balance was 61 % for As, 80 % for Cd, 68 % for Hg, 39 % for lead and 9 % for Zn (Herzel et al., 2015).

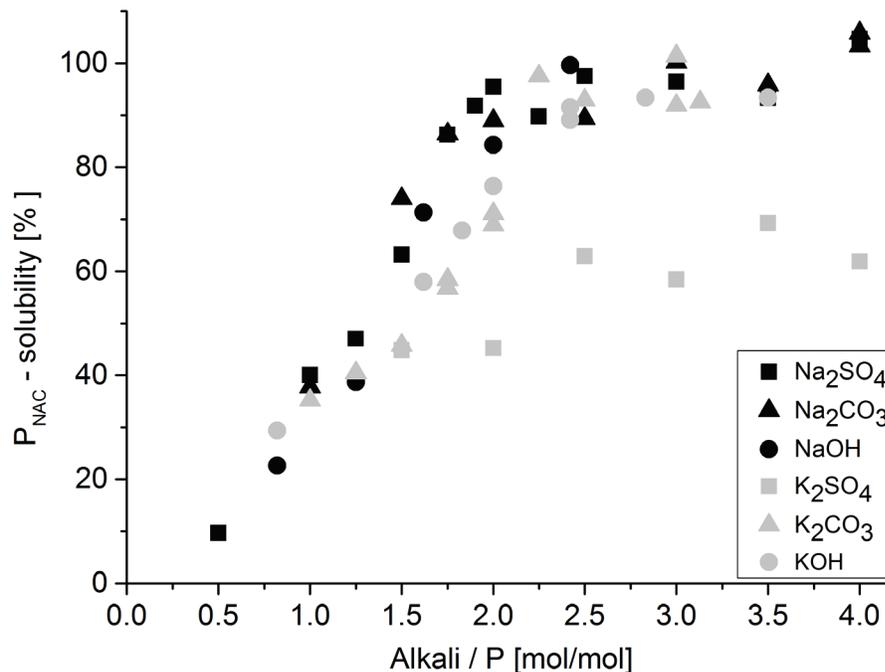


Figure 1: P_{NAC} -solubility of products from SSA calcined at 1000°C with Na_2SO_4 , Na_2CO_3 , $NaOH$, K_2SO_4 , K_2CO_3 and KOH as a function of Na/P or K/P ratio [mol/mol]. Na/P or K/P ratio corresponds to the molar ratio in the starting material

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1.6 Session F: Drivers and Barriers for Landfill Mining





LANDFILL MINING: ON THE POTENTIAL AND MULTIFACETED CHALLENGES OF IMPLEMENTATION

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Introduction

In recent years, a revived interest in landfill mining has occurred influenced by industrial ecology perspectives aiming for resource-effective and cyclical uses of materials and energy resources. Several studies point at a significant societal potential in terms recovery of large amounts of previously abandoned strategic materials (e.g. metals), reduced life cycle environmental impacts, reclamation of urban land, improved regional material autonomy, pollution prevention related to the vast number of (old, poorly equipped) landfills in most regions and strengthening of local economies through employment and spill-overs in Cleantech, Recycling and other sectors.

But several studies also conclude that realizing this potential involves many challenges in terms of knowledge production, technology innovation and adjusted policy and market conditions. In essence, landfill mining implementation is restricted due to limited knowledge and experience on how such resource extraction can be executed cost-efficiently and with clear societal benefits. Recent assessments of (often hypothetical) landfill mining cases also display somewhat different results, indicating that there seem to be many factors that may or may not turn out to be significant for the outcome depending on where, how and for what objectives projects are to be realized. In this presentation, I will go through the main findings of these economic and environmental systems analyses in order to specify key challenges for landfill mining implementation.

What do recent systems analyses say about economic and environmental performance?

For municipal landfills, the main topic of this emerging knowledge area, most of the reviewed studies support a conclusion from the past that resource extraction alone can seldom motivate landfill mining financially. Despite recent decades of increasing commodity prices and progress in technology, the costs for excavation, material processing and site-restoration are often still too high in relation to anticipated revenues for the extracted resources. In fact, even (theoretical) landfill mining cases involving non-commercial technologies and assuming high recovery efficiencies seem to experience a hard time financially when applied on such “low-grade” deposits. A fundamental problem is that in most markets only a small share of the exhumed

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resources will generate any significant income (e.g. metals) for the landfill mining practitioner while the remains typically will involve low material revenues (e.g. aggregates) or, even worse, disposal costs (e.g. gate fees for waste fuel to incinerators). Several studies therefore stress the importance of further upgrading to more high-value products and business models internalising revenues from, e.g., energy recovery of exhumed combustibles (i.e. heat and electricity) into the landfill mining project organization.

The few exceptional cases displaying economic incentives concern more high-grade industrial deposits and/or seem to rely on other drivers than resource recovery such as high alternative site-specific costs (e.g. for remediation or creating new landfill space) that could be reduced or postponed through landfill mining, i.e. integrated remediation and resource recovery. However, given the lack of real-life projects, there are still large uncertainties regarding how current policy and market conditions will influence feasibility of landfill mining and such implications are often not fully addressed in the assessments. Some studies conclude that current landfill taxes and prohibitions could, if applied on the landfill mining residues in need of re-deposition, generate significant waste disposal problems and additional project costs (for an example, see Figure 1).

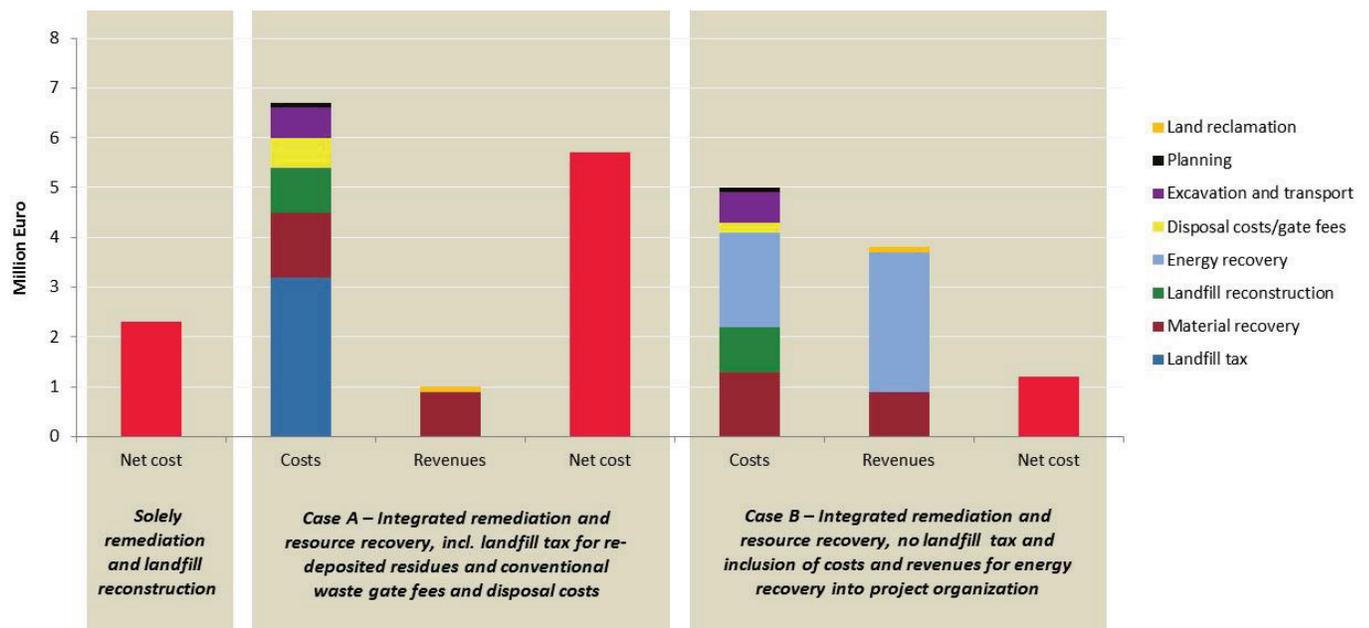


Figure 1 Private economic assessment of solely remediation vs. integrated remediation and resource recovery of a typical Swedish municipal landfill (Frändegård et al., 2015). Potential economic impacts of the landfill tax and business models including costs and revenues for energy recovery into the project organization are also shown (i.e. difference between Case A and B).

When it comes to the restricted number of recent environmental evaluations of landfill mining, they are based on life cycle assessments and all of them except one only consider climate impacts. In essence, the reviewed studies cover two different types of municipal landfills (i.e. the Remo landfill in Belgium (several studies) and an estimated average Swedish municipal landfill) and an industrial shredder deposit. What is interesting is that the assessments on the municipal

landfills display both positive and negative climate impacts, mainly dependent on to what extent landfill mining contributes to avoided emissions from long-term landfill gas generation and replaced primary material production and energy generation (Figure 2). Such differences in performance drivers among the studies are in turn influenced by several inter-related parameters such as landfill-specific factors (e.g. specific material composition, landfill gas potential and treatment), project settings (e.g. resource intensity and efficiency of selected separation and treatment technologies), systems conditions (e.g. assumptions about the extent and type of avoided primary material and energy production) and modelling choices (e.g. applied landfill gas generation and carbon footprint models and accounting principles for biogenic emissions generated during thermal treatment processes).

Regarding more homogenic and inorganic deposits rich in both ferrous and non-ferrous metals, i.e. the industrial shredder landfill, drivers for obtaining net avoided climate emissions seem to be less complex and more or less solely related to the efficiency of metal recycling and thus avoided virgin mining activities. These results clearly motivate further research on how landfill mining should be evaluated and which critical factors and conditions that needs to be in place in order to obtain climate-motivated projects.

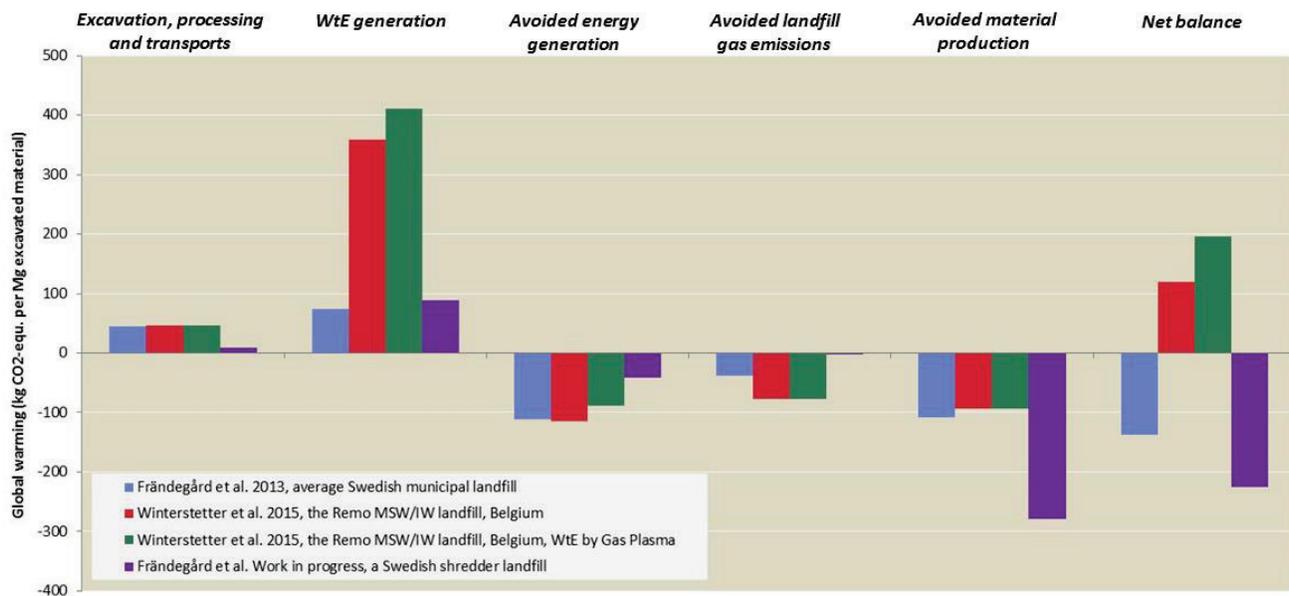


Figure 2 Reported added, avoided and net emissions (in kg CO₂ equ. per Mg excavated material) for three different landfill mining case-studies involving different landfills, separation and treatment technologies, system conditions and modelling principles (based on Frändegård et al., 2013; Winterstetter et al., 2015; Frändegård et al., submitted).

Although climate emissions could be seen as a first order indicator for resource-and-energy-related impacts, such a perspective is also insufficient given the multitude of societal and environmental consequences that landfill mining potentially could generate. In one of the several assessments performed on the Remo landfill in Belgium, for instance, it is concluded that landfill mining would be unfavorable from a climate perspective but provide benefits in all

other studied impact categories (e.g. acidification, eutrophication, particular matter, metal depletion, fossil depletion, land occupation, and so on). There are however still several important societal impacts that typically are not accounted for in Life cycle assessments such as local, site-specific pollution and health risks and the societal resource potential of landfill mining. For example, in Sweden, municipal landfills are estimated to contain base metals, waste fuel and earth construction materials of such quantities that if exploited they could double the annual domestic flows of these secondary resources for decades – injecting additional circularity and resource autonomy into the economy.

Key challenges for facilitating landfill mining implementation

When going through the literature, it is striking that virtually all initiatives and projects have involved more or less “randomly picked”, heterogenic and “low-grade” municipal landfills. Given the importance of site-specific factors for the environmental and economic outcome, reliable prospecting methods and knowledge that enable a more strategic selection of suitable landfills for mining are needed. With that said, increasingly going for more homogenic and metal-rich industrial landfills, preferable then poorly equipped sites that already involve high alternative costs in terms of urgent remediation needs, might be a fruitful and complementary development of the area. Such a strategy displays clear similarities to the evolution of primary production, in which the most mineral rich deposits were mined first and then in tandem with progress in prospecting and mining technology more “low-grade” reservoirs became exploitable.

Although the reviewed studies provide important insights of how to execute landfill mining on some specific landfills, the knowledge is yet limited and also largely theoretical implying a need for comprehensive real-life projects developing feasibility and performance further in practice. For instance, virtually all of the assessments simply assume that most of the separated materials and energy resources will be accepted by the markets – an assumption which if not true will have detrimental environmental and economic implications. In a recent Swedish pilot project on an industrial shredder landfill, only the metals turned out to be directly saleable while the remaining 90 weight-% of the extracted materials (i.e. mainly mixed combustibles, fines and aggregates) did not fulfil current market demands and/or the strict regulatory values for organic and heavy metal contents (for some of the materials not even for re-landfilling). This was so despite of that a highly advanced separation plant was used involving multiple and sequential processes such as screeners, air classifiers, magnetic and eddy-current separators, sink-and-float units, and so on. However, the challenge does not stop there. Simply speaking, in order to economically motivate resource recovery from landfills, it is not enough to be able to extract just some usable commodities (e.g. metals, combustibles and aggregates), but the final outputs must involve fine raw materials generating significant net revenues to the project (high-quality RDF instead of mixed combustibles, plastics for material recycling, and so on). Such a challenge will most likely not be solved by solely focusing on technology development and/or hoping for significantly increasing raw material prices. New business models, an extended collaboration with (end-user) material and energy companies and adjusted regulatory demands on secondary resources (e.g. limit values for heavy metals, which in some EU-member states are extremely strict) are presumably equally important measures for securing a high-added value output for the extracted resources.

What history tells us is that the evolution of any industrial sector relies on significant and continuous investments in technology innovation, specialization and know-how. In principle, such long-term learning processes and curves can only be sustained and motivated by a clear political direction and support (cf. renewable energy technologies). When it comes to landfill mining, however, such a political commitment is absent. For instance, the European landfill policy framework instead strongly advocates isolation, control, final closure and post-monitoring for decades. Private actors' willingness to invest in repeated pilot trials and gradual learning about how to best plan, organize and execute landfill mining is further restricted by unclear regulatory and market conditions, for instance, regarding how the landfill tax, prohibitions and after care obligations will be applied on this type of projects. On a more general level, a central policy challenge for landfill mining is that most of the main (potential) benefits only materialize on the societal level. Several of the reviewed studies therefore stress the importance of policy measures for internalizing such externalities into private actors' economy. However, given the limited and also varying results provided so far, I believe that there is a lot of work left to be done, not the least when it comes to developing more comprehensive, precise and commonly accepted methods for specifying the various environmental and societal impacts that landfill mining projects could generate.

This extended abstract is based on the following references:

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ENHANCED LANDFILL MINNIG – A CASE STUDY IN BELGIUM

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Introduction

The goal of the Closing the Circle project at the Remo landfill site is to set up an activity that allows to gain revenue from the mining of landfilled materials. The Remo site is one of the biggest landfill sites in Flanders with more than 18 million tons of waste on an area of more than 140 ha. The site is in operation since the 70's and divided in several cells. Part of the cells is dedicated to municipal solid waste and the other part to industrial waste. The landfill is currently only accepting industrial waste. The annual accepted amount is more than 200.000 tons. In the past up to 5 times more waste was stored annually. Besides gaining revenue, it is the ambition to mine as much as possible materials and integrate the excavated area in the surrounding nature reserve.

The final destination of the area is thus nature and not for example residential area. Therefore, the business of this project cannot rely on the value of land which is low. Simple and fast excavation to free land and reallocate the waste to other landfill sites is in this case not an option. That is way the basis for the economics of the project must be the waste itself.

Stakeholders (the community, local politicians, ...) see advantages in the mining operation taking in to account the later integration in the surrounding nature reserve, acceptable impact on the neighborhood by selection the best available technologies and involvement of the local community.

A project was designed taking in to account all the stakeholders requests. Instead of fast excavation and reallocating waste, a long term project was set up in different steps taking more than 20 years to excavate and treat waste on site with innovative technologies. The selected technologies and valorization strategies need to fit in the EU's strategies and ambition for environment, waste and resources.

Status of the project

Although GM is working already since 2008 on the closing the circle project, the project did not start yet on an industrial scale. Several research activities were conducted to learn more about the quality of material in the landfill and of technologies for excavating, separation, sorting and valorization. Life Cycle Assessment was conducted to clarify the environmental impact of the full process and of the impact of the different unit operations. The assessment was used to identify

and improve the process steps with a high environmental impact. After research, the preparation for permitting was initiated.

The permitting process consisted of several steps and is almost finished. The Remo landfill operation is conceived as such that at the end nature with specific value is installed on top of the landfill body. Along the years, that nature got protected. In order to allow renewed industrial activity on the site, the destination of the area had to be changed again from nature in to industrial area. That process already took several years. The landfilling activity, which is still going on, is part of a process consisting of the excavation of sand, commercialisation of sand, landfill construction, landfilling, closure of a landfill cell, installation of the final destination of the landfill which is nature.

The Environmental Impact Assessment was initiated and performed by several independent environmental experts (air, water, noise, nature, human, ...). This study is currently finished and the different public departments are consulted and asked to provide an advice or come up with additional questions.

In order to interact with the local community and with academia, several activities and consortia were set up. These communities increase awareness of the potential of landfill mining and stimulate strategic thinking and the stimulate selection/development of new technologies and products for landfill mining.

One of the outcomes to differentiate between a classic landfill mining activity in which an excavation is performed to generate space and/or make value from land in a short time was to define the Enhanced Landfill Mining (ELFM) activity. **Enhanced Landfill Mining is defined as “the safe exploration, conditioning, excavation and integrated valorisation of (historic, present and/or future) landfilled waste streams as both materials (Waste-to-Material) and energy (Waste-to-Energy), using innovative transformation technologies and respecting the most stringent social and ecological criteria.”** ELFM is part of a wider view of a circular economy and is perfectly complementary to urban mining and recycling in general (Eurelco, 2015).

Besides compiling the environmental permit, also the building permit is under construction. Several engineering companies are appointed to define the technical requirements for a waste treatment plant, to perform the engineering work that will lead to the construction of a plant and to work out a building permit.

Drivers and Barriers

The main contribution of this work is the discussion of drivers and barriers of the Closing the Circle Enhanced Landfill Mining project of Group Machiels in Houthalen-Helchteren (Belgium). This abstract addresses some drivers and barriers giving a first impression of the discussion.

Drivers

Business: Not only company drivers are required to get a successful landfill mining project. Also stakeholders (community, authorities, academia) should be convinced that there is something in

there for them as well. The company is willing to set up a business and perform landfill mining projects throughout the world including the mining of landfills for which the value of land is low and cannot be the main or the only driver. The Remo landfill site in Houthalen-Helchteren is owned by Group Machiels. The Closing the Circle project should serve as a first example of EFLM. Group Machiels will learn from this experience and replicate at least parts of it in other projects. A spillover of the landfilling mining activity at the Remo site is that Group Machiels will increase its activity in the fresh waste market.

Jobs: The local authorities are interested in the generation of additional jobs in the community. Houthalen-Helchteren is a community with an immigration history thanks to the past coal mining activity. Although this immigration took place 3 generation ago, the community still suffers from a relatively low degree of integration and education. The degree of unemployment is close to 10%. The average Flemish unemployment rate is 7.44%. The combination of specific demographic characteristics, low degree of integration, relatively high degree of unemployment showed to be a flammable cocktail in certain parts of the community. Projects generating jobs for low educated people is therefore high on the agenda of the community but also on the agenda of the province and of national policy. The ambition to export knowledge from the Closing the Circle project is of communities interest because it will as well provide additional job opportunities.

Environmental benefit: Local stakeholders see the advantage of the innovation in the region that landfill mining project can generate. New companies can provide service or generate activities linked to the landfill mining activity. Another driver for local stakeholders to support the project is the later integration of land in one of the biggest Flemish nature reserves increasing the ecological value of the area and increasing eco-tourism. The mining of waste guarantees that the surrounding soil and ground water will not be polluted several generations later.

Barriers

Selection of Concept and market: The Closing the Circle project was conceived in 2008 at a time that the cost for materials and energy were sharply increasing. The need for alternatives was proposed to be urgent and later on also Europe developed its Resource Efficiency strategy. In line with these signals, the Closing the Circle project was conceived as a project to recover materials and generate power. Support mechanisms for power generation were in place and innovative technologies almost ready. Group Machiels performed a wide range of tests supporting the idea to build a large plant able to treat all waste present in the landfill in 20 years. The idea was to excavate waste, perform separation and sorting generating as main product a refuse derived fuel (RDF) for conversion in to power using gasification technology. The RDF partially consists of bio-mass (wood, paper, textiles, ...). Power production could benefit from green current certificates helping to establish an interesting business case. Later on, partly due to the crisis, the government decided to reduce this support mechanism, urging for a change in the approach of the project. Although Europe promotes resource efficiency no significant increase in the demand for recycled products is observed. Instead of producing power, more advanced technologies are currently included in the project allowing to make synthetic natural gas (SNG) and/or hydrogen. Particularly the production of hydrogen from syn gas seems interesting provided that the market is ready to pick up all the hydrogen that can be produced.

Instead of building directly a big plant, it was decided to start with a smaller demo plant to reduce the investment risks and scale up in three steps.

Waste characteristics: Waste from landfills typically contains much fines. The majority of fines in landfilled municipal solid waste result from the bio-degraded fraction and sand used for daily cover. Fines of industrial waste is very divers and results from sludge, ashes and landfilled polluted soil. Removing these fines in order to generate quality material useful for recycling or for making RDF requires intensive separation and sorting. Besides the fines content, waste from old landfills can still be very wet requiring drying for good separation. The fines in combination with the moisture content is a major disadvantage of landfilled waste compared to fresh selectively collected waste. Another disadvantage of landfilled waste is that separation and sorting cannot be paid by gate fees. The Closing the Circle project will therefor combine fresh and landfilled waste streams. Combining both streams improves the business case of the project.

Permitting: In the past, landfills were considered to be lost land. Installing nature on top of a closed landfill was a strong argument for improving the quality of a landfill area. It made landfilling more acceptable and allowed the continuation of the landfill activity. At that time, there was no interest in the mining and often nature on top of landfills, like a normal nature elsewhere, got protected making the mining a lot more difficult. This made permitting a lot more difficult for the Closing the Circle project. Several stake holders are more in favour of the existing nature than of a renewed industrial activity. The timely loss of nature needs to be compensated. The excavations need to be organised as such that disturbance of the existing ecosystems is minimized.

Conclusions

The potential of landfill mining in Europe and abroad is discussed in the several studies and by several research consortia such as EURELCO. It is clear that the number of landfills is huge. It is however not always clear how to make a landfill mining case a success. It seems that some kind of support mechanism is required, certainly in the current economic conditions with low prices for gas, fuel and power. The Closing the Circle project envisages to generate business based on the valorization of excavated waste. The use of advanced technologies and the production of new products show that it is possible to obtain an interesting business case. However, the business depends of the value of products such as hydrogen, methane, recycled plastics and new types of building materials and products. The market for these products is not mature yet. A coordinated push to stimulate these new markets and the market uptake of recycled products together with a push of landfill mining is needed. It is not questioned that landfill mining will once become normal practice. The question is when and what push is required to get there.

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CRITICAL FACTORS FOR THE CLIMATE IMPACT OF LANDFILL MINING

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Introduction

Landfills have been the major way of waste disposal on a global level. Although direct landfilling of waste is on the decrease in most modern waste management systems, particularly in Europe, landfilling remains the dominant practice for handling waste in many parts of the world (OECD 2008). Therefore, in Europe several hundreds of thousands of old landfills exist, which are associated with long-term environmental impacts (e.g. El-Fadel et al. 1997), extensive aftercare periods (Laner et al. 2012), and land-use restrictions potentially interfering with regional development plans. Landfill mining has been proposed as an innovative strategy to address these implications and to recover valuable secondary raw materials (Jones et al. 2013). However, in the past landfill mining has been mainly used to solve traditional waste management issues such as lack of landfill space and local pollution concerns (Krook et al. 2012).

The potential of landfill mining for mitigating environmental pollution and valorising deposited wastes via material and energy recovery was investigated in various studies addressing ecologic and/or economic implications of landfill mining. Danthurebandara et al. (2015) assessed the environmental and economic performance of the Enhanced Landfill Mining (ELFM) project, highlighting the importance of factors related to the energetic valorisation of the excavated wastes for the results of the assessment. The crucial importance of case-specific factors for the evaluation of a landfill mining project was also emphasized by Winterstetter et al. (2015), who used the ELFM project as a case study to illustrate the application of resource classification systems to anthropogenic material stocks. In addition to economic factors, both studies assessed the potential contribution of the ELFM project to global warming in a life cycle assessment and found that mining the landfill would lead to higher greenhouse gas emissions compared to the do-nothing scenario. In contrast, Frändegård et al. 2013 show that the potential mining of old landfills in Sweden would result in a lower climate impact than the do-nothing alternative. Although all of these studies state that the actual environmental performance of landfill mining depends on various case-specific factors, so far a systematic assessment of the importance of individual factors for the environmental impact of landfill mining is missing (cf.

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Krook et al. 2012). This research gap is addressed by the present study, which aims to quantitatively assess the importance of specific factors and conditions for the contribution of landfill mining to global warming. Therefore, we identify site-specific factors, project settings and system conditions, which are potentially relevant for the climate impact of a landfill mining project. Based on the investigation of the influence of these factors, settings, and conditions, respectively, on a landfill mining's contribution to global warming, we discuss the practical implications of our findings in terms of strategies and measures for implementation of landfill mining.

Material and Methods

The focus of this study is on the systematic investigation of critical factors for the climate impact of landfill mining to identify situations when landfill mining can be seen as a measure to mitigate global warming. Therefore the analysis is limited to aspects which may differ from one landfill mining project to another (e.g. composition, upgrading technologies, background energy system, etc.) and does not specifically address the effect of modelling choices on the results. Instead, the modelling builds on recommended/standardized methods and principles.

Based on previous research addressing economic and environmental implications of landfill mining, relevant factors for assessing the climate impact of landfill mining are identified (see Table 1). We distinguish between site-specific factors, project setting, system conditions, and intermediate factors between project and system level. Site-specific factors are specific to a certain landfill, including the reference case (i.e. do-nothing alternative), such as the composition of the deposited materials, the location of the landfill, the landfill infrastructure, etc. Contrary, project settings are more or less open for deliberate choices such as the objectives/ambitions of the project (e.g. resource recovery, land reclamation, development of landfill void space, etc.), employed separation and processing technologies, as well as anticipated markets for secondary raw materials. System conditions are external to the landfill mining project and cannot be influenced by the authority of an individual actor such as prevailing electricity and heat generation systems, primary material production systems, or landfill aftercare obligations. Intermediate factors between project and system level are to some degree determined by the project setting, but also depend to some degree on the system conditions. For instance, the energy recovery technology employed within the project is a project-specific choice, but it also depends on available plant infrastructure and valorisation options for the products (e.g. heat and electricity).

Several different sets of parameters are defined for each of the eight investigated factors listed in Table 1. The composition of the landfilled waste and its landfill gas (LFG) potential are considered via three different sets (#1). Alternative 1 represents a relatively young municipal solid waste (MSW) landfill with a high amount of metals and combustible materials as well as with a high LFG potential. Alternative 2 describes an average landfill composition and associated LFG potential based on various landfill mining case studies. Alternative 3 is representative for a rather old MSW landfill with a low amount of combustibles and a low remaining LFG potential. Three different sets of parameters are also considered for the reference case (#2), the excavation and separation technology (#3), the energy system (#6),

and the primary raw material production system (#7). Only two different sets are considered for the quality of the separated material and its market potential (#4) as well as the transport of recovered materials (#8), whereas the Waste-to-Energy treatment is reflected via five sets (there are Alternatives 1A and 1B and 2A and 2B, respectively, cf. Table 1). Overall this results in 4860 ($=3*3*3*2*5*3*3*2$) possible combinations of the different sets. Thus, the climate impact of landfill mining is calculated for 4860 scenarios in total, each scenario representing a specific combination of the parameter sets outlined in Table 1.

Table 1 Selected factors to be considered by different alternative parameter sets in the analysis of the climate impact of landfill mining. Each alternative set for a specific factor designates a specific situation which could be encountered in a landfill mining project.

#	Factor-type	Description	Alternative 1	Alternative 2	Alternative 3
1	Site-specific	Landfill material composition and landfill gas potential	New, rich MSW landfill	Average MSW municipal landfill	Old, poor MSW landfill
2	Site-specific	Landfill gas emissions of reference case	Reference: no LFG collection	Reference: Active collection + destruction/oxidation	Reference: Active collection + utilization
3	Project setting	(Excavation) and material separation	Mobile separation unit	Advanced separation plant	Future potential
4	Intermediate project/system level	Quality of separated material and raw material potential	Low	-	High
5	Intermediate project/system level	Waste-to-Energy (WtE) treatment	Incineration, average efficiency. A: Basic metals recovery; B: Advanced metals recovery	Incineration, high efficiency. A: Basic metals recovery; B: Advanced metals recovery	Gas Plasma technology (ELFM project) with advanced metals recovery
6	System condition	Energy systems (heat & electricity)	Heat and electricity mix with high fossils share	European average	Heat and electricity mix with high renewables share
7	System condition	Primary material production systems	Low efficiency & high fossil intensity	Average production	High efficiency & low fossil intensity
8	Intermed. project /system level	Required transport for recovered materials	Small distances	-	Large distances

The different model parameters are quantified building on data from previous studies and literature (e.g. reported variations in landfill material compositions, separation and recovery efficiencies, material qualities, etc.) and life cycle data bases (i.e. inventory data are retrieved from the ecoinvent version 3, Swiss Centre for Life-Cycle Inventories (2014)). In addition to the alternative parameter sets of the eight factors in Table 1, several groups of model parameters are the same for all scenarios. These set-independent parameters are related to the physical properties of the excavated waste fractions (e.g. water content of organic waste), physical constants (e.g. molar mass of CO₂), the management of re-deposited materials, and the characterization factors used to quantify the global warming potential (e.g. GWP₁₀₀ of 1 kg CH₄ is 25 kg CO₂ equivalents) of an emission.

The climate impact of landfill mining is calculated for a functional unit of excavating one metric tonne of waste. The life cycle assessment model to determine the effect of mining one tonne of

waste on global warming consists of 283 parameters in total. Except for physical constants, all parameters are probabilistically treated in the model. Hence, they are defined via normal distributions given by mean value and relative standard deviation. The relative standard deviation is fixed to be 10% for all uncertain parameters, because the investigation of parameter variation aims at identifying the most important parameters in the model (i.e. sensitivity analysis) and does not aim at quantifying realistic ranges of output variation for a specific scenario (this is covered by scenario analysis). The model results are calculated using Monte Carlo Simulation with 1000 runs. Thus, 1000 results are produced (summarized via mean value and relative standard deviation) for each of the 4860 scenarios. The model calculations are performed in MATLAB® (MathWorks, Version 2013a).

In this study, two kinds of global sensitivity analysis are performed (Saltelli et al. 2008). i) Variance-based methods are employed to investigate the effect of factor variation on the variation of the scenario results. Sensitivity indices are calculated with respect to the variation of a factor and its effect on the scenario outcomes. Thus, the scenario analysis allows for identifying critical factors for the climate impact of landfill mining. ii) Step-wise regression modelling is used to determine the most important parameters for each scenario calculation. The frequency of a parameter being included in the regression models of all the scenarios indicates the general relevance of this parameter among all scenarios.

Results and Discussion

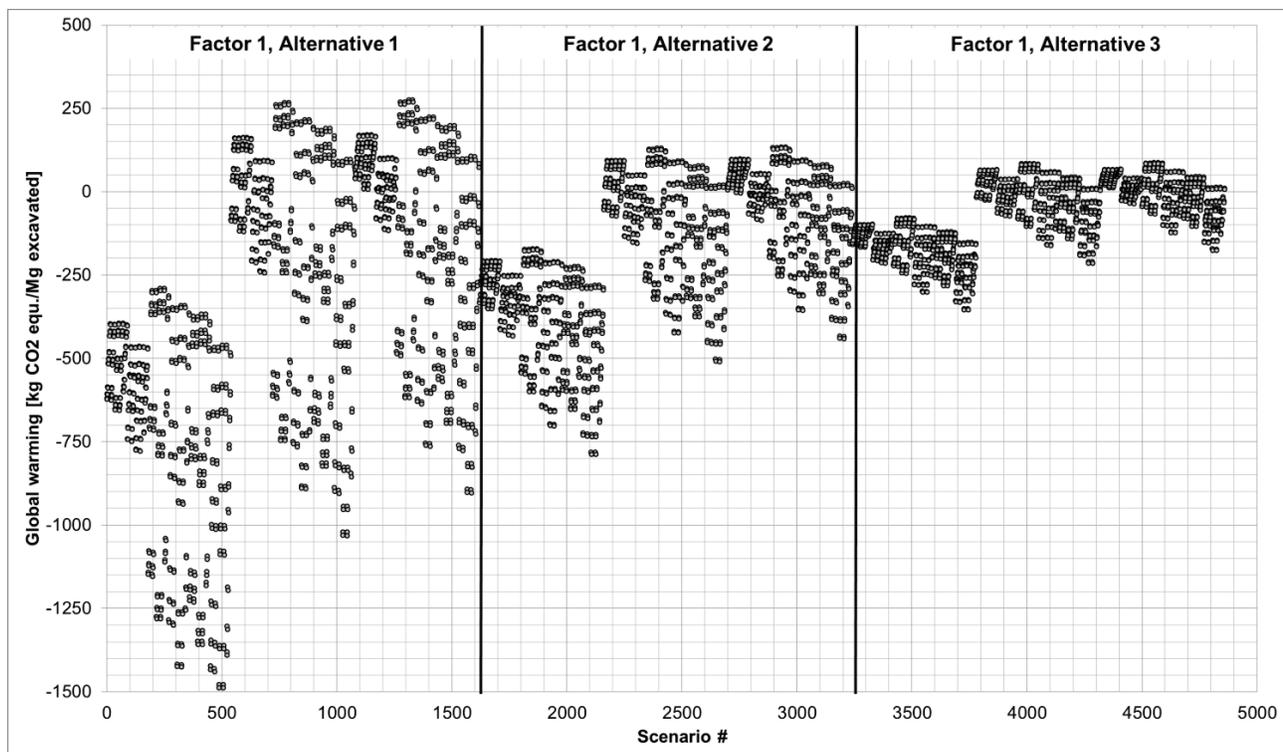


Figure 1 Plot of the scenario results (mean values) for the contribution to global warming due to mining one tonne of landfilled waste. Three sections are separated by vertical lines highlighting all scenarios with the same set for Factor 1 (waste composition & LFG potential).

The scenario results for the climate impact of landfill mining are shown in Figure 1. The results range from a saving of 1500 kg CO₂ equ. per Mg of excavated waste to a positive contribution to global warming of 250 kg CO₂ equ. per Mg of excavated waste. The mean over all the scenarios being -204 kg CO₂ equ./Mg of waste (saving). The clustering of the results plotted in Figure 1 indicates that specific factors are more important for the scenario outcomes than others. For instance, it is visible that the realization of Factor 1 (Landfill material composition and LFG potential) has a significant effect on the spread and tendency of the scenario results. In case Alternative 1 (new landfill, high content of valuable and combustible material, high LFG potential) is chosen, the spread of the results is very large including the most beneficial as well as the worst scenario outcomes. Whereas if Alternative 3 is chosen, the results' range is much narrower and the potential for high savings of greenhouse gas emissions due to landfill mining is lower. From this crude graphical analysis, it can be expected that Factor 1 is important for the climate impact of landfill mining.

The importance of different factors for the variation in the scenario results is presented in Table 2. Variance-based sensitivity measures indicate the direct effect of factor variation on the scenario results variation (first-order effects) and the total contribution (including interaction effects) of factor variation on the scenario results' variation (total order effects). From Table 2 it is obvious that variation of Factor 2 (Landfill gas emissions of reference case) has the highest direct effect on the scenario results ($S_i=0.3088$). Factor 2 has also the highest total effect on the output variation ($S_i=0.3865$), but higher-order effects (difference between first-order and total-order indices) are less important in case of Factor 2 compared to the other most important factors 1 (Landfill material composition and LFG potential) and 6 (Energy systems). This is explained by Factor 2 being mainly associated with direct greenhouse gas emissions (i.e. methane emissions) and the factors 1 and 6 interacting with many other factors in the model. Thus, what would happen to the landfill, in particular with respect to LFG management, in case the landfill is not mined, is a very important factor with respect to the climate impact of landfill mining. Other important factors are the landfill composition and the associated LFG potential (F1), the heat and electricity system (F6), and the sorting technology applied for the excavated waste (F3).

Table 2 Global sensitivity indices to characterize the direct contribution of a single factor to the output variation (first-order indices) and the total contribution of a single factor (including interaction effects) to the output variation (total-order indices)

S_i	Factor 1	Factor 2	Factor 3	Factor 4	Factor 5	Factor 6	Factor 7	Factor 8
First-order indices	0.1574	0.3088	0.0662	0.0181	0.0034	0.1808	0.0093	0.0001
Total-order indices	0.3640	0.3865	0.1555	0.0484	0.0088	0.3276	0.0137	0.0001

Finally, the step-wise regression analysis identifies model parameters related to the characterization factors used for greenhouse gas emissions (GWP₁₀₀ of CO_{2,fossil} and CH₄), the LFG management in the reference case (CH₄ oxidation rate, collection efficiency, CH₄ content of LFG), the LFG potential of the re-deposited waste (i.e. fines), the recovery of metal scraps (substitution ratios for Al and Fe scrap, CO₂ equ. associated with primary metals production), as well as to the LFG potential and metal contents of the excavated waste as being relevant for the output variation in more than 90% of the scenarios.

Concluding remarks and outlook

Based on the analyses performed in this study, it was possible to identify critical factors for the climate impact of landfill mining and investigate their effect on the contribution of a landfill mining project to global warming. In most cases, landfill mining leads to a negative contribution to global warming, particularly in situations when the alternative to landfill mining (reference case) would be associated with high greenhouse gas emissions due to significant LFG potentials and poor LFG management. From a climate perspective, strategies for enhanced landfill mining should focus on relatively young landfills with a relatively high content of organic waste, with currently poor LFG management. The landfill mining project should be implemented using advanced material sorting technologies to maximize metals recycling and energy recovery from waste and the recovered energy should be fed into energy systems which still rely to a large degree on fossil fuels.

Future research should address the role of model uncertainty in the evaluation of the climate impact of landfill mining, because methodological and model choices may have a potentially high effect on the assessment results. Such an analysis can build on the set theory approaches and statistical methods used in this work, which allow for a comprehensive investigation of different choices' effect on the model outcomes.

Acknowledgements

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1.7 Session G: Resource Potential of the Building Stock



ON THE USE OF HISTORICAL MATERIAL DATA

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The importance of historical knowledge

The necessity of a sustainable development of the built environment has drawn the attention to the composition and dynamic of the building and infrastructure stock. Several approaches to evaluate their environmental impacts (LCA, MFA, Technology assessment, LCC, etc.) are based on the establishment and interpretation of the (same) construction material flows. The difference of the approaches appears in the definition of the spatial and temporal system limits and in the definition of the reference (functional) unit. The contribution will deal with historical insights from the development of the stock and from an example of urban mining. The size and dynamic of the building stock related flows depend on the composition (what?) of the stock, on the dynamic (how long?) and on the configuration of the artefacts (how and where?). Due to different time spans of the components there is no direct relation between mass input and mass output in a given year. The predictions of future flows are based on models and scenarios, which partially relay on historical knowledge of the stock. The research on building stock is typically of transdisciplinary nature: the problematic is societal and the combination of methods is defined ad-hoc. As an example two questions implying the use of historical information and knowledge will be discussed:

- The historic diffusion of concrete in the German building stock since the 19th century
- The organisation of the reconstruction of the destroyed German cities after World War II

Most research proceeds by combining bottom up approaches (material quantity per building element) and top down approaches (national material production per year). The necessary historic information is however rarely directly accessible, it has to be extracted from text sources (manuals of building construction, statistics) and from the description of built objects (Bauforschung). Among the multitude of construction materials, cement (and concrete) play a particular role. Developed in the middle of the 19th century concrete has become the dominant building material in Europe with a share of more than 50 % (weight) one hundred years later.

The historic diffusion of concrete in the German building stock

Four questions will be discussed:

- How much concrete is there in the building stock and how did it develop (1)
- Where (in which elements) and when concrete appears historically (1)
- What is the influence of scarcity and substitute materials (1)
- What are the main factors for the future of concrete (2)

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a) The development of the cement production:

Over a period of 150 years the evolution of the German cement production resembles a logistic function. The long invention phase in the 19th century was followed by the innovation phase (begin of the massive diffusion) around 1910. The two world wars produced reductions that were rapidly compensated in the post war boom periods. The end of the boom and crisis of 1972-74 is reflected by a strong reduction and marks the beginning of the saturation. The short boom following the German reunification did not allow reaching the 1973 peak level any more. The flattening of the curve is the sign that “Germany is built”. After a long development of new construction and little demolition, the growing investment needs of renovation limit the share of new construction. The worldwide centre of the (logistic) development of cement and concrete has shifted to China and partially to India. The average age of these young, fast growing, stocks is still diminishing where as the average age of the stable European stocks increase. The growing interest in urban mining results also from these structural changes In the historic diffusion of concrete in the stock sub-process appear:

b) Substitution process

The three components of concrete: binder, aggregate and additives have not developed linearly. Regional scarcity has led to substitution process, which did however not change the long-term domination of Portland Cement. This might change in the future through the use of recycled aggregates, the diversification of binders and the development of new binders with lower environmental impacts.

c) Diffusion of concrete in the stock

Between 1860 and 1920 the concrete share in domestic buildings was below 10 % weight. This fraction has increased gradually and reaches today more than 50 % in most building. The application started in foundations and basement walls. In the next step combined reinforced floors replaced the wooden floors. Columns, stairs and balconies followed. Flat roofs were realised in concrete. Concrete did not simply replace other materials; it also changed the interfaces between elements, the performance of the envelopes and the succession of construction operations. Through the continuity of concrete structure the static system and the seismic behaviour changed completely. In the first part of the 20th century the concrete construction techniques were still regionally diversified. Many systems and patents had a promising departure but disappeared again. The historic information is often reformulated into typological classifications without the necessary historical validation

d) Cement and concrete technologies

Not all industrial concrete building technologies have been successful. An example is heavy prefabrication (panels) where in spite of a large share in post-war reconstruction and domination in Eastern Europe, the technology stabilized at a low level. The German cement consumption between 2000 and 2014 has diminished by nearly 10 %. The large European cement producers gradually orient their activity to fast developing countries. Concrete will continue to be the dominating building material; there is no alternative material insight. The main problem will be the reduction of the environmental impact of the cement production. Despite considerable efforts the cement industry will remain one of the main industrial contributors to global warming. For the reduction of environmental impact three solutions are developing in parallel:

- Increase the life span of the buildings i.e. reduce the obsolescence instead of demolition-reconstruction. The impacts per year can be distributed over a longer period
- Reduce the environmental impact of the cement production (green chemistry (2)).
- Increase the recycling of concrete with the aim of closing loops

2. Historic examples of urban mining

The efficiency of urban mining process and recycling of building materials (closing loops) is based on information about the material deposits, the development of deconstruction process and the logistic integration of operation, renovation, deconstruction and new building process. It could be interesting to look for similar historical experiences and possible lessons for the present situation: What can be learned from the reconstruction following massive destruction of the German building stock after the second world war ? Urban mining was the only solution to reconstruct the large German cities. How this was possible in detail is only partially documented. Under the dramatic circumstances a book, "Trümmerverwertung" (recovery of debris) published in 1947 played an important role (3). The authors had already begun to work on it in 1944 anticipating the end of the war. The situation in Berlin was dramatic: 30 % i.e. 420 000 apartments were totally destroyed. 75 % of the female population of 1938 remained in town. There were practically no new construction materials, little energy, destroyed infrastructure and few qualified workers. The problem was to clean out the debris, transport and recover useful building materials and components and of course reconstruct the buildings with the debris. The existing tramways were used for transport, the main work force were women (Trümmerfrauen). The book centralised and classified the existing information about the composition of buildings and of the reuse of debris. It presented a complete chain to analyse and qualify the ruins and the debris, to estimate the quantities of available materials, to choose the best recycling techniques and quality criteria and to organise the deconstruction and reconstruction process. The basis for new standards for recovered building materials were developed in short time. The 277 pages of the book reflect the complexity of the situation and define pragmatic and flexible solutions.

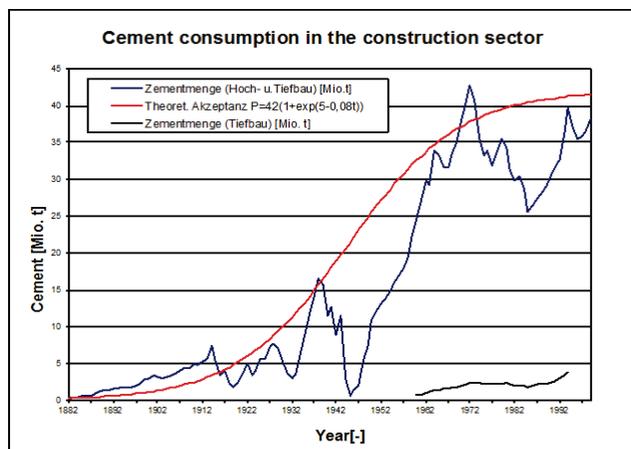


Fig. 1 Cement production in Germany since 1882 (1)

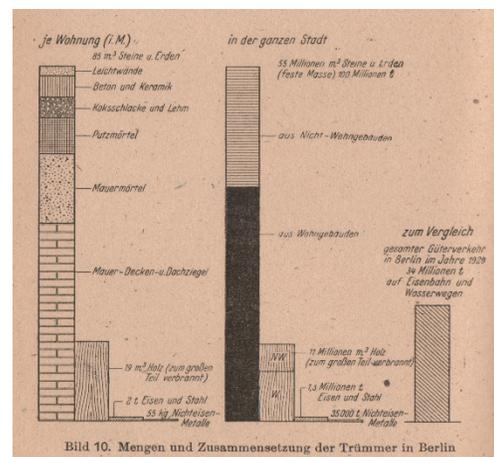


Fig 2: Quantities of debris per apartment (2)

4. Some conclusions

Concrete has become a dominating buildings material in Europe in 100 years. It gradually replaced a number of other materials. A similar process takes now place in the rapid construction of the buildings stocks of developing countries. The diffusion process seems to follow a logistic curve and the distribution inside the buildings is rather similar. There are still considerable possibilities to diversify the material, find new applications and reduce the environmental impact. There seems to be no alternative to concrete. Historic information about this process allows to improve the maintenance, deconstruction and reuse process and to appreciate the social and cultural value of buildings and groups of buildings. Deconstruction and urban mining are considered as relatively new fields of knowledge. In fact there are examples of recovering of societies after catastrophes where the reconstitution of the building stock was a central problem and where practically the only resources were debris. The success of the reconstruction as a type of resilience was only possible through the combination of human capital (skills and knowledge) and social organisation. It shows that successful examples of resource logistic management of resources are based on shared information and efficient social organisation.

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BUILDINGS AS AN URBAN MINE – THE CASE STUDY OF VIENNA

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INTRODUCTION

Buildings and network infrastructure are greatly contributing to material turnover and accumulation within cities. Besides gaining knowledge about the material stock in buildings in the city of Vienna, information about waste flows arising through demolition activities are investigated within this research. The recycling of construction and demolition waste (CDW) has a significant potential to reduce the use of primary resources and landfill space. To make use of this potential, buildings play a crucial role, as they on the one hand are responsible for a large share of the overall CDW, and on the other hand they are much more complex with regard to their material composition compared to civil infrastructure (e.g. roads, railways, pipe networks). This complexity also makes material recycling, aiming at the production of high-quality secondary raw materials, much more challenging. Detailed information about the material composition of buildings, however, may facilitate high quality recycling. Hence, the aim of the present study is to determine the material composition of buildings in Vienna, to categorize this information, and to finally combine it with data about the building structure (spatial distribution of different building categories). This allows estimating the overall material stock in buildings, and mapping the material distribution in a resource cadaster. Based on the resource cadaster and on information about the demolition activity the amount and composition of CDW can be estimated.

METHOD

Material intensities

Case studies

In a first step, 14 buildings of different utilization and construction period are investigated regarding their material composition prior to their demolition (see Kleemann et al. 2014). Built-in materials are quantified before the demolition of the building through:

- analysis of available documents (construction plans, expert's reports) and
- on-site investigation and selective sampling. During the on-site investigation, data about built-in materials are collected through measurements and selective sampling prior to the deconstruction/demolition of the building. It includes the dismantling, weighting, and measuring of components such as windows, doors, partitions, ceiling suspensions, floor and roof constructions, wires, pipes, etc.

Based on the collected data, information about the different built-in materials is aggregated, and the mass is calculated based on volume, area or number of the particular material and on data

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about its specific density or weight. The built-in materials are then divided by the gross volume (GV) of the buildings in order to determine comparable specific material intensities per volume (kg/m^3 GV) for each building category.

Construction plans of demolished buildings

In order to acquire a higher number of buildings sampled, in a second step available construction documents (mainly construction plans – Figure 2) of all buildings reported to be demolished in Vienna in the years 2013, are analyzed leading to an additional sample of 40 buildings. Depending on the quality of the documents, building materials used for wall, ceiling and in some cases also for roof or floor construction can be determined. Data gaps, mostly regarding materials of low concentration (e.g. plastics/metals in installations and fittings, copper for electricity, wood/aluminium/PVC for windows), are filled by using specific material intensities taken from the 14 buildings analyzed in detail.

Data about new buildings and literature

Not all building types could be covered with the first two steps described. Especially data about new buildings was rarely available. In order to complete the data set, data on the material composition of buildings not covered so far (buildings constructed after the year 2000) were derived from life cycle assessments, tendering documents, construction plans, and accounting documents. To put the collected data in context to other studies carried out in this field, data about the material composition of buildings reported in the literature were used.

Building structure

The building structure in Vienna is analyzed through the collection and combination of different GIS data (Geographical information system) from several municipal authorities. Table 1 shows the sources used to generate a data set which comprises information about GV, utilization, and construction period of each building in Vienna.

Table 1 Data used to investigate Vienna's building structure

Information	Source of information
Area & height of buildings	Department of city surveying
Construction period & utilization of buildings	Department of city planning and land use
Construction period in the building block	Department of city development and planning
Utilization in the building block	Department of city planning and land use
Construction period & utilization of new buildings	Department of building regulation

RESULTS

Table 2 shows the material composition of different building categories in Vienna (kg/m^3 GV), based on the above described data. Not surprisingly, Table 2 indicates a dominant share of mineral materials within buildings (96%). In contrast organic materials (2%) and metals (2%) occur in rather low concentrations. Newer buildings tend to have higher metal contents (mainly because of reinforcement steel) and a lower content of organic material (mainly due to a decreasing usage of wood).

Table 2 Specific material intensities [kg/m³] of different building categories in Vienna (round to two significant digits)

Period of construction	Utilization	Mineral materials	Organic materials	Metals	Total
before1918	residential	390	19	3.1	410
	commercial	430	3.7	4.4	440
	industrial	280	8.5	8.6	340
1919-1945	residential	410	13	4.8	430
	commercial	340	7.1	6	360
	industrial	320	28	5.8	350
1946-1976	residential	430	6.5	7.3	450
	commercial	350	6.5	7.3	450
	industrial	340	6.5	13	350
1977-1996	residential	430	6.7	7.1	460
	commercial	380	1	13	400
	industrial	170	1	15	180
After 1997	residential	380	10	15	410
	commercial	320	5.7	10	340
	industrial	290	5.6	13	310

By combining the data of with geographical data about the building structure of Vienna, the spatial distribution of different materials in buildings can be mapped (Figure 1)

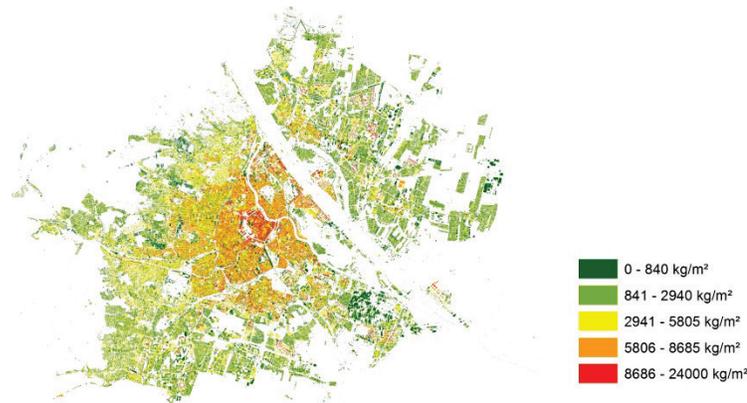


Figure 1: Spatial distribution of mineral material intensity (in Vienna [kg/m² built-up area])

CONCLUSIONS AND OUTLOOK

The described approach allows the assessment of the overall material stock in buildings in Vienna by linking material intensities with data about the building structure (construction period & utilization). Combining the thereby generated data about the material stock with information about the demolition activity in Vienna will allow quantifying the amount and composition of CDW and thus also the materials potentially available for recycling. The data basis on specific material

intensities of different building categories will be improved with ongoing research in this field. This is especially important, as at this stage, not all building types are represented sufficiently in the data base. Finally, the development of a resource cadaster mapping the location and quantities of materials in buildings in the city of Vienna is planned. An outlook of how such cadaster could look like is given in Figure 2.

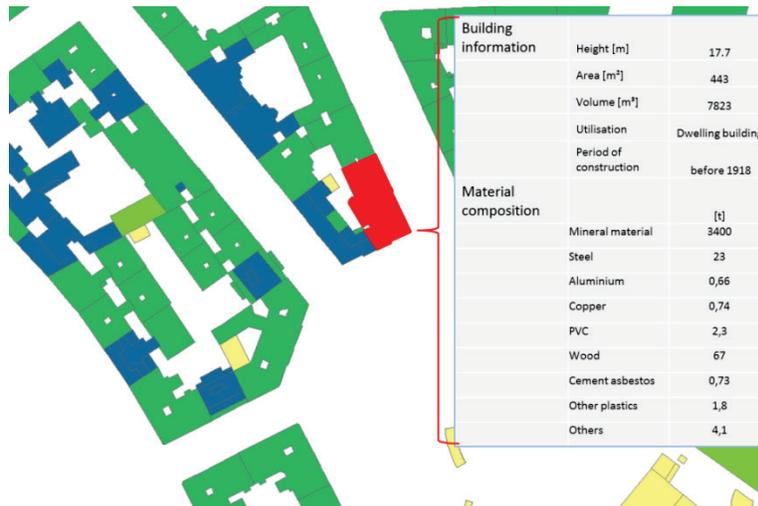


Figure 2. Resource cadaster with material information on a building level

Acknowledgements

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BUILDING MINING - SENSITIVITY STUDY OF RECYCLING POTENTIALS UNTIL 2050

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Overview

The goal of the project was to investigate the potentials of high-value recycling of building rubble and construction site waste in building construction by means of sensitivity studies. For this purpose, the existing material flows for the common construction materials concrete, tile, lime sand brick, porous concrete, gypsum, wood, mineral rock wool and rigid foam insulation material, glass and plastics were analysed. One sub-goal was to research the origins, composition, collection, sorting and recycling paths of the stated material fractions, and to arrive at a consensual agreement with experts from the construction industry and industry association representatives on “optimistic” RC material share for selected construction product groups. An additional sub-goal was to develop a bottom-up material flow model and a quantitative picture of future construction activity – new construction, renovation, modernization and demolition – with a perspective for 2050. The project focusses on mass - eco balance calculations were not carried out.

Concept

In order to calculate the input and output streams of building activities in Germany, the construction (new construction, conversion, modernisation, demolition) activity was quantitatively ascertained top-down, and compared with bottom-up calculations. “Top-down” refers to the use of all available statistics, including the differentiations additionally ascertained and published by the industry associations. “Bottom-up” refers to the material flow model with the aid of which the data of the construction activity statistics for 2010 on residential and non-residential buildings were recalculated into material flows and material stocks. For this purpose, the development of a building typology was necessary which, based on detailed descriptions of projects (plans, structural details, descriptions of the materials), and allows the calculation of the material key values of the design and the technical equipment for representative buildings. In the context of the project, it was thus possible to depict the process of construction along 16 construction product groups.

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Table 1 Building Products - Stocks and Flow 2010 including Recycling aggregates in Germany. (Please note, that the bottom up model covers only the statistically registered building activities) (Deilmann et al. 2014)

Total	2010			
	Stock	Input	Output	RC
Building products	Mio. tons	Mio. tons	Mio. tons	Mio. tons
Concrete	6388,564	41,708	12,228	0,167
Bricks	1874,387	7,905	3,489	0,000
Lime stone	1231,589	5,252	2,055	0,000
Foam concrete	178,758	1,500	0,300	0,000
Other mineral	3484,658	42,055	14,850	2,523
Gypsum board	9,820	0,168	0,032	0,000
Other gypsum	159,003	3,244	0,680	0,000
Construction, wood	295,754	2,710	1,130	0,000
Other wood (floor cover)	39,450	1,017	0,908	0,041
Glas	334,236	2,518	1,171	0,378
Mineral insulation	65,223	0,550	0,235	0,148
Synthetic insulation	83,023	0,874	0,343	0,087
Plastic doors/windows	82,911	0,591	0,209	0,077
other plastics (floors, pipes, wires)	68,333	1,012	0,354	0,010
Metals (wires)	898,428	9,206	3,059	4,603
others (carpets, coating)	62,133	0,757	0,650	0,000
Total	15256,272	121,066	41,693	8,036

In order to answer the question regarding recycling possibilities, ascertainment and preparation technologies for 16 construction product groups needed to be investigated. Moreover, studies and research results and possible admixture of recycled materials were evaluated; however, this generally are innovative recipes which have been tested in only a single case. Points of discussion with representatives of the construction industry thus on the one hand involve how the priority goal of high-value material recycling of construction waste materials could be implemented while taking into account the applicable waste hierarchies and the currently established rules, and which admixtures of recycled material might be conceivable, under optimistic assumptions, for 2030 and 2050, respectively.

The calculations on secondary raw material input in new products of construction are based on information derived from the different building industry associations for the year 2010. It was a difficult process to get the questions above answered by the building industry and to reach consent when it came to estimate the potential use of secondary raw material in new products in an outlook until 2050. The authors had investigated the construction firm's research and experiments on secondary raw material use and confronted them with the technical possible and the economic feasible. The agreement was reached by a promise of the authors to only talk

about sensitivity studies and not about scenarios or prognoses. The basic assumption is that the present framework for RC-Use would change in favourable until 2050.

Test calculations – sensitivity studies – were carried out using assumptions on innovative recycling technologies and potential shares of recycled materials in construction products, and regarding the quantitative development of construction activity generally (2030/2050). Four sensitivity studies were agreed upon with the contracting institution: business-as-usual (BAU), recycling oriented (BAU-RC), sustainable (NA) and NA recycling oriented (NA-RC).

Table 2 Estimated Recycling aggregates in new Products (Deilmann et al.2014)

Building Product	Recycling-Material Admixture in different Products for Construction (in %)		
	2010	2030	2050
Concrete	0,4	6,0	12,0
Bricks	0,0	10,0	15,0
Lime stone	0,0	5,0	5,0
Foam concrete	0,0	2,0	5,0
Other minerals	6,0	21,0	21,0
Gypsum boards	0,0	30,0	50,0
Other gypsum	0,0	0,0	5,0
Construction wood	0,0	0,0	0,0
Wood boards	4,0	10,0	20,0
Glas	15,0	25,0	35,0
Mineral. insulation, incl. ca. 40 % RockWool with RC 0, 15, 20	27,0	42,0	56,0
Synthetic insulation material	10,0	19,0	19,0
Plastic doors/windows	13,0	25,0	50,0
Other plastics/PVC-sheets/carpets	1,0	5,0	10,0
Metals (not part of the investigation)	50,0	60,0	70,0
Others	0,0	2,0	5,0

Results

The general perspective on material flows through 2050 is that building construction through 2030 (new construction, remodelling, modernization) will not change dramatically in quantitative terms. The construction activity “dismantling” (demolition and the output from reconstruction and modernization), on the other hand, is assumed to increase by some 50 % in 2030 compared with 2010. In the fourth decade, ending in 2050, a dramatic break in the trend is forecast. New

construction activity will have dropped by 40 to 50 % by 2050, compared with 2010. At the same time, more than twice as much “dismantling” as in 2010 will take place. The savings in primary raw materials (absolute, in millions of tonnes) due to the incorporation of recycled material from the construction industry into construction products for new construction and rehabilitation of buildings will depend on this general construction activity.

The calculated recycling utilization for 2010 in building construction, amounting to some 8 million t, is very modest in view of the calculated total input quantity of primary raw materials of approx. 120 million t.

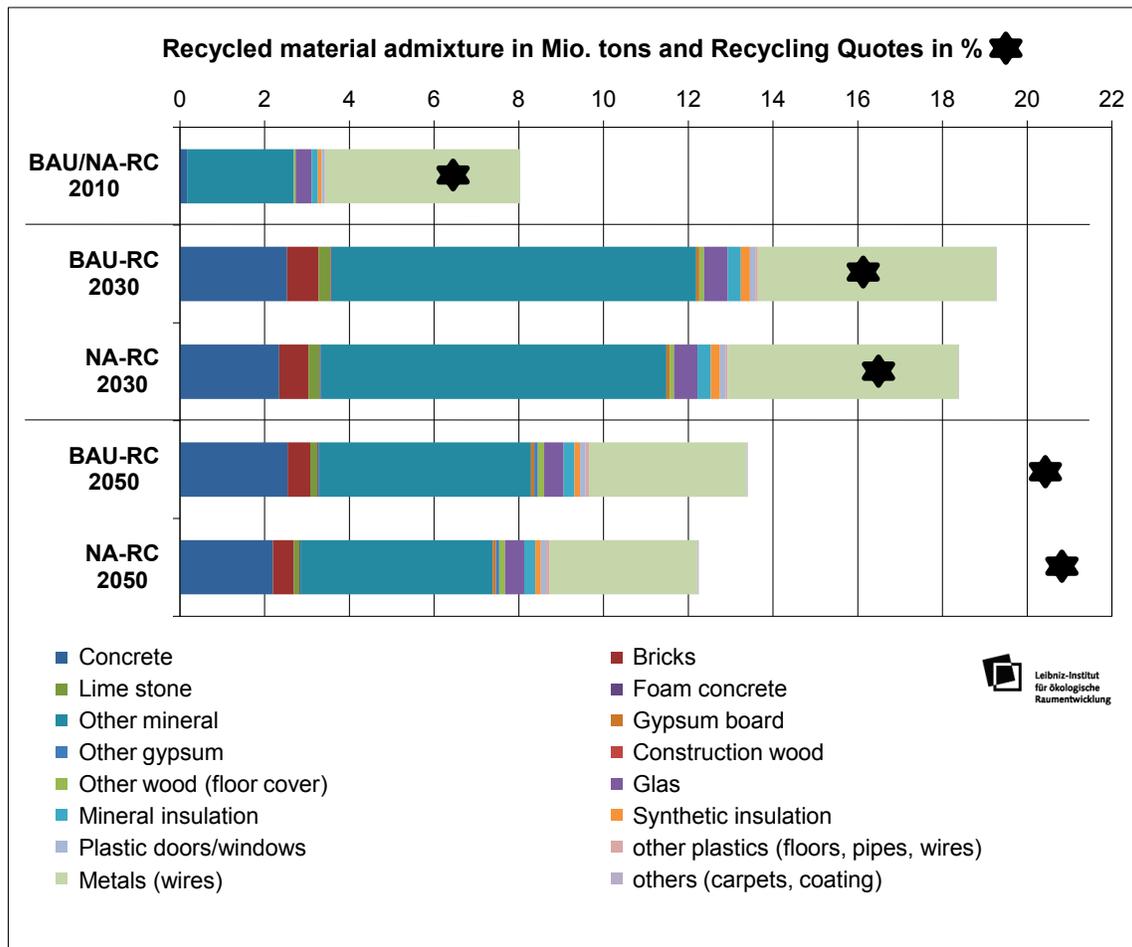


Figure 1 Sensitivity Study Secondary Raw Material Use – 2010-2030-2050 (Deilmann et al. 2014)

Following these Sensitivity Studies the use of the admixture of recycled materials could optimistically reach a quantitative peak around the middle of the third decade of between 18 and 20 Million t, and then drop to approx. 12 to 14 Million t by 2050, due to the drop in construction activity, in spite of the relatively high RC shares. In this context it was instructive to see the strong dependence on the overall building activity trends and the moderate contribution by even favourable RC conditions in the future.

From a technological point of view, it is clear that in order to achieve these levels of recycling in building construction, a certain quality, non-hazardous nature and separability of the various construction rubble materials must be provided (recyclability), and procedures are necessary with which the materials can be prepared suitably for reuse, and ultimately transferred into the market-suitable products. I. e., costs incurred must be in the range of existing equivalent alternatives (especially in the primary materials sector), or be covered by means of cost compensation mechanisms.

Acknowledgement:

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1.8 Session H: Recovery of Waste in and from Thermal and Metallurgical Processes





THE RECYCLING POTENTIAL OF METALS FROM MSW INCINERATION RESIDUES

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Introduction

Beginning with the book "The Limits to Growth" [1] started a change of thinking in industrial nations in the early '70s, in the sense, that necessary protection of the environment and natural resources became high ranking targets of modern societies, especially for young people. Today it is recognized by society, that the own civilization waste has to be recycled and treated among others in waste incineration plants [2] and can no longer be left as burden for future generations in a landfill.

The world is in a global structural change, which is particularly visible in the developments of commodity markets since the turn of the millennium with sharp price increases of metals since 2003, the global economic crisis in 2008/2009 and the continuous decrease in metal prices since 2011, due to lower needs in China. It is obvious that the traditional industrial nations [3] are no longer in the center of these developments. As in the latest past, in future the supply of raw materials, will be determined, often detached from actually existing reserves and resources, by dependencies which are caused by changing market structures on the supply and demand side in combination with oligopolistic and monopolistic market structures. Against this background, it is for resource-poor industrial nations of vital importance to strengthen the efficient use, recycling and substitution of raw materials, where appropriate, to develop corresponding processes or to optimize existing procedures.

From this point bottom ash is an interesting metal containing waste material. In Germany about 5,6 million tonnes of bottom ash is produced per year and about 400.000 t of iron scrap and about 37.000 t of non-ferrous scrap are separated in small, medium and large size pieces with state of the art technology today. Nevertheless about 107 kg per tonne of dry ash [2] is still in bottom ash after that treatment. Especially the fine fraction < 2mm contains higher amounts of metals. To answer the question, if these metals can be recycled economically, it is necessary to know in which condition they are.

In which structures do we have metals as function of grain size?

The fine and coarse fraction of bottom ash was examined [2]. After sieving, the bottom ash was classed into different particle fractions. There was no difference in concentration of aluminum and copper across the various particle sizes in contrast to lime, silicon dioxide and zinc. The zinc and lime contents are higher in smaller particle size fractions and silicon dioxide contents are higher in larger particle size fractions.

In the fine fraction iron is totally oxidised and exists mainly as magnetite (Fe_3O_4), very often grown together with other oxides. Regarding different possibilities of metal separation the existence of magnetite is very important, because magnetite (Fe_3O_4) does have a higher solubility [4] for other metals. Due to the spinel structure of magnetite, Fe^{2+} and Fe^{3+} ions can be replaced by Co^{2+} , Ni^{2+} , V^{3+} , Cr^{3+} ions, so magnetite acts as a collector for tramp elements of iron. In **Figure 1** is shown how tramp elements like Co, Ni and Cr can be separated from bottom ash with the separation of magnetite with magnetic separation systems.

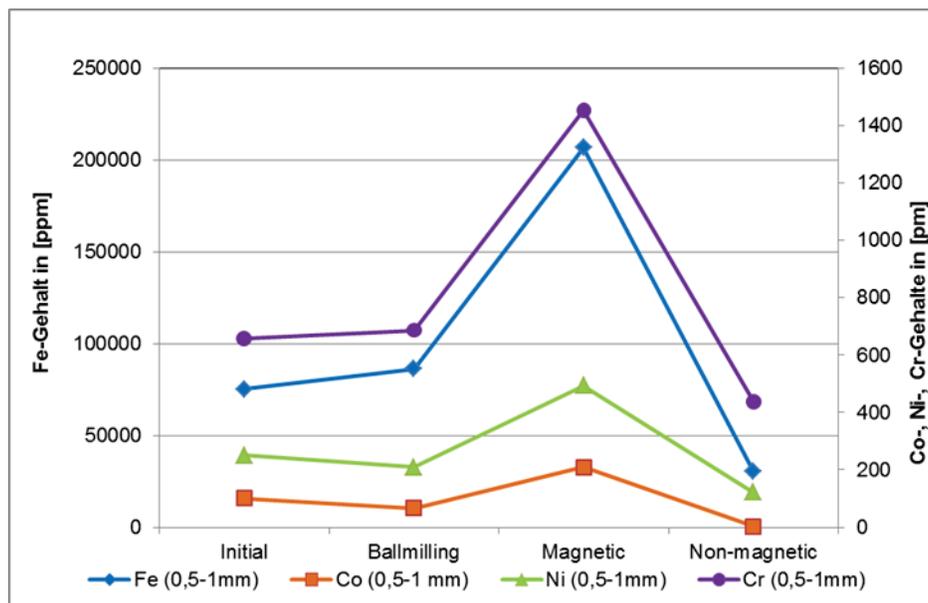
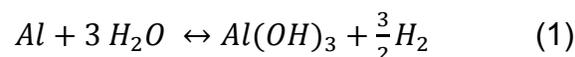


Figure 1 Reduction of tramp elements of iron in the non-magnetic mineral fraction with the magnetic separation of magnetite

With this method it was possible to reduce the Ni content to 121 ppm in the non magnetic mineral fraction. That is below the Ni content of natural rocks, such as Basalt (134 ppm) [5].

If bottom ash is used in road construction the aluminum content may cause problems because of formation of hydrogen due to the following reaction:



$$\Delta G (298 \text{ K}) = -427283 \frac{\text{J}}{\text{mol Al}} \quad (2)$$

$$\Delta H (298 \text{ K}) = -418630 \frac{\text{J}}{\text{mol Al}} \quad (3)$$

In [6] it is shown, when an aluminum lid, for example of a quark packaging is heated up to about 900°C under normal atmosphere, the aluminum became liquid (melting point temperature 660°C) but no formation of aluminum droplets could be observed. That no aluminum droplets were formed under these conditions is reasoned by the fact, that the aluminum is covered very fast with a small layer of aluminum oxide. Because of the high melting point temperature of aluminum oxide (2072°C) it

- is still solid,
- protects the liquid aluminum between the upper and lower layer from further reaction with oxygen
- prevents the formation of a liquid aluminum droplet

The aluminum oxide layer is too thin for normal detection with SEM and EDX. At Fraunhofer Institute of Silicate Research it was possible to measure the thickness of the aluminum oxide layer (**Fig. 2**) with X-ray photoelectron spectroscopy (XPS).

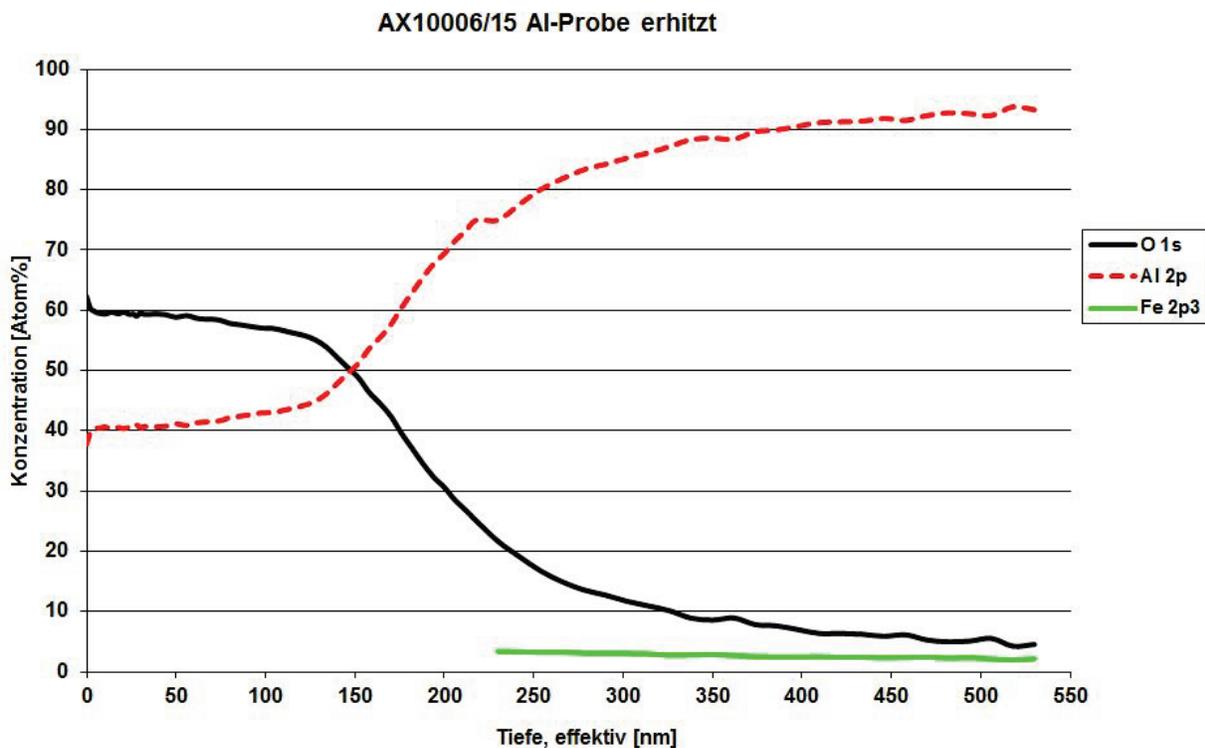


Figure 2 Thickness of aluminum oxide layer of 150 nm on an aluminum lid heated up to 900°C.

From **Fig. 2** it can be seen, that the aluminum oxide layer does have a thickness of 150 nm when the aluminum is heated up to 900°C. In the non-heated area a layer thickness of the aluminum oxide of about 15 nm was measured.

Future recycling potential of metals from bottom ash

Especially in the fine fraction of bottom ash [2], at a conservative estimate, copper is included in the order of 0.3-0.4%. Since today poor copper ores with copper in this order are degraded, bottom ash has to be considered in this respect as a valuable material, regardless metal price fluctuations. Moreover, according to [7], based on data of a waste incineration plant in Switzerland, it can be assumed that gold in the order of 1-2 ppm will be in the fine fraction, probably associated mainly with copper present. Thus, the gold grades in the fine fraction of the bottom ash comply with typical ore grades of geogenous mines [8]. Since the gold occurs predominantly associated with the copper, it could be recovered by the removal of the copper and subsequent copper smelting according to the state of the art technologies. Gold is collected in the copper electrolysis anode sludge from which it can be extracted with existing technologies.

Outlook

A high potential of metal recycling from bottom ash is available, but economical and simple recycling processes are needed, to separate the valuable metals from bottom ash. In Germany in addition it is needed, that the remaining mineral fraction can be used in other industrial areas as raw material. If the remaining mineral fraction has to be landfilled, the recovery of metals can most likely not be economical.

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RESOURCE EFFICIENT MANAGEMENT OF MSW INCINERATION FLY ASH – THE CASE OF VIENNA

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Introduction:

Pursuant to the commission decision (2000/532/EC) municipal solid waste (msw) incineration fly ash (19.01.13) and boiler dust (19.01.15) containing dangerous substances are classified as hazardous waste. In respect to the council directive (1999/31/EC) on the landfill of waste these materials have to be landfilled on a landfill for hazardous waste or on a landfill for non – hazardous waste if the matter is stable, non – reactive and has equivalent leaching behaviour to non-hazardous waste (e.g. solidified, vitrified). Criteria regarding the leaching behaviour are general described in Annex II of directive (1999/31/EC) and specified in detail in council decision (2003/33/EC), which contains requirements with respect to the eluate quality as well as proposals for the leaching test. The described legal framework is implemented into Austrian law (AWG, 2002), (DVO, 2008) and (AbfallverzeichnisVO, 2003). Additionally to the Austrian eluate requirements criteria on the total content of the elements As (5.000mg/kg), Hg (20 mg/kg) and Cd (5000 mg/kg) are given for non-hazardous waste landfills (Reststoffdeponie). In (Quina M., 2007) the maximum content of As detected in an APC residue was 960 mg/kg and 1660 mg/kg for Cd. For Hg a maximum amount of 51 mg/kg was given. Maximum values given in (IAWG, et al., 1997) are lower than the ones given in (Quina M., 2007). This means that according to the Austrian law beside eluate quality only total Hg contents in the fly ash have to be considered regarding landfilling at non-hazardous waste landfills.

Without any preliminary treatment (solidification/stabilization) the generated fly ashes (filter ash, boiler ash) do not fulfil criteria to be landfilled at non-hazardous waste landfills, neglecting fly ashes from fluidized bed incinerators. Reasons why a fly ash cannot be disposed without prior treatment are the content of total dissolved solids as well as the leachability of certain elements like Pb, Cd and Zn (Purgar A., 2013). Also the total content of Hg might be exceeding limit values for waste acceptance at non-hazardous waste landfills. In Vienna fly ashes that cannot be disposed of on a landfill for non-hazardous waste are stabilized with Portland cement prior to their disposal or are exported to be stored at hazardous waste landfills. The main aim of the ongoing project is to investigate alternative treatment methods to the stabilization process under consideration of the existing infrastructure in Vienna regarding waste incineration and residue disposal.

Methods:

In cooperation with the plant operators, detailed information about the waste incineration cluster of Vienna including the infrastructure for the disposal of the residues (cement stabilization facility and residue landfill) is acquired. Based on information about the operational concept of the incinerators together with data about the residue composition (total contents and leachability), determined by the obligate basic characterisation of waste, the basis for the decision whether a literature proposed technology is suitable or not is gained. Treatment methods for MSW fly ash have been investigated for decades and a variety of different concepts can be found in (Thome-Kozmienky, 2013) or (Astrup, 2008). Within this work it is investigated whether fly ashes can be used as secondary resource in the cement industry (Lederer, Rechberger, & Fellner, 2015) or for phosphorus based fertilizer production (Egle, Rechberger, & Zessner, 2014). Additionally the possibility of Zn recycling with the FLUREC process is investigated (Schlumberger & Bühler, 2013). Wet chemical treatments as well as thermal treatments, similar to the MR or 3R process, described in (IAWG, et al., 1997), are considered.

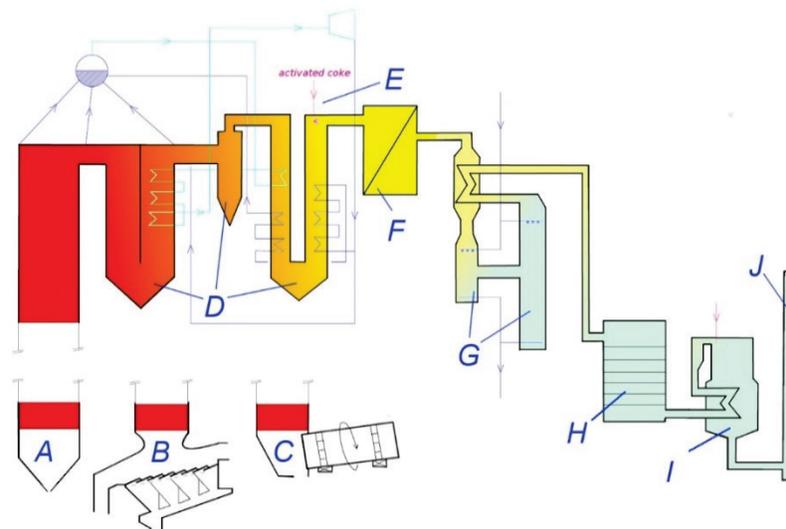


Figure 3 A) fluidized bed combustor, B) grate furnace, C) rotary kiln, D) heat recovery zone E) activated coke injection (optionally) F) electrostatic precipitator or fabric filter G) wet scrubbers H) fixed bed coke adsorption (optionally) I) selective catalytic reduction J) chimney

Results:

About 1 million tons of waste, derived from 13 combustion lines in Vienna, are incinerated annually, whereby 45,000 tons of fly ash are produced. As combustion technology rotary kilns, fluidized bed combustors or grate furnaces are used. All the incinerators are equipped with a heat recovery system followed by the flue gas cleaning system. The flue gas cleaning system consists out of electrostatic precipitators or fabric filters optionally with the injection of activated coke prior to the filtration. All the incinerators are additionally equipped with a wet flue gas cleaning system. Incinerators without activated coke injection use a fixed bed coke adsorber. As

a final step of the flue gas cleaning process a selective catalytic reduction system (SCR) is installed. A simplified scheme can be found in Figure 3.

Fly ashes investigated from fluidized bed combustors fulfil criteria for landfilling at a non-hazardous waste landfills. Only fly ashes resulting from mono combustion of sewage sludge may be used for the production of phosphorous based fertilizer. Neglecting heavy metal contents, other fly ashes investigated do not show phosphorus concentrations high enough to be potentially classified as fertilizer according to (DüngemittelVO, 2004). Untreated fly ashes from grate furnaces and rotary kilns exceed the limits for a disposal on a non-hazardous waste landfill. They can also not be used as substitute for marlstone in the cement industry, though the composition of the Matrix elements (Ca, Al, Fe) is similar. The main reasons are the increased chlorine and heavy metal contents (Lederer, Rechberger, & Fellner, 2015). Zn recycling, based on (Schlumberger & Bühler, 2013), is not economically reasonable due to low concentrations of Zn in the investigated fly ashes and the high alkalinity according to (Fellner, et al., 2014).

Considering that the investigated fly ashes meant to be disposed of at a non-hazardous waste landfill (“Reststoffdeponie” in Austria), a contamination or stabilization process is necessary. As a cement stabilization process is in operation currently, the focus is on thermal and wet chemical treatments or a combination. All fly ashes, except for those from fluidized bed combustors, show an increased content of total dissolved solids. A “pH-neutral” fly ash washing might reduce the content below the threshold parameter of 100 g/kg. Several fly ashes show increased leachability of Pb, Cd or Zn. Fly ash washing may improve these values not considering other elements. If the flue gas cleaning system of an incinerator is equipped with coke injection upstream the baghouse filter, the total content of mercury might exceed the threshold parameter of 20 mg/kg. Fly ash washing (acidic or neutral leaching) is not appropriate to reduce the Hg concentration decisively from fly ashes investigated (Purgar, et al., 2014), therefore an additional thermal treatment is envisaged and investigated.

Acknowledgements:

The presented work is part of a large-scale research initiative on anthropogenic resources (Christian Doppler Laboratory for Anthropogenic Resources). The financial support of this research initiative by the Federal Ministry of Science, Research and Economy and the National Foundation for Research, Technology and Development is gratefully acknowledged. Industry partners co-financing the research centre on anthropogenic resources are Altstoff Recycling Austria AG (ARA), Borealis group, voestalpine AG, Wien Energie GmbH, Wiener Kommunal - Umweltschutzprojektgesellschaft GmbH, and Wiener Linien GmbH & Co KG.

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MATERIAL FLOW ACCOUNTING AT PLANT LEVEL CASE STUDY: HEAVY METAL FLOWS IN BLAST FURNACE PROCESS

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Introduction

In a blast furnace process a huge turnover of materials takes place. The bulk of this material turnover is made up by Iron (Fe) and Carbon (C). However, the different input materials utilized carry to a minor extent also other substances, such as heavy metals (e.g. Zinc (Zn), Lead (Pb), Chromium (Cr), Nickel (Ni), Cadmium (Cd), Mercury (Hg)) into the blast furnace, which are subsequently transferred to different output flows. These so called micro-elements are either of interest for the quality of the product and by-product, due to their contents of Cr, Ni, for the process stability (e.g. Zn, Pb) or due to environmental reasons (e.g. Hg, Cd).

Material flow analysis (MFA) represents an appropriate tool to trace the flows of substances through industrial processes and to derive characteristic patterns for their behavior and distribution to certain output flows. The latter also named as transfer coefficients (TC) can (after their determination and validation) be used to substantially reduce sampling and measurements efforts usually necessary to adequately monitor substance flows through industrial processes. This has been successfully demonstrated by Morf and Brunner (1998), who applied the MFA for analyzing the feed composition of waste incineration plants by simply analyzing the residues arising from the process.

However, with respect to the application of MFA to the blast furnace process some challenges are to be tackled. These challenges include sampling and measurement methods as well as the variable behavior of some elements in the process (Trinkel et al., 2015a).

In the following the application of MFA for tracing Pb through a blast furnace and its satellite processes (top gas cleaning) is discussed. The heavy metal Pb has been chosen since it is relevant with respect to environmental and process control aspects.

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Materials and Methods

The investigated blast furnace process (including satellite processes necessary for top gas cleaning) with all its input and output flows is illustrated in Figure 1.

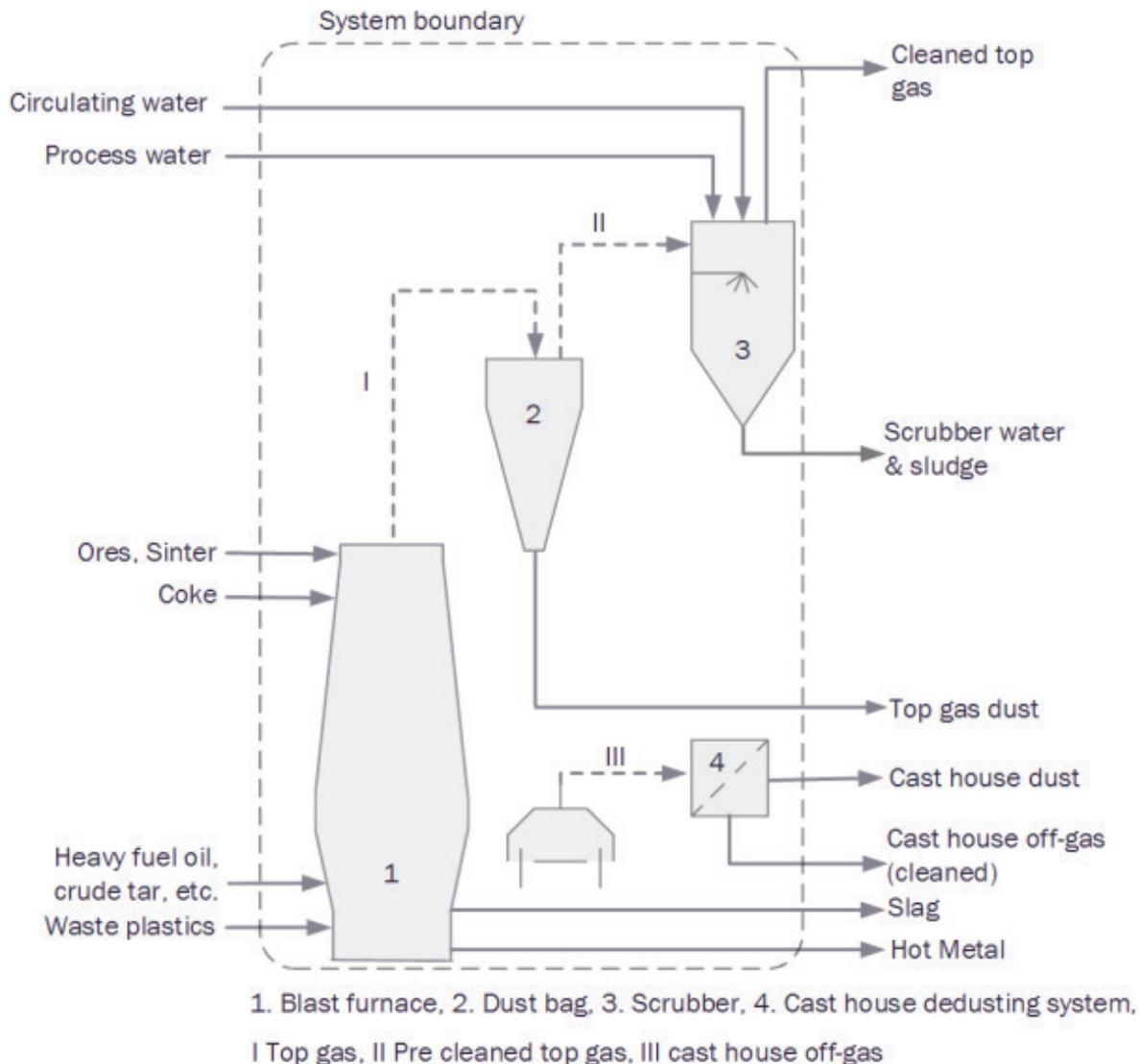


Figure 1: Blast furnace model (based on Trinkel et al. (2015b))

For the investigated plant annual mean values including mass/volume flow measurements for all inputs and outputs and concentration measurements for the respective heavy metal were available. These data were used to trace Pb flows by applying MFA.

In addition to routinely measured data by the plant operator, comparative measurements for the Pb content of the hot metal (HM) using X-ray fluorescence spectroscopy (XRF) and optical emission spectrometer (OES) were available.

Results

MFA was conducted for different heavy metals (Zn, Pb, Ni, Cr, Hg, Cd) for the years 2009 to 2012. In the following, the results for the Pb MFA are exemplary discussed. Based on the MFA conducted for the years 2009 to 2012 it became obvious that the differences in balances (total input versus total output of Pb) as well as the output of Pb via the HM have risen every year (see Figure 2). Whereas, in 2009 the difference in balance for Pb amounted to about 20 %, it doubled to almost 40 % in 2012 (output > input). During the same time the output of Pb via the HM indicated a similar development: in 2009 about 25 % of the total Pb output were discharged via the HM, this share increased to more than 35 % in 2012.

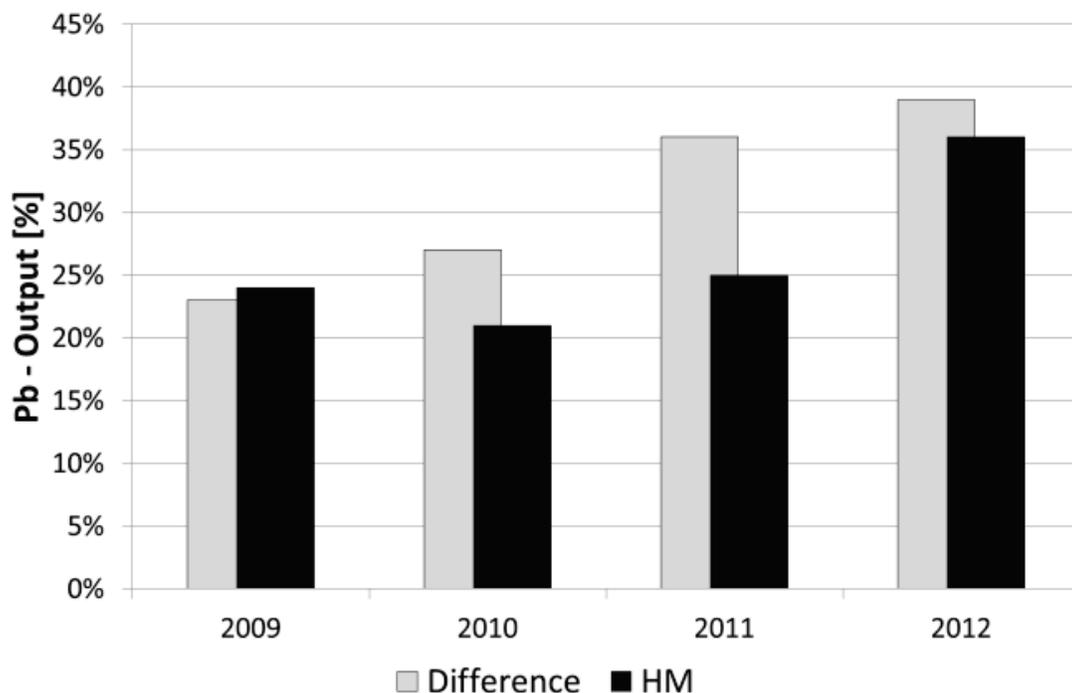


Figure 3: Differences in Pb balances and Pb output via HM for the years 2009 – 2012

The Pb discharge via the HM observed for the years 2009 to 2012 is in contradiction to the investigations conducted at the same plant by Morf (2007) in earlier times. His results indicate that only 10 % of the total lead input leaves the blast furnace process via the HM.

In order to clarify these differences, the output flow HM was investigated in more detail. Since the HM represents the sellable product of the process, its mass is well determined. Thus, the concentration measurement of Pb can be considered as the only potential source for errors, which could explain the observed differences.

The Pb content of the HM is measured by XRF. In a first step HM samples (which are characterized by a flat cylindrical shape) have been analyzed for their Pb content at both sample surfaces (the upper and lower base area of the cylindrical sample) using XRF. Thereby, it turned out that Pb is not equally distributed within the HM sample, since the analysis results for both

sides showed large differences (see Figure). The Pb concentration on one surface of the sample can even be twice as high as the concentration on the opposite sample side (e.g., Sample 1, Sample 2 and Sample 5 in Figure).

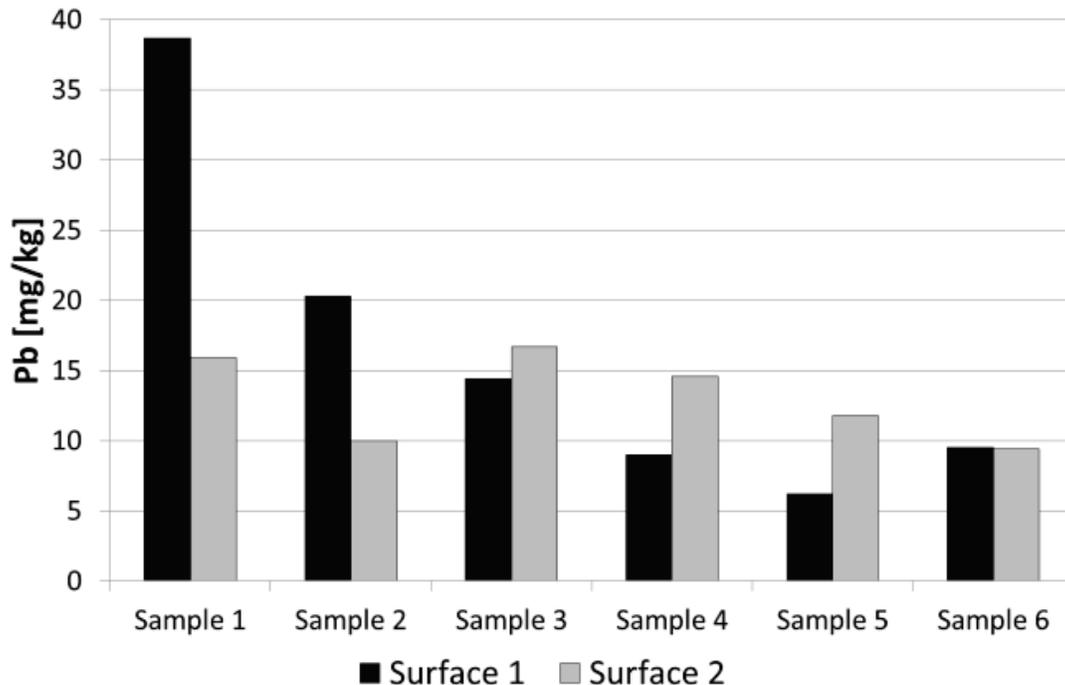


Figure 3: Pb concentration in HM samples measured at the upper and lower surface of the cylindrical HM samples

In a second step the reliability/accuracy of the measurement method itself was questioned. Pb occurs only in trace amounts in the HM and thus, the applicability of XRF is limited. Comparative measurements using OES demonstrate that the Pb concentrations determined by XRF most likely overestimate the Pb content of the HM (see Figure).

From these two findings it can be concluded that the Pb concentration is most likely overestimated and an improvement of the sampling and measurement technique would enhance the reliability of MFA for tracing Pb through the blast furnace process.

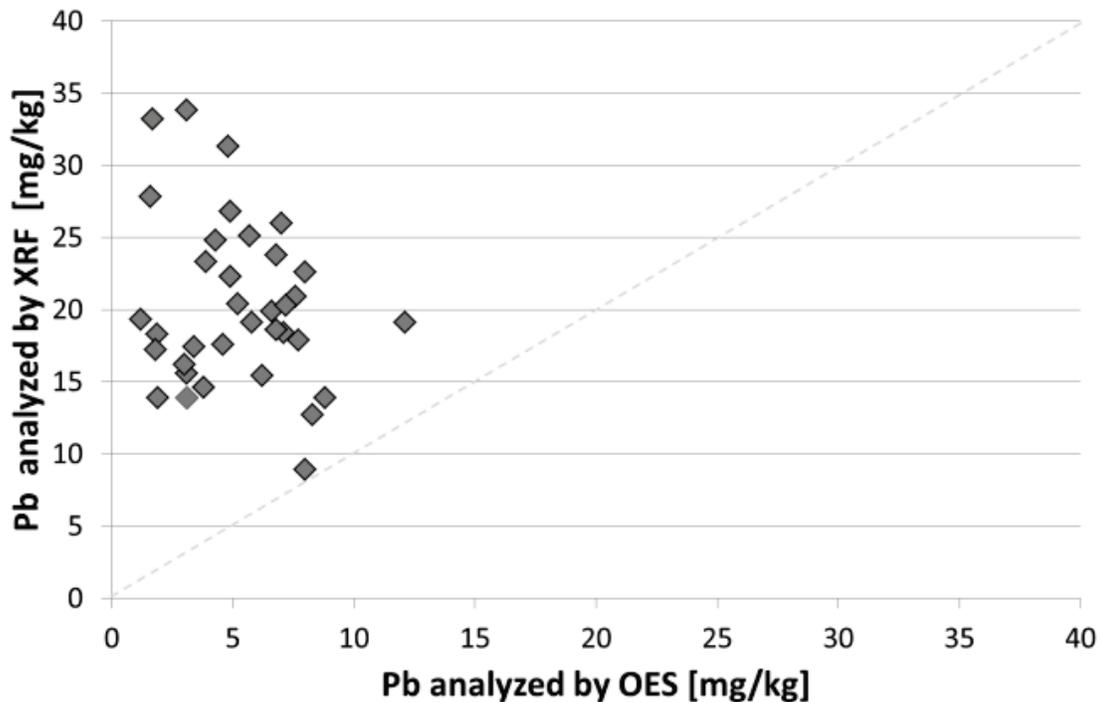


Figure 4: Comparative measurements of Pb in HM samples using XRF and OES

Conclusion and Outlook

MFA was conducted for different heavy metals for a blast furnace process. However, the application of MFA in this context faces different challenges, especially because heavy metals might be present at very small contents in different input and output materials. Moreover, input and output materials are sometimes extremely heterogeneous with respect to their composition, which makes representative sampling and subsequent analysis difficult.

Exemplary for all investigations the MFA results for Pb have been presented and discussed. Thereby, it was shown that the major challenge for this element with respect to enterprise level material flow accounting represents the analytical determination of the Pb content in the HM. Different analysis methods resulted in different Pb contents. Moreover, it was shown that Pb is also unequally distributed in the HM sample, questioning current practice of sampling and analysis.

In a future step, it should be investigated how to obtain a more homogenous HM sample for Pb and how to improve the analysis of the Pb content. Afterwards, it should be evaluated if these enhancements improve MFA results for Pb and allow determining constant TC for the process.



Acknowledgements

The present work is part of a large-scale research initiative on anthropogenic resources (Christian Doppler Laboratory for Anthropogenic Resources). The financial support of this research initiative by the Federal Ministry of Science, Research and Economy and the National Foundation for Research, Technology and Development is gratefully acknowledged. Industry partners co-financing the research center on anthropogenic resources are Altstoff Recycling Austria AG (ARA), Borealis group, voestalpine AG, Wien Energie GmbH, Wiener Kommunal-Umweltschutzprojektgesellschaft GmbH, and Wiener Linien GmbH & Co KG.

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CHALLENGES AND LIMITING FACTORS FOR THE RECYCLING OF STEEL SCRAP IN EUROPE

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Scrap availability – Trading and consumption of steel scrap

Steel scrap will be one of the future key raw materials in case of crude steel production. Due to the fact that the primary steelmaking involves CO₂-intensive processes CO₂ certificates has to be paid. One limiting factor for the expansion of secondary steel production (mini-mill route) is the global availability of steel scrap. J. Oda et al. investigates the future scrap availability (2013-2050) on the basis of historical trends in the past (1870-2013). The outcome of the study shows that:

- (1) in the past the consumption of old scrap was insufficient compared to the amount of discarded scrap, and
- (2) in the future the availability of scrap won't be sufficient to satisfy the steel demand. [1]

The regional material flow analysis to quantify past steel scrap usage points out a substantial difference between the estimated amount of discarded steel and the consumption of old scrap. These regional analysis based on investigations of Michaelis and Jackson [2], Davis et al. [3] for the UK, Fenton [4] for the US and Daigo et al. [5] for Japan. [1]

The global scrap trade in the year 2014 amounts to 92.8 million tons whereas the extra regional exports totted up to 54.5 million tons. NAFTA and the EU-27 are net exporters of steel scrap, which means they are exporting more steel scrap compared to the total imports. [6] The industrialized markets are the major net scrap exporters, where the scrap export of the EU-27 and NAFTA corresponds to more than 10% of their finished steel products (2013). Turkey has a strong dependency on scrap import because the scrap amount sums up to 79% of finished steel products. [6,7]

The report of the Bureau of International Recycling [8] shows short deviations compared to the values of the World Steel Association [6,7], whereas the trend of the EU-28 exports and the NAFTA is the same. Turkey is the most important buyer for both countries. In the year 2014 they imported 9.936 million tons from EU-28 and 3.616 million tons from the USA. [8]

Figure shows the crude steel production and the scrap consumption in Europe in the year 2014. The graph displays clearly that scrap is the major ferrous charge for the crude steel production in

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the European Union-27. The scrap ratio in percentage of the crude steel is nearly 54%. The EU-27 net scrap export of 13.4 million tons corresponds to 14.7% of the scrap consumption for the production of crude steel. [6,7]

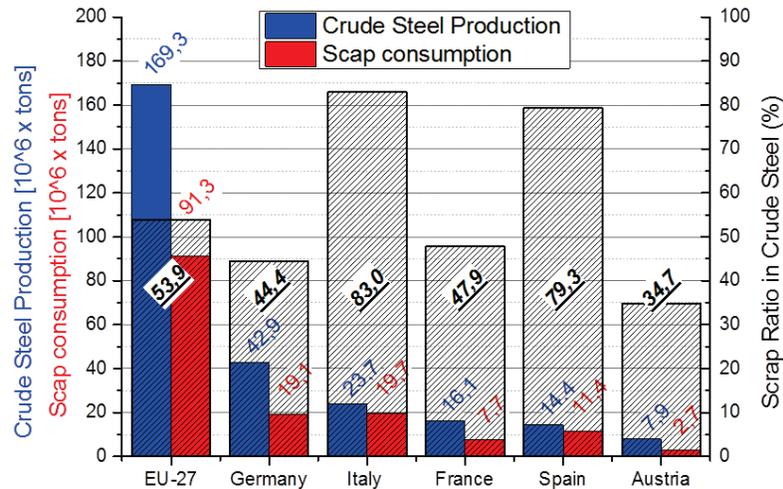


Figure 1 Comparison of the crude steel production and the scrap consumption in EU-28

Due to the limited availability of scrap in global terms, future R&D main interests should focus on innovative low-carbon technologies for primary steelmaking and investigate their economic feasibility. This aim should be to achieve the required reduction of the CO₂ emissions in the steel industry. [1]

Technological boundaries in the steel industry for charging scrap

Table displays the theoretical minimum values and the best achieved practical energy demands in case of the integrated and the mini-mill route. The main reasons for the higher practical values are:

- (1) the energy demand for production of the reducing agent from fossil fuels (coke, gas reforming),
- (2) the losses due to cooling and re-heating (e.g. coke, sinter, slabs) and
- (3) the energy efficiency for energy transfer and conversion. [9]

Table 1 Comparison of energy demands of routes per ton liquid steel [9]

	Integrated Route (Iron ore – Blast furnace - LD)		Mini-mill route (Scrap-EAF)	
	Theoretical minimum values (per ton of iron)	Reduction	6.7 GJ	Reduction
	Heating + Melting	1.3 GJ	Heating + Melting	1.3 GJ
	Σ	8.0 GJ (2222 kWh)	Σ	1.3 GJ (361 kWh)
Best practical values achieved “Top Performer” (per ton of iron)	17.0 GJ (4720 kWh)		4.6 GJ (1270 kWh)	

Figure illustrates the influence of the trace elements content on the steel quality. Further it is compared to the impact of scrap type. Thus not even the usage of own scrap is allowed for the production of extra deep draw quality (EDDQ-) steel due to a too high level of tramp elements. [9]

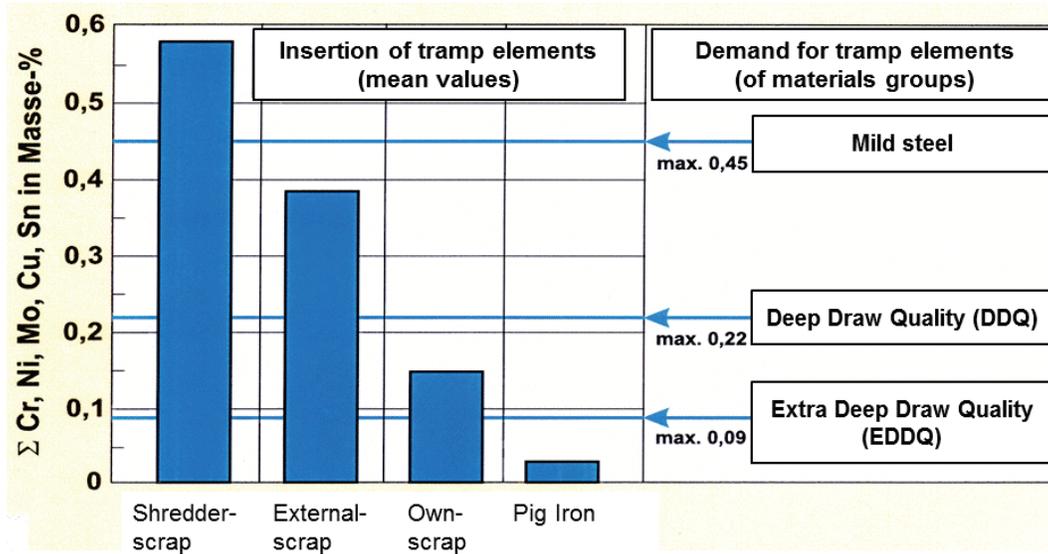


Figure 2 LC – steels: The influence of trace elements [9,10]

The ternary diagram in Figure compares the possible ratio of charging between hot metal, DRI/HBI and scrap with steel production technologies and steel qualities which allows different trace elements contents. The two steel grades which are extrema, are highlighted. On the one hand S235 where the production is possible with 100% heavy metal scrap (HMS#1). On the other hand the production of the Ultra-Low Carbon – interstitial free (ULC-IF) steel grades allows 0% input of HMS#1 owing to the amount of trace elements. [9]

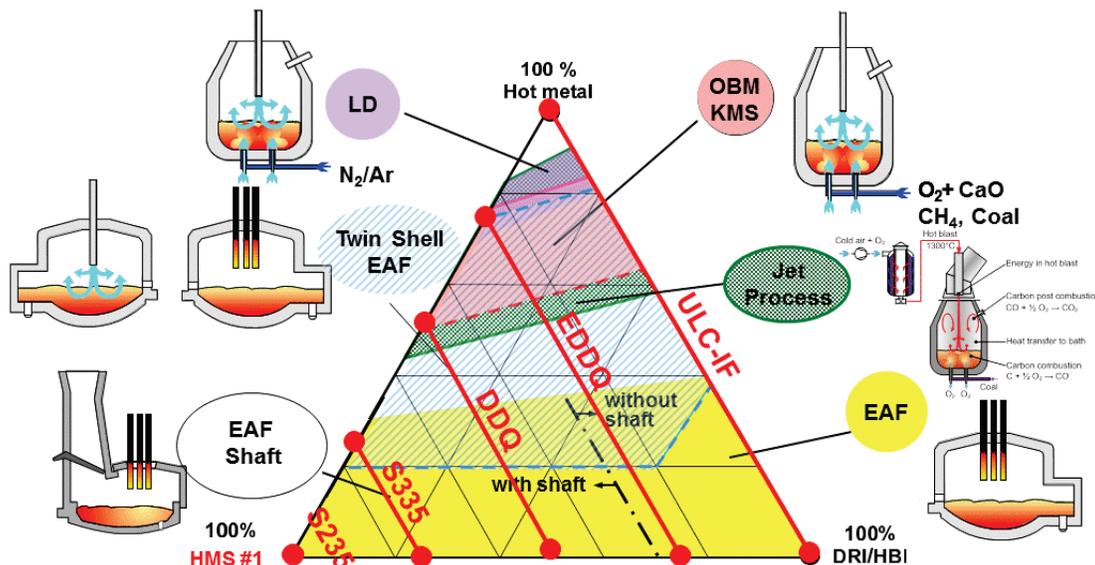


Figure 3 Steel production technology based on the charge mix [9]

Challenges for sustainable scrap utilization in Europe

One of the main future challenges will be a stronger cross linkage of the steel industry in the life cycle of steel. Therefore the steel industry should be a partner of customers in product design for improving the recyclability of the products. Furthermore it is necessary that an improvement of monitoring the qualities, detection and prediction of resulting steel grades for recycling takes place. [9]

The development of alternative processes for crude steel production and casting products makes it possible to decrease the percentage of scrap export. One of these new innovations is called Jet Process, which was developed to increase the charging rate of scrap and hot briquette iron HBI (Figure). This process uses the chemical energy of the coal injection to the maximum extent, which allows the rate of scrap and/or HBI charging to the converter up to 100%. Economical attractive market exists where scrap or HBI are relative cheap related to hot metal, with a hot metal bottleneck in virtue of a blow down of a blast furnace or limitations of hot metal production because of CO₂ emissions. [11]

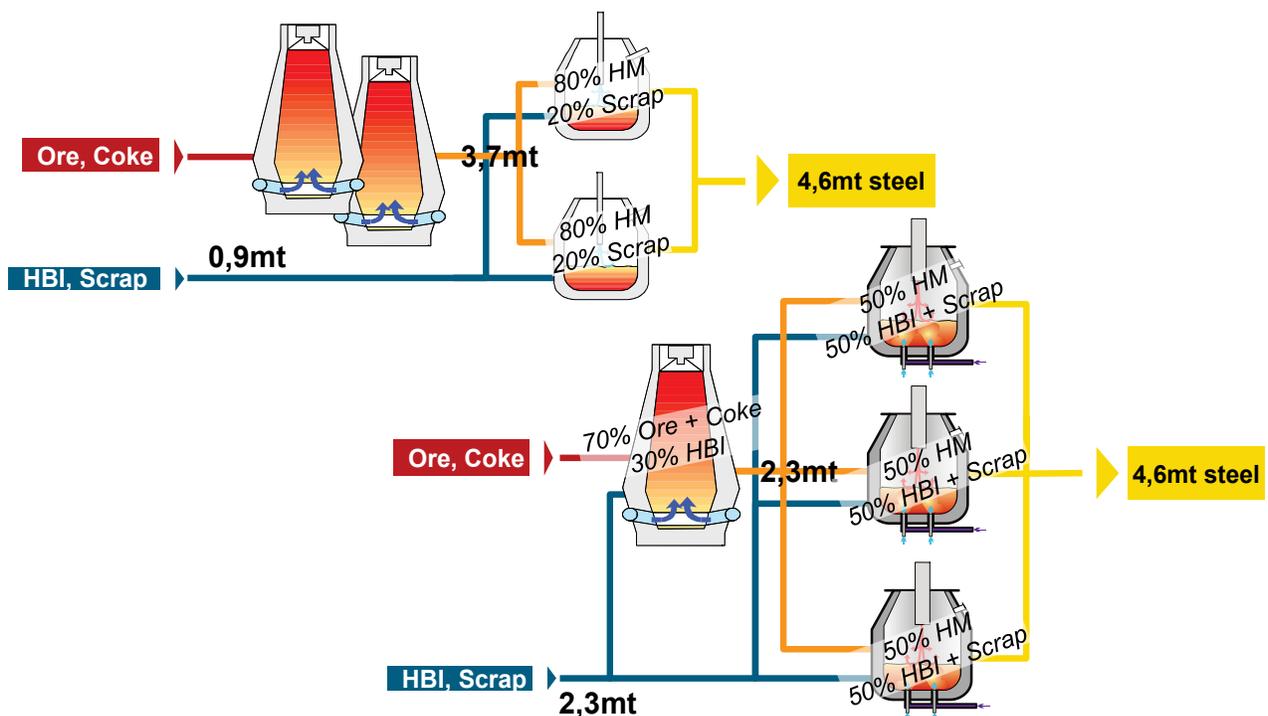


Figure 4 Modification of the process route for maximum scrap or HBI rate [11]

Furthermore the opportunity exists for the production of steel grades with higher tramp elements with new casting and rolling technologies (Figure).

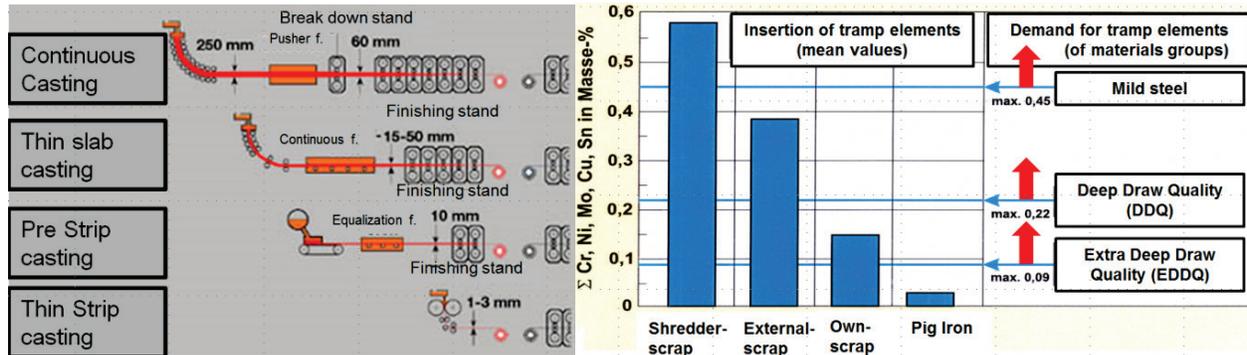


Figure 5 Changing the casting/rolling process to increase the possible amount of trace elements [9,10]

Conclusion

To sum up, the findings in this paper clearly display the driving forces and borders for scrap utilization at EU-28. Scrap is currently the main feedstock for the crude steel production (~54%). The lowest energy consumption and emissions could be achieved with maximum scrap selection. The percentage of trace elements limits the use for high and highest steel grades. The future challenges for the steel industry will be the protection of the scrap resources (i.e. raw material source), to hedge the scrap quality, to develop or install more flexible crude steel processes and casting processes that allow the production of high and highest steel grades with higher scrap input (i.e. more trace elements).

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2 Poster



ANALYZING THE INFORMATION BASIS: CHARACTERIZATION AND QUALITY EVALUATION OF RESOURCE BUDGET DATA

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Data availability and data quality as limiting factors of regional MFA

Material flow analysis reveals otherwise unknown information (Chen and Graedel 2012). Recent regional material flow analysis on a national level, i.e. resource budgets, (for example, for phosphorus (Zoboli et al. accepted), industry metals (Buchner et al. 2014) or critical metals (Habib et al. 2014), among others) contribute the necessary knowledge base for efficient resource management and policy. Though revealing new information, regional MFAs also depend on information in their production process. Missing data or data of poor quality are frequently observed as major limiting factors. Data gaps can often be bridged by balancing of closed systems (Binder et al. 2001; Cencic and Rechberger 2008; Yoshida et al. 2009; Laner et al. 2015). Still, it is not clear how to approach quality problems in *a priori* model input data. Input data of regional MFAs have varying formats, may diverge from the system context and come from heterogeneous sources, such as authorities, science, expert estimates, speculations or consumer behaviour studies. Most often, these are not frequentistic datasets but isolated values. Consequently, these cannot be approached by statistical methods. Having said that regional MFAs reveal new information, the initial basis of this new information is most often put second place. The quality of the input information may be relevant for subsequent decision making. It is thus beneficial to analyse, evaluate and communicate input data of regional MFAs. Consistent data evaluation is also important for analysis of data uncertainties (Laner et al. 2014) and is thus an essential component of MFAs (Rechberger et al. 2014). While techniques for modelling material systems and for considering uncertainties have been developed, adequate methods for data characterization and data quality evaluation in regional MFAs remain matter of research. An overview on a new approach designed for regional MFAs is provided in this conference contribution.

Data characterization and database analysis

To approach these shortcomings in MFA input data evaluation, a data characterization framework has been developed (Schwab et al. accepted). The core of the framework is a data characterization matrix for documenting, structuring and analyzing the database of a regional MFA (Figure 1).

Flow	Attribute 1	Attribute 2	Attribute 3	...
F1				
F1.1	<i>quantity</i>			
F1.2	<i>concentration</i>			
F1.2a	<i>concentration (ref. A)</i>			
F1.2b	<i>concentration (ref. B)</i>			
F2				
...				

Figure 1: Structure of the data characterization matrix and its two applications "database analysis" (columns) and "data quality evaluation" (rows).

The DCM is applied in the three-step procedure of MFA database analysis. That is, (1) a full information inventory of all input data of a resource budget, (2) the description of all information elements by data attributes and (3) attribute-wise (that is, column-wise) analysis of data attributes. Data attributes are data-associated annotations concerning statistical properties, meaning, origination and application of the data. The DCM and the database analysis procedure facilitate the representation of heterogeneous MFA databases as simple bar charts, the communication of extensive databases and the comparison of resource budget databases to another. For more information on the procedure, terminologies and a case study, please see Schwab et al. (accepted).

Data quality evaluation

Two general categories of data uncertainty can be distinguished (Morgan et al. 1992). First, this is aleatory uncertainty which is irreducible variability inherent in nature. Second, this is epistemic uncertainty which results from a lack of knowledge and is reducible by further investigation. Aleatory uncertainty appears to be more relevant when available data is rich, and epistemic uncertainty to be more relevant when available data is scarce. The relevance of the latter, epistemic uncertainty, for regional MFA has been outlined by different scholars (Gottschalk et al. (2010), Laner et al. (2014)). In the particular case of national resource budgets with spatially and temporally explicit system boundaries, it can be said that data quality is not an aleatory but an epistemic phenomenon. It can thus be defined as a *degree of belief* in given data to be true or not. Data quality is not a quantitative measure, but a qualitative perception of an agent, which makes data quality evaluation particularly intricate and barely replicable.

Epistemic data quality shortcomings can also be regarded as "information defects" (Dubois and Prade 2010). Information defects in the case of MFA appear for example when the data producer or the data collection method is unreliable, when data diverges from the context of a system, or when the meaning of data is not precisely known. The above outlined data characterization matrix can also be applied to approach information defects and consequently systematic, transparent and replicable data quality evaluation (Figure 1). Data quality can be

regarded as a function of various data attributes. A procedure for calculation of semi-quantitative data quality indicators (DQI) is proposed in Schwab et al. (2015) and illustrated in Figure 2.

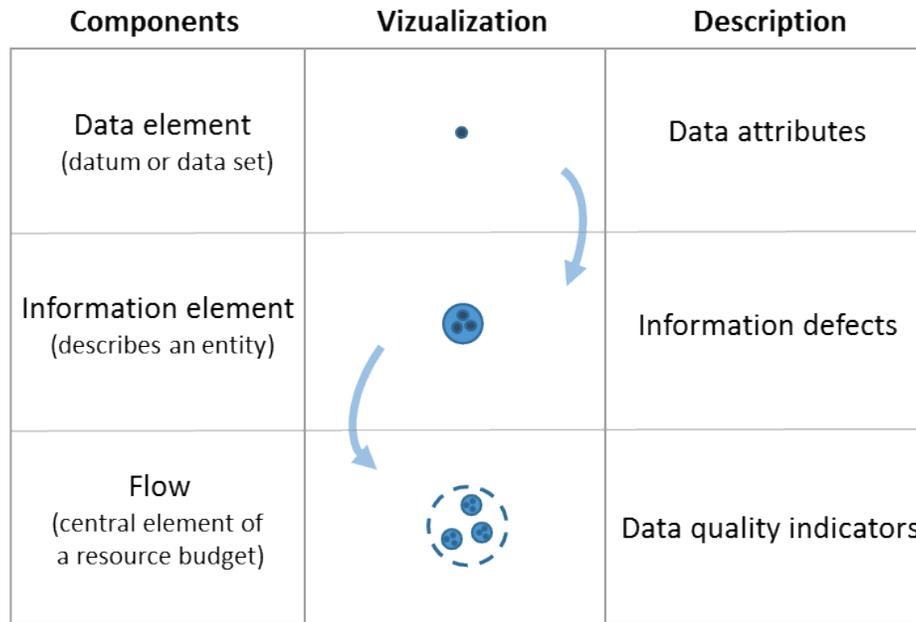


Figure 2: Scheme of the procedure for data quality evaluation.

As a result, each flow of a regional MFA gets assigned a unique number, a DQI, between 0 and 1. A DQI indicates whether data are non-defectuous (0) or do have severe defects (1).

Comparing the information content of regional MFAs

The introduced DQIs are a new quantity which can be applied in further meta-analyses of regional MFAs. First, they can be used to assign uncertainty ranges to material flows by application of uncertainty coefficients. More than that, they can be applied for measuring the complexity and information content of resource budgets. This can be useful for tracking the process of learning over time about a particular resource budget. Also, it enables comparing resource budgets of different regions or different materials to one another and can provide useful information for MFA-based decision-making in resource management and policy making.

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‘URBAN MINING CADASTRE’ BASED ON EUROPEAN SPATIAL DATA INFRASTRUCTURES

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Introduction

The basic idea of resource recycling from the anthropogenic stock is not new; a systematic and transferable approach is still missing. One of the biggest warehouses unlocked is the current building stock. Different projects are focusing on the calculation of resource inventories of cities (SCHEBEK et al.; IÖR 2014; REISINGER et al. 2015; Ergebnisbericht des Projektes Hochbauten als Wertstoffquelle, 2015 and other projects). For smaller regions so called bottom-up approaches, based on data from geographic information systems (GIS) are commonly used. The essential need for detailed building information is crucial. The initial setup for urban mining bottom-up approaches is to analyze single reference buildings (used materials, technical facilities, specific resource parameters) and later relate them to buildings from the “Cadastre” (e.g. the so called “ARK-Method” introduced by LICHTENSTEIGER 2006). As a result, this will provide a projection of the spatial distribution for the accumulated materials within a region. On top of the initial inventory, material flow analytics can forecast temporal and future availability of resources.

Beside the necessary detailed information about resources of specific representative buildings (individual perspective), the overview of building types, volume, building-use and age-classes for the complete area of interest is necessary (spatial perspective).

Status Quo – current results

Following the “GIS-approach”, the initial step within the research project PRRIG (TU Darmstadt, Germany) (SCHEBEK et al.) was to map current geospatial datasets within the region of interest to the target building typology. The specific building typology for non-residential buildings has been developed to meet individual characteristics in the Rhine-Main Region. The special characteristic of this typology is that it considers different already established typologies and merging them together to focus on urban mining. The typology is a hierarchical organized to allow different levels of abstraction. Currently the typology subdivide non-residential buildings into 12 main-categories (e.g. office and administration buildings, factory and warehouse buildings, commercial buildings, storage buildings) (see SCHNITZER & KÖHLER 2015 for more details). For residential buildings already established typologies like “Deutsche

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Gebäudetypologie” – IWU can be used and adopted (LOGA et al. 2011). The official German federal Cadastre “ALKIS” (German authoritative real estate information data) is used as official foundation dataset for the ‘Urban Mining Cadastre’. In the first step the official building function is mapped to the destination typology. To gather more detailed information about the building use, current land-use definitions from ALKIS can be merged with the building dataset. The combined dataset offers a first overview of all building types in the area but shows a lack of semantics within the authority dataset. Because of missing information about building ages in the official ALKIS datasets another dataset called “RegioMap” (REGIONALVERBAND FRANKFURTRHEINMAIN 2013) defining polygons of historical city development was identified and intersected to create a general orientation of the building-age. Unfortunately, no information about refurbishments and other recent buildings changes can be acquired using this datasets.

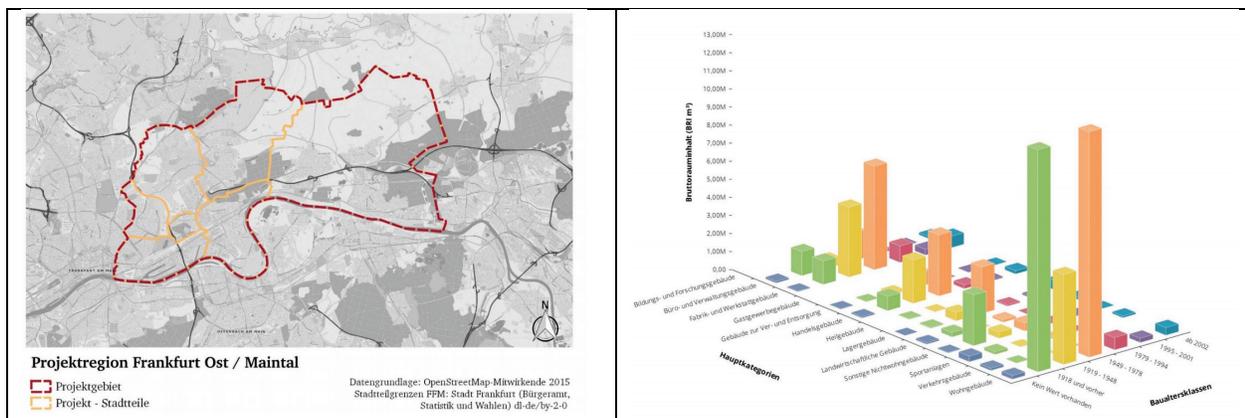


Figure 1 Research area and initial results of building information to establish the Urban Mining Cadastre (building types and age-classes in cross value)

Overall, these geospatial requirements are similar to wider geospatial data demands in other research areas (e.g. energy-demand calculations) and different regions across Europe. What can be stated here is that currently most projects in the field of urban mining (and correlation projects from the energy sector) develop specific building typologies, individual geospatial data models and collection methods. In all projects, due to the variety of required information’s (building use, renovations, building-age etc.) many different data sources are needed. Currently they are mostly merged together individually, because of unstructured and disharmonized data models. As well, mostly non-harmonized geospatial-data-models are used as destination model.

SDI and Urban Mining

Here is where Spatial Data Infrastructures (SDI) are playing a unique role in future geospatial strategies. Especially in Europa where a wide SDI called INSPIRE (Infrastructure for Spatial Information in the European Community) (INSPIRE - 2007/2/EG, 2007) is currently established. The overall goal of SDIs (in this case INSPIRE) can be described as: “A combination of technological and non-technological set-ups within and between organisations to facilitate access, use and sharing of spatial data thereby contributing to the performance of work processes” (VANDENBROUCKE et al. 2009). INSPIRE focuses on the harmonization of spatial data in 34 data themes, from addresses to sea regions in four main areas of interoperability: technology, geometric, semantics and legally. The INSPIRE 3D-data model for buildings (one of

the 34 themes) is based on the common OGC¹⁸-standard CityGML. CityGML is a widely used 3D-standard to establish a solid and adaptably model for a wide range of use-cases within the scope of building information (INSPIRE Thematic Working Group Buildings 2013). Following this approach different spatial data warehouses, combined in a harmonized INSPIRE based data model can be the key element for a secondary resource inventory Cadastre. Other than “2D Cadastre approaches”, 3D-building models can as well be used for automatic calculations of the cross-value (German BRI (DIN 277-1, 2015)) to transfer specific resource values from one building to another in more detail, keeping in focus that type and age classes are corresponding.

To demonstrate the use of the harmonized INSPIRE data model for buildings as a standardized and exchangeable foundation for urban mining the following steps need to be done. The already defined model (INSPIRE BU 3.0) needs to be analyzed and partially adjusted to meet all described criteria (geometric and semantics). Next, the current code-lists (e.g. current-use) needs to be extended to fit together with specific needs of urban mining considerations (extensibility = narrower).

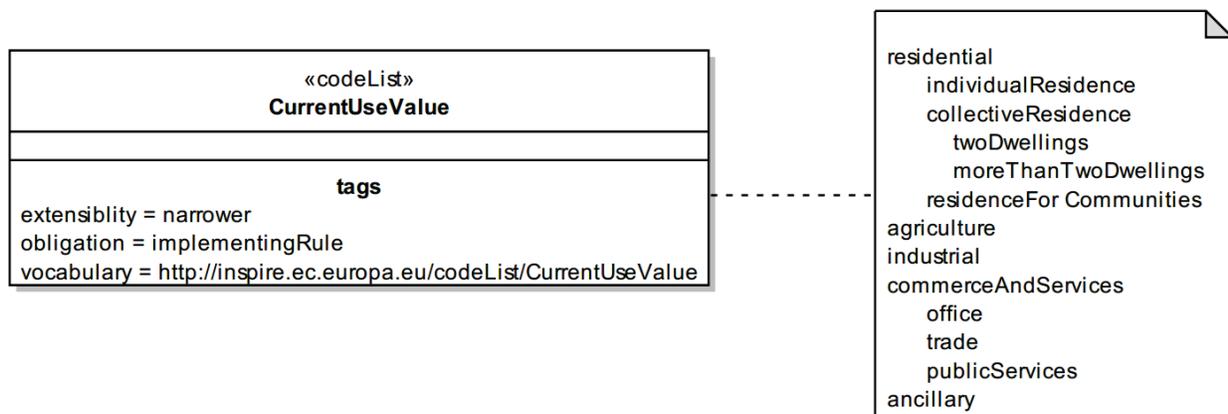


Figure 2 INSPIRE BU Codelist "CurrentUse" (UML Model - INSPIRE Thematic Working Group Buildings 2013) – the codelist is expandable to meet the urban mining criteria

Following this step current data models (schemas) can be mapped and converted to the ‘urban mining destination information model’.

Summary

From the overall geospatial perspective, the requirements can be divided into two main parts: the data modelling and data-collection. The data collection is data transformation (from current datasets) and gathering of new information’s (by hand, semiautomatic or via crowdsourcing). The destination model needs to be as complex as necessary and as easy as possible. One of the main problems to establish an ‘Urban Mining Cadastre’ is the knowledge of the buildings stock in total and specific resource values of reference buildings. With the harmonized data model, a common framework for the building stock within Europe could be initiated. It will not

¹⁸ Open Geospatial Consortium

directly solve the question of missing initial datasets but it can support building a sustainable model for data exchange between research project and similar approaches.

This poster will analyze and highlight how the INSPIRE data model for buildings can be used to set up such a transferable 'Urban Mining Cadastre'.

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POTENTIALS OF PHYTOMINING FROM WASTE INCINERATION BOTTOM ASH USING HYPERACCUMULATOR PLANTS

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Introduction

Facing a predicted shortage of resources (EC 2014) different strategies to deal with a decreasing availability of raw materials have been proposed, involving the use of so-called anthropogenic resources. Recently efforts are taken to evaluate the potentials and the availability of secondary raw materials from these resources (Allegrini et al. 2014, Lederer et al. 2014).

Waste incineration is a technology widely used in Western Europe, combining a reduction of waste volume with efficient energy recovery. Bottom ash represents around 15-30% of the waste mass incinerated. Incineration of waste leaves mainly bottom ash, as well as scrap metal and fly ash. On the one hand, metals in the bottom ash are present as scrap metal. On the other hand, metals are bound in the mineral form in the aggregates that are formed during the incineration. (Allegrini et al. 2014). Around 580.000 tons of waste incineration residues were produced in 2012 in Austria, with 455.000 tons of that material being landfilled (FMA 2014). In that way, a great resource potential is lost every year.

Phytomining could offer an environmentally sound and cheap technology to recover valuable metals from waste incineration residues. Phytomining is the use of plants to concentrate specific metals from a substrate in the plant tissue for the commercial use of the extracted metal. Successful application of this technology on soil has been shown by several research groups (Bani et al. 2015, Van der Ent 2013). However, the use of this green technology in combination with waste incineration residues is a novel and innovative approach.

The aim of our research work is to investigate the potential of phytomining from waste incineration bottom ash by growing metal accumulating and hyperaccumulating plants on these substrates and treat the metal enriched biomass to obtain a bio-ore. Waste incineration bottom ash was characterised and conditioned to obtain a suitable substrate for plant growth and tested in a greenhouse experiment.

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Materials and methods

As a first step, material from Vienna's municipal solid waste incineration, as well as residues from hazardous waste incineration were analysed. Samples were sieved to < 5 mm for further analyses. Pseudo-total metal contents were determined after aqua regia extraction. The plant-available metal fraction was determined after extraction with 1M NH₄NO₃ solution. These extracts were screened for trace elements using ICP-MS (Elan 9000 DRCE, Perkin Elmer). Further characterisation of the material included pH, electrical conductivity, total carbon and nitrogen content, TOC and cation exchange capacity. Different amendments, mainly from the waste industry, as well as different acidifying agents were tested for optimising the substrate for plant growth.

A pot experiment was set up in an experimental greenhouse in December 2014. The experimental setup consisted of a full factorial design involving five plant species (*Brassica napus*, *B. juncea*, two different clones of *Nicotiana tabacum*, *Sedum plumbizincicola*, a hyperaccumulator for Zn and Cd, and *Alyssum serpyllifolium*, a hyperaccumulator for Ni) with two different substrates and an unplanted control. The substrates consisted of a mixture of waste incineration bottom ash (70%), residues from mechanical biological treatment (MBT) of municipal solid waste (20%) and biochar (10%). Two different mixtures were used, including either bottom ash from hazardous waste incineration (HWI) or municipal solid waste incineration (MSWI).

Results

In general, the bottom ash can be characterised by a very high pH (up to 12.5), high salinity (2-8 mS cm⁻¹) and high heavy metal concentrations. Washing the bottom ash with diluted HNO₃ showed to be effective for lowering the pH of the material. The acid treated substrate in combination with material from MBT of municipal solid waste and biochar proved to be a promising substrate for plant growth.

Using this conditioned substrate, a pot experiment was carried out in the greenhouse. The two hyperaccumulator species *S. plumbizincicola* and *A. pintodasilvae* grew slowly and seemed to have problems to cope with the difficult substrate. Nevertheless, they showed elevated concentrations of Ni and Zn, respectively, in the above ground biomass. Half of the tobacco plants showed moderate growth, whereas the other half died off, independent of the substrate mixture and clone. *B. napus* and *B. juncea* were growing well and showed good potential to cope with the substrate.

Outlook and current investigations

With the knowledge gained from the first pot experiment it was clear that not every plant species is growing well on the waste incineration material. Thus, further substrate mixtures, involving aging for pH control, are under investigation. Moreover, a pot experiment using metal accumulating species in combination with microbial inoculants in order to improve plant growth and health was carried out and is currently under investigation.



Acknowledgements

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METHANOGENIC POTENTIAL OF MACROPHYTES AT BLACK SEA SHORE TO BE USED IN BIOGAS PRODUCTION

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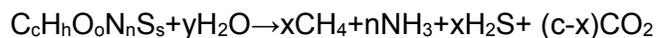
Introduction

The potential of algae for bioenergy is great since they are very fast growing organisms. Among other ways to produce energy from algae, the production of biogas is feasible but seems less attractive in comparison with obtaining biofuels of third generation [1]. In a town situated on the sea shore, the perspective of using naturally grown macrophytes in order to obtain biogas is a good point to be taken in view.

This work aims to establish the potential of macrophytes naturally grown at the the Black Sea shore to produce biogas, if integrated with the wastewater facilities of Constanta town, by using unexploited active sludge.

The anaerobic treatment of municipal sewage sludge

The anaerobic treatment of municipal sewage sludge is a complex process involving biochemical reactions, in four steps: hydrolysis of polysaccharides, fats and proteins, acidogenesis of converted products from first step, acetogenesis and methanogenesis. During this process, the biogas is formed, this containing methane, carbon dioxide, ammonia and other gases in smaller concentration. The stoichiometry of the global reaction is in connection with the elemental analysis of sewage sludge [2]:



where:

$$\begin{aligned} x &= \frac{4c + h - 2o - 3n - 2s}{8} \\ y &= \frac{4c + h - 2o + 3n + 3s}{4} \end{aligned} \quad (1)$$

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c, h, o, n, and s in Eqs. 1 represent the weight fractions of carbon, hydrogen, nitrogen and sulphur from elemental analysis of biomass dry matter.

On this basis, the methane production can be calculated not only from the sewage sludge but from any organic carbon source, such as biomass and more specifically algae, provided that anaerobic fermentation conditions are fulfilled.

The municipal wastewater treatment facility processes 28 millions m³ wastewater per year and produces 780,000 Nm³ biogas. Depending on the wastewater organic content, the characteristics of the primary sludge could vary. In average, the dry matter (d.m) content was 4.28%, with 59% volatiles (weight loss by heating up to 600 °C). During anaerobic fermentation, the d.m. content of sludge decreases to 2.66% and the volatiles content decreases to 49,8%. The biogas contains 6,7% methane, in average.

Two third of the sludge remain unprocessed due to the insufficient capacity of fermentors. This could be used to the anaerobic fermentation of algae in order to obtain supplementary methane.

Experimental

Bench scale experiment was performed with one batch reactor at 28 °C. 1L of activated sludge from an industrial fermentor was used, with a concentration of 2.06 g/L d.m and pH=8.1.

For reference, 1L of activated sludge produced 16 mL biogas, in 24 h. By comparison, a quantity of 7 g algae *Ulva lactuca* with the composition presented in Table 1, added to 1L sludge, produced 25 mL, in 24 h, so that 9 mL proceeded from algae.

Table 1 Chemical composition of *Ulva lactuca* used in biogas production

Component	Content, %
Humidity	15.45
Glucides in d.m. [3]	54.95
Proteins in d.m. [3]	14.58
Lipids in d.m [3]	0.69
Total Volatile Solids	70.22
Nitrogen [3]	2.32
Elemental composition, wt fraction	c=0.331, h=0.184, o=0.406, n=0.023, s=0.056

The experiment continued after adding 30 g algae and monitoring the gas production in a laboratory gasometer. The anaerobic fermentation took 18 days until gas volume stabilization. As one can see in Figure 1, in the first 13 days, the volume grew negative due to oxygen consumption, during the acidogenesis and the acetogenesis. After this, 350 mL biogas accumulated. In total, 359 mL biogas was obtained from 37 g algae (5.72 g d.m), in an anaerobic fermentation process using activated sludge as basis.

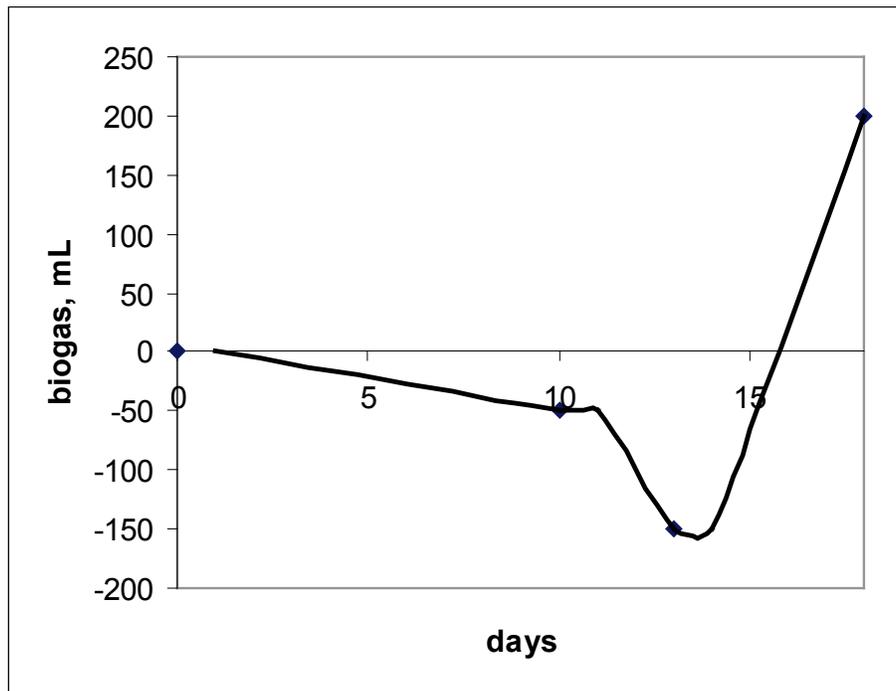


Figure 3 Time course of biogas production during anaerobic fermentation of *Ulva lactuca* with active sewage sludge

The concentration of methane in the biogas was 13.9% wt, comparing with 6.7% in the biogas produced from sewage sludge only.

Conclusions

The study demonstrated the potential of alga *Ulva lactuca* for the production of biogas, in the same process as in the anaerobic fermentation of sewage sludge. The algae fermentation is slow in these conditions and supplementary activation of sludge is needed in further studies along with optimization of process parameters.

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ENHANCED LANDFILL MINING AND ITS SOCIO-CULTURAL CONTEXT

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The debate about resource efficiency and scarcity of resources has emerged in the 18th century in Saxony because of deforested forests in the same region due to the smelting of ores (Carlowitz 2000 [1713]). Discussions continued with the report “Limits to Growth” from the Club of Rome (Meadows et al. 1972) and was taken up in the political agenda at the United Nations Conference on Environment and Development in Rio in 1992 (UN 1992). Since the early 2000s, the fear of the consequences of dwindling raw material deposits, rising prices, and supply security risks for the economy not only in Europe, but also in Germany have led to the development of Raw Materials Strategies, which is supported by research initiatives on innovative technologies for resource efficiency (BMW 2010; EU 2010; BMBF 2012; Werland 2012).

Before the European and the German raw-materials strategies were developed, the idea of using landfills as secondary sources for raw materials had emerged in Europe (Bockreis et al. 2011; Krook et al. 2012; Jones et al. 2013; Gäth/Nispel 2011). Landfill studies in the 1980s and 1990s focused on traditional landfill issues like lack of space or pollution prevention, as the first German landfill mining project “Burghof” had in 1993 (Krook et al. 2012; Krook/Bass 2013; Jones et al. 2013; Bernhardt et al. 2011; Bockreis et al. 2011). The concept of enhanced landfill mining (ELFM) draws attention to the exploitation of resources and the processing of energy from landfills, and aims to meet ecological and social criteria (Jones et al. 2013) like resource-efficiency, recovery of landfill space (Fricke 2013), reduction of aftercare period (Hölzle 2010; Fricke 2013), as well as reduction of CO₂ emissions (Jones et al. 2013). The goal of ELFM is to use the potential of resources that have already been mined and can be further used in new product cycles. Some authors claim that closing the cycle should be done not only during periods of resource shortages as is the current protocol, but should also be integrated as a general approach in resource policy (Jones et al. 2012).

Up until now, exogenous technologies like drills from mining have been used in pilot projects (Johansson et al. 2013; Hartman/Mutmansky 2002). Research in this area focuses on the development of technologies for dumping, operating, and the recycling of materials (Jones et al. 2013). A gap currently exists between the debates on the ELFM concept in scientific discourses, the development of innovative²¹ technologies, and the dealing with landfills in daily practices as final step to store waste materials (Krook et al. 2012). Understanding socio-technical system of

²¹ Innovations are symbolic or material artefacts which are perceived as something new as well as an improvement in comparison with the existing (Braun-Thürmann 2005).

waste management and resource use will enable the successful development of innovative technologies for resource efficiency. Technical innovations are always socially embedded; moreover, social-cultural factors influence the development of technologies (Sismondo 2010; Bijker 1995; Krook/Baas 2013).

Research Objective: In order to understand the concept of ELFM and its inert uptake in German waste management, I will focus on the socio-cultural context of this innovation. The question “How is the innovative practice of enhanced landfill mining designed and shaped by its socio-cultural context?” will be raised. The aim is to understand how the socio-cultural context influences and is influenced by the innovative practice of enhanced landfill mining as well as the identification of drivers and barriers. Drawing on the German case as a first step, the relevant actors as well as the interplay of interests, conflicts, and controversies linked to ELFM will be identified and analyzed in addition to its impact on the development and application of innovative methods and technologies. Methodically, the research relies on literature review and qualitative analysis of interviews and documents.

Preliminary Results and Hypotheses: The research on the issue has recently begun, and some of the preliminary results from expert interviews will be presented in the form of hypotheses. In the past, the waste management sector was characterized by innovative practices and innovative technologies. However, innovative practices and technologies were only introduced because of external dynamics, like modifications in the legal framework because of serious issues like piles of rubbish, polluted groundwater, and lack of space. This led to the assumption that the exogenous pressure on the existing socio-technical regime is too low to shape the political agenda in order to implement innovative practices and technological innovations in waste management like ELFM, and research on ELFM currently takes place in socio-technical niches (Geels 2007).

Mining landfills in its current state is characterized by uncertainties and lack of reliable information. The research projects, which are conducted on a laboratory scale, have shown that thus far, no reliable prediction can be made for landfills on the question, ‘what was dumped where at what time?’ (Bernhardt et al. 2011). Further research, however, could address this question and could make better predictions concerning its economic feasibility. The literature discusses economic feasibility on the one hand in the context of the quality of the output of the thermal treatment process, and on the other hand as a lack of resource potential (Winterstetter et al. 2015; Bernhardt et al. 2011; Danthurebandara 2015a and 2015b). Experts have predicted, that employment protection is a huge cost factor in Germany due to the hazardous working conditions like heat, gas evolution or hazardous waste.

Furthermore, local authorities and policy makers play an important role when it comes to the practices of granting approval concerning mining the landfill and providing funding for it. In addition to the argument of economic feasibility, the fear of possible contaminated land also hinders further research. Landfills in the EU as well as in Germany are the responsibility of public authorities. As of yet, German policy makers have not given the innovative practice of ‘mining landfills’ a priority in the political agenda because there are still easier ways to satisfy the growing hunger for raw materials.

Lastly, the “fear of the public” (Smolders et al. 2013) also hinders the implementation of enhanced landfill mining. This means experts expect that people living near landfills will reject the entire idea of ELFM. Experts explain this presumed public reaction by the NIMBY-Phenomena because mining a landfill could cause unpleasant odor, noise (Craps/Sips 2011), and changes to the familiar landscape. In contrast, the media response to the idea has been very positive, and landfills have been compared and framed as “gold mines” (Bäumer 2014; Hoferichter 2014). No valid prediction about the reaction of the public can be made because no visible, public opposition has been observed during any research project. Whether CO₂ emissions can be reduced is a controversial issue being discussed. Moreover, it depends on the certain landfill and its systems boundaries because landfill mining might also cause more emissions compared to “doing-nothing-scenarios” (Danthurebandara et al. 2015; Winterstetter et al. 2015).

There are potential drivers of ELFM. Framed as an environmental innovation, ELFM could be successful because it can return benefits to the society. First, mining raw materials from secondary sources affects the environment to a lesser extent than primary mining and promotes resource-efficiency. Secondly, landfill mining prevents future contamination of water and soil, and regrows current landfill space. This shows that despite the nuisance it may cause initially, enhanced landfill mining returns benefits for the landfill owner, environment, and society.

In conclusion, potential barriers to ELFM are the lack of exogenous pressure on the socio-technical regime, its economic feasibility, the fear of contaminated soils, and possible public protest. In contrast, the potential drivers could be forward-thinking local authorities and policy-makers, and a framing of ELFM as an environmental innovation.

Further research on the question, “How the practice of enhanced landfill mining is shaped by its socio-cultural context” will be done. More precise research will be done about the impact of conflicts and controversies. Based on this knowledge, governance options can be developed to cope with barriers, conflicting aims, and potentials.

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ANALYSIS OF COMPOSITION OF BOTTOM ASH FROM MUNICIPAL SOLID WASTE INCINERATION PLANTS

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Introduction

Incineration of municipal solid waste with simultaneous gaining of energy in waste to energy (WtE) plants plays a very important role in waste management system. Reduction of the volume and weight of incinerated waste and the possibility of electricity and heat production belong to the main advantages of the incineration process.

Solid residues remaining after the waste incineration are considered to be a special category of waste and it is therefore necessary to treat them in suitable way. Bottom ash presents the majority of all solid residues from incineration. In the Czech Republic the most of bottom ash is landfilled, but, in accordance with contemporary trends, its material reutilization should be preferred. Metal and glass recovery followed by the reuse of residual fraction as a construction material would be an appropriate option. In order to assess the material recovery potential of bottom ash, the knowledge of material composition and physical-chemical properties is needed.

Material and Methods

Bottom ash samples for the analysis were taken from two WtE plants in the Czech Republic. Bottom ash is a very heterogeneous material and its composition is strongly dependent on the composition of incinerated waste. For observation of the ash heterogeneity in a short time period, one day samples were collected in summer season 2014 (BAP1, BAP2, BAP3). The composition of the ash could also vary with the season and that's why integral samples were collected in different seasons (BAP4, BAL1 – summer season 2014, BAP5, BAL2 – winter season 2014). Samples BAP come from the plant with typically urban collecting area, while samples BAL come from the plant that is collecting waste from the city but also from surrounding countryside area. The influence of WtE plant location was observed through the comparison of the bottom ash composition from both WtE plants. The analytical methods were based on procedures described in literature (e.g. Chimenos et al., 1999).

In the first step the bottom ash samples were sieved and separated into eight size fractions, <2 mm, 2–4 mm, 4–6 mm, 6–8 mm, 8–10 mm, 10–15 mm, 15–20 mm a >20 mm. Each size

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fraction, except the fraction <2 mm, was manually sorted into the following material components: glass, ceramics and porcelain, magnetic fraction, ferrous scrap, nonferrous metals, organic material and residual fraction. The magnetic and residual fractions were further treated by grinding and sieving (0.5 mm sieve) for determination of all metallic and magnetic particles which could be trapped in the ash agglomerates.

Glass was sorted into primary glass which can be potentially recovered and into secondary glass contaminated with other materials and therefore is hardly recoverable. The content of metals in magnetic fraction was determined by digestion in aqua regia, followed by MP–AES analysis (atomic emission spectroscopy with microwave plasma). Nonferrous metals were sorted into coloured, silver low density and silver high density metals. Coloured metals, were separated visually. Silver metals were sorted in sodium poly-tungstate solution (density 2 820 kg/m³ at 20 °C) into the low and high density metals. Elemental composition of the low density metals was determined by smelting the metallic particles to the ingot, which was analysed by XRF analysis (X-ray fluorescence). Elemental composition of other nonferrous metals was refined also by using XRF analysis but only for typically occurring particles.

Because of the small particle sizes it is very difficult to recognize aluminium in size fraction 2–4 mm. That's why this fraction was tested for the metallic aluminium content by using the reaction of aluminium in alkaline solution, when hydrogen is generated. The amount of generated hydrogen is proportional to the amount of aluminium in the sample.

Results

Results of particle size distribution have shown that fractions <2 mm constitute the significant part of mass of bottom ash. Values range between 18 and 38 % depending on the sample. As shown in Table 1, amounts for one day samples are quite different, which highlights the heterogeneity of the samples, whereas results for integral samples are similar for both WtE plants. The wt. % of other fractions are decreasing with the growing particle size.

Table 4 Material composition of the bottom ash samples (wt. %)

Sample	BAP1	BAP2	BAP3	BAP4	BAP5	BAL1	BAL2
Glass	18	20	19	15	24	10	13
Ceramics and porcelain	2.7	3.1	3.3	1.8	3.5	2.3	2.7
Unburned org. matter	0.18	0.24	0.25	0.25	0.30	1.0	0.63
Magnetic fraction	15	24	21	20	16	19	18
<i>Share of Ferrous scrap</i>	0.83	8.9	5.5	3.1	4.5	2.0	1.1
Nonferrous metals	1.9	1.9	1.8	1.7	2.0	1.4	1.3
Particles < 2 mm	22	18	30	34	33	36	38
Residual fraction	40	33	24	27	21	31	27

Table 1 summarizes the results of material analysis of bottom ash. As it can be seen glass is one of the major components. Its content is between 10 and 24 % and varies with the sample. Results indicate influence of ash particle size on the material composition of bottom ash. Amount of glass increases with the increasing particle size of the ash fraction as shown in Table 2 and

Table 3. In size fraction above 20 mm its content is significantly lower which is probably caused by fragility of the glassy materials. From 50 to 70 % of the glass in each size fraction could be potentially recovered. Present technologies for the glass fragments recovery are limited by particle size (only pieces > 7 mm) (Makari, 2014). Glass particles above 6 mm constitute approx. 70 % of all the glass in the analysed samples.

The content of ceramics and porcelain varies between 2.7 and 3.5 % and is similar for all the samples. Most of the ceramics was found in size fractions over 15 mm. Ceramics is mostly present in a form of residues of construction materials (tough and not easy to break).

Unburnt organic matter was constituted by pieces of paper, synthetic or cotton textile and fibres, unburnt pieces of wood or plastics, etc. Its presence was higher in bigger size fractions (see Table 2 and Table 3).

Magnetic fraction, included ferrous scrap, ranged from 15 up to 24 %. Separated ferrous scrap formed between 0.8 and 9 % of all bottom ash weight. The content of magnetic fraction decreases as the size fractions increases. On the contrary, the amount of ferrous scrap increases in this direction and presents the highest value in >20 mm fraction (see Table 2 and Table 3). The ferrous scrap is more interesting for the recovery. Magnetic fraction, ferrous scrap excluded, has rather mineral than metallic character. The results of analyses have shown that magnetic fraction contains approx. 15–20 % of iron, around 4 % of Al, almost 10 % of Ca and other metals like Cu, Zn, Mn in minor concentrations. Its magnetic properties are caused by iron species and very likely by iron oxides.

Nonferrous metals were quite equally distributed in all size fractions and even total content in all samples is comparable (see Table 1 – Table 3) and ranged between 1.3 and 1.9 %. The lower content of nonferrous metals in BAL samples compared to BAP samples was one of the observed differences. This could be caused by different characters of collecting areas of the WtE plants ('Materials and methods' section). Another difference was observed in composition of metals. Majority of non-ferrous metals was formed by Al and its alloys, which were determined as the low density metals. XRF analysis showed presence of other metals like 0.6 % of Mn, 0.1–0.6 % of Cu, 0.5–0.8 % of Fe, 0.2 % of Cr, 0.1 % of Zn, or 2–4 % of Si in this fraction. These are mostly alloying elements of aluminium or could be impurities obtained during the incineration process. The coloured metals are mostly copper and its alloys. The fraction of high density metals was formed by stainless steel and alloys of zinc and copper. Also alloys of aluminium with higher content of Si, Cu and Zn, Mg were determined. Nonferrous metals in all BAP samples had approximately the following distribution: 80 % of Al, 10 % of coloured metals, 10 % of high density metals. Different distribution was found at BAL samples: 60 % of Al, 30 % of coloured metals and 10 % of high density metals. This differences could be also caused by WtE plants location characteristics and particularly by different social aspects.

The residual fraction was calculated as the rest to the 100 % and as shown in Table1 its amount varies between 21 and 40 %.

Table 5 Influence of particle size on the material composition of sample BAP4 (wt. %)

Fraction (mm)	2–4	4–6	6–8	8–10	10–15	15–20	> 20
Glass	17	22	33	34	34	28	3.1
Ceramics and porcelain	0.22	0.15	0.55	1.5	3.6	10	8.9
Unburned org. matter	0.20	0.25	0.37	0.33	0.42	0.39	0.86
Magnetic fraction	45	26	23	19	23	24	36
<i>Share of Ferrous scrap</i>	0.77	1.3	2.3	2.8	6.2	4.4	17
Nonferrous metals	1.9	2.9	2.9	3.2	3.2	1.6	2.8
Residual fraction	35	50	41	42	36	36	48

Table 6 Influence of particle size on the material composition of sample BAL1 (wt. %)

Fraction (mm)	2–4	4–6	6–8	8–10	10–15	15–20	> 20
Glass	8.7	13	22	19	24	20	8.6
Ceramics and porcelain	n.f.	0.53	1.3	2.9	5.2	14	12
Unburned org. matter	0.10	2.6	1.8	2.2	1.7	1.2	3.3
Magnetic fraction	48	25	17	16	19	20	39
<i>Share of Ferrous scrap</i>	0.34	1.2	1.4	2.1	3.0	3.6	17
Nonferrous metals	1.6	2.9	2.2	2.1	3.0	2.3	1.2
Residual fraction	42	56	56	58	47	42	36

n.f. – not found

Conclusions

The results have shown that the material composition varies with the bottom ash particle size. The bottom ash samples contained 10–24 % of glass, 2–3 % ceramics and porcelain, 0.2–1.0 % of unburnt organic matter, 1.3–2.0 % of nonferrous scrap and 15–24 % of magnetic fraction included ferrous scrap. The sorted ferrous scrap formed 1–9 % of all bottom ash weight. The results suggest that bottom ash from both incinerators can be potentially reutilized further. Development of proper treatment methods from technical and also economical point of view should be the following step.

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DEVELOPMENT OF A MODEL FOR DETERMINING THE COMPOSITION OF A RESIDENTIAL BUILDING STOCK

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1. Background

Due to increasing worldwide consumption of finite natural resources, the recycling economy and resource efficiency are gaining importance. Very large amounts of resources can be found in the building and infrastructure stock. During demolition the resources come up as demolition waste. An increase in the recycling rate of demolition waste is needed, since there is already a lack of landfill volume in parts of Germany which will grow over the next couple of years [Reppold, 2015]. To enable the resources to lead back into the closed-loop, they must fulfil specific requirements. The type of building demolition is, among other things, substantial for the quality of secondary resources. Only a well-planned selective building dismantling allows an unmixed collection of material and its best possible utilisation. A dismantling concept is created to outline the details of the dismantling. A part of this concept is the exact description of the building stock and planning the disposal method.

2. Deconstruction project of a former military property

Several former military properties in Germany must be converted into civilian use. This study deals with a sizable deconstruction project of a former military property, which consists of roughly 2000 buildings on a covering area of 385 ha. The building stock includes infrastructure facilities, schools, churches, administrations buildings, as well as 1380 housing units in 680 residential buildings, which can be divided in 20 different building types from the 1950s and 60s. Within the conversion, there are no after use plans like living or settling industry or business. Therefore, the deconstruction of the whole building stock is intended.

To avoid increasing costs, detailed preparatory work must be done prior to the tender. This can be a dismantling concept with the exact description of the building stock, including expected material flows and specifications for destruction and disposal methods.

Due to the large number of buildings, and consequently a large amount of demolition waste, a resource efficient deconstruction is required to regain extensive quantities of recyclable material.

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A selective building dismantling can be cost efficient because of the large number of buildings of the same type and the associated unique planning and repeating dismantling works.

The aim of the project was to describe the material composition of the housing development as precisely as possible to create a basis for the further deconstruction planning. Since the building stock is divided into 20 building types, it is sufficient to determine the material composition of each house type only once to calculate the material composition of the whole residential building stock.

The inventory construction plans do not cover all building types and they hardly contain any references to the used building materials. Therefore, the blueprints do not suffice to analyse the building with respect to the material composition. A determination of all missing information by on-site exploration would be very time-consuming and cost-intensive.

A model provides the opportunity to make an estimate of the materials and masses of the building stock. All on-site buildings can be referred to the same building period. The differentiating characteristic of the various building types is the particular floor plan, while used materials, as well as type and dimensioning of building components are comparable over the various houses. This makes the site well-suited for an analysis using models.

3. Development of a model

The material compositions of two exemplary selected building types, covering a large range of different characteristics in the building stock, were determined to generate material's masses and reference values that the presented model is based on. The model uses specific ratio values, calculated for every building material's mass on the base of the different reference values, which were then compared and evaluated.

3.1 Determining material compositions

The two selected buildings were chosen for their different floor plans, which are the primary differentiating characteristic of the houses in the site. Because of the large difference, they are expected to feature a varying material composition. This data was fed into a model, which was then used to calculate approximations for material masses in the other residential buildings.

The target materials were determined on the first on-site inspection. They provide a sufficiently precise characterisation of the buildings. The method used was a combination of evaluation of construction plans and on-site explorations. According to the approach of [Kleemann et al., 2014] the volumes of the bulk materials, such as concrete, timber and bricks, were calculated based on the construction plans. Their material properties and those of materials used to a lesser extent, such as metals and furniture wood, were determined by on-site sampling and examination. The result of one of the analyses is presented in the following table with the material groups already summarised.

Table 1 Material composition of a building

		mass	share
total mass		457,7 Mg	100 %
minerality	brickwork (incl. mortar, plaster and paint)	157,0 Mg	443,7 Mg 96,9 %
	concrete (excl. steel)	253,4 Mg	
	roofing	8,3 Mg	
	cement screed	19,0 Mg	
	asphalt screed	4,6 Mg	
	natural stone tile	1,4 Mg	
wood	construction timber (roof truss, staircase)	2.748 kg	5,4 Mg 1,2 %
	interior construction (furniture, doors, ...)	1.646 kg	
	wooden floor construction (parquet)	1.010 kg	
metal	Fe-metals: radiator + pipes	3.082 kg	5,2 Mg 1,1 %
	Fe-metals: reinforcement	1.996 kg	
	other	101 kg	
other	synthetic floor construction (carpet, PVC)	1.573 kg	3,5 Mg 0,8 %
	windows	1.477 kg	
	other	445 kg	

3.2 Development and comparison of specific values

To validate the assumption of a comparable material composition in the entire residential building stock, it was assessed whether the construction methods and materials of the entire housing development are comparable. Detected differences, which were not characteristic for the building type, but only affected a particular building, lead to an adjusted recalculation in order to create a common data basis and ensure the comparability of the specific values.

There are specific ratio values for each building type, calculated by dividing the mass of every considered material by every reference value, where a causal relationship is assumed. These values differ from building to building. If several buildings have similar specific ratio values, it is probable that more buildings of the housing development can be estimated well with this measure. The used method requires the material composition of the whole building stock to be similar, but the different building types to differ on their geometry or their floorplan, respectively. This has shown to apply to the sites residential building stock. Therefore, it is possible to calculate estimates of the material masses with specific ratio values.

The specific ratio values of both buildings are never identical, so the model uses the mean of both. The mean ratio value with the lowest statistical dispersion was chosen as the primary characteristic. This dispersion value is therefore also a measure for the uncertainty of the results.

4. Results

As a result, the material composition and masses of the residential building stock in the former military property could be approximated. For example, while less than 600 kg of PVC floor tiles can be found in a single building, in the entire residential building stock this mass adds up to about almost 400 Mg. The residential building stock comprises a total of a quarter million Mg. Due to the knowledge about the expected construction materials and their corresponding masses, this result constitutes a basis for the upcoming dismantling concept including possible disposal methods.

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CADMIUM INDUSTRIAL METABOLISM IN JAPAN FROM A SUSTAINABILITY PERSPECTIVE

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Introduction

Cadmium metal (Cd) naturally presents with zinc and lead in sulfide ores (Nordberg et al., 2015). Despite its properties as harmful pollutant, Cd has been used for various purposes including in batteries, pigments, alloys, and solar panels.

Japan has an unpleasant history with Cd. The Cd production in Japan increased before the World War II, caused the release of Cd in significant quantity to the river and water bodies. People in Toyama prefecture ate Cd-contaminated rice, and made them exhibited symptoms (weak and brittle bones) which was not immediately recognized as Cd poisoning. In 1968, the Ministry issued statement about the cadmium-poisoning as 'itai-itai' disease, and it became a starting point for Japan to start managing the Cd emission flows and initiating emission reduction.

Since then, several environmental regulations has been issued to reduce the emission of hazardous pollutants. Previous studies mentioned that Cd production in Japan has been reduced due to the decrease of average zinc intensity for galvanized sheet and technological development (Matsuno et al., 2012). Nevertheless, due to the high past usage, Cd is currently being stocked in the form of various products in the Japanese society, and in the future eventually the Cd will be either carried out to the recycling systems or discarded to the lithosphere. Thus, it will be very important to know how big the present stocks compared to the past as well as the current flows of Cd that will be potentially emitted in the future based on its present stocks in the society.

Methodology

The Cd flows and stocks data until in Japan are collected from secondary data sources, including reports (e.g. Japan Oil, Gas, and Metals National Corporation (JOGMEC)). Historical Cd emission and its current flows and stocks in Japanese society are analyzed by using substance flow analysis (SFA). The data was obtained until the year 2010 and the SFA result is presented in the form of flow diagrams. The present stocks of Cd, in the form of products, will be compared to their stocks in the past, in order to show how big the reduction of Cd. Furthermore, to make the result more easily understandable and has wider significance, 'sustainable Cd utilization' indicator is developed by using PICABUE method (Mitchell et al., 1995)

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Result

The SFA data is based on 2010 report from JOGMEC. From the result, the export of Cd in Japan is higher than its import in smelter and fabrication processes. On the fabrication box (Figure 1), the usage of Cd is divided into three categories: battery (Ni-Cd), alloy, and other usage. The 2010 in-use stock of Cd (3956 tonnes) is also estimated by performing simple stock and flow modelling of Cd from 2001 to 2010. Compared with study by Matsuno et al. (2012) which estimate the Cd stock in 1993 and 2008, the 2010 in-use stock found in this study is smaller. Therefore, there is a decreasing trend of Cd stock accumulation in Japanese society, which is consistent with previous finding.

In the SFA result, it is assumed that the landfill system in Japan is already safe. In other words, there is no emissions are leaked within one year. The air and water emissions that goes to the environment in waste management are from the incineration process (Ono, 2013).

In the future, Matsuno et al. (2012) mentioned that, due to its toxicity and regulations, the domestic Cd demand will decrease. It means, the in-use stock will keep decreasing. However, due to potential increase of global demand of steel and zinc (e.g. for building, automobile), Cd export demand will increase. When it happens, we identify two critical flows in the SFA diagram, marked by (a) and (b). The (a) flow is a material extraction from lithosphere to smelter, while (b) flow is Cd emission from smelter. When Cd export demand increase in the future, it is very important to pay more attention at flow (a) and (b), to make sure that the extraction is done in a sustainable way that does not cause the risk of Cd scarcity or resource depletion, and make sure the Cd emission from smelter does not exceed the environmental allowable limit. The other flows, such as flow to the use phase is not critical since the domestic Cd demand in Japan is decreasing.

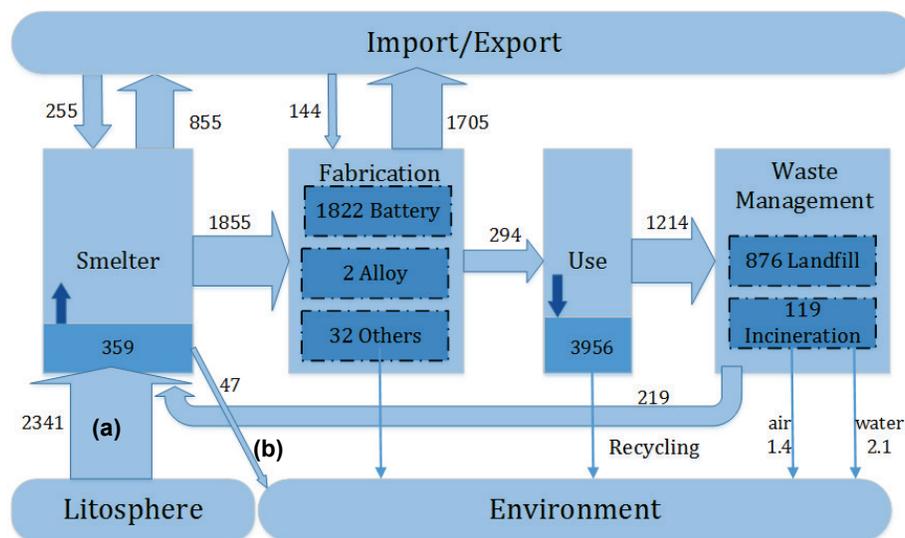


Figure 4 Cd substance flow analysis (SFA) in Japan in 2010

Furthermore, by utilizing the data from reports mentioned above, and by performing simple stock and flow modeling, the present and past usage of Cd can be compared.

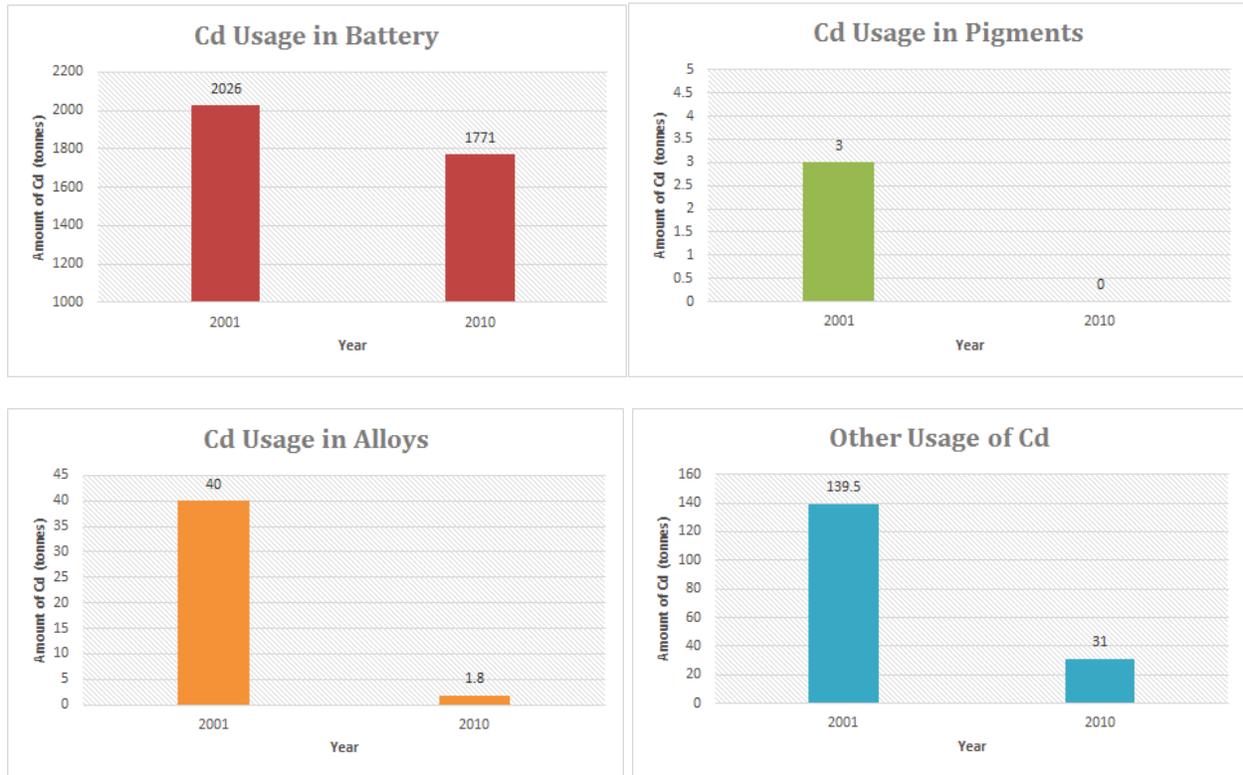


Figure 2 Japan Cd Usage Comparison in different products in 2001 and 2010

Figure 2 shows the Cd usage comparison of battery, alloys, pigment, and others in the year 2001 and 2010. It is clear that Cd usage in alloys, pigment and others have decreased significantly. Even in 2010, the usage of pigment was negligible or nearly zero. The Cd usage in battery is also decreasing, even though the decreasing rate is smaller due to the demand of Ni-Cd batteries.

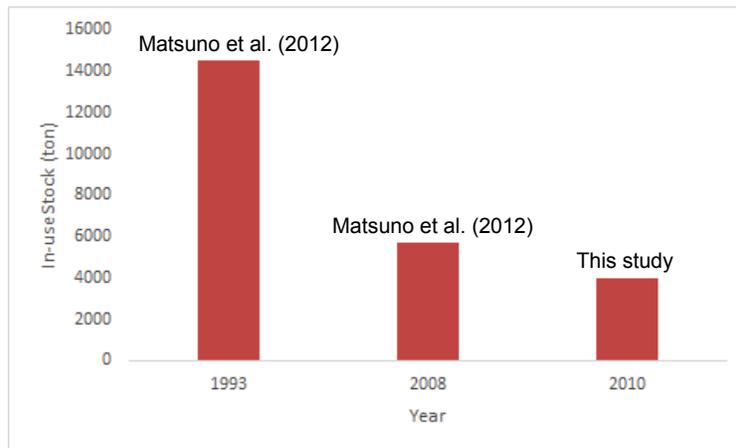


Figure 2 Comparison of Cd in-use stock comparison in Japanese society found in this study and Matsuno et al. (2012) study

This paper also develops an indicator to assess the sustainability of Cd utilization rate in Japan by using PICABUE method. Henckens et al. (2014) define the term 'sustainable extraction rate' for materials as the amount of extracted materials which will not cause the depletion of resource within 1000 years after 2050, at 9 billion global population. This global 'sustainable extraction rate' for Cd, which becomes an 'augment' or 'sustainable limit', in the sustainability indicator in this paper is 4.33 g/capita.year (Table 1).

Table 1 Japan sustainable Cd utilization indicators

Indicator	Amount	Unit
Global Cd sustainable extraction rate (Henckens, 2014)	4.33	g/capita.year
Japan Cd domestic utilization rate	2.31	g/capita.year
Japan sustainable Cd utilization indicator	0.53	-

Since large portion of Cd from smelter and fabrication processes are exported, it is difficult to estimate the total number of people utilize the Cd extracted from Japan. Nevertheless, when only domestic Cd utilization is taken into account (only the Cd that goes to the use phase), the Japanese people only utilize 2.31 g Cd/capita.year. When this value is divided with the 'augment', the indicator result is 0.53. It means the 2010 Cd domestic utilization rate in Japan is still within the sustainability limit, or in other words it will not cause Cd resource depletion within 1000 years.

Summary

The Cd in-use stock in Japan is continuously decreasing. In 2010, the in-use stock of Cd in Japanese society is under 4000 tonnes, and it is expected that the stock will decrease even more in the future. However, there is a concern about future increase in Cd export demand. When it happens, there are two critical flows that need to be taken into more consideration: the flow from lithosphere to smelter which related to Cd resource depletion, and the flow of Cd emission from smelter to the environment. Meanwhile, the present domestic utilization rate of Cd in Japan is still within a sustainability limit, which show the success of Japanese government effort to reduce the Cd consumption.

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STATUS QUO AND FUTURE DEVELOPMENT OF ECOLOGICAL FOOTPRINT, RESOURCE CONSUMPTION AND DIRECT CO₂ EMISSIONS OF THE VIENNESE PUBLIC TRANSPORT SYSTEM

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Introduction

Public transportation had a share of 39% of the Viennese modal split in the year 2012 (Wiener Stadtwerke, 2013). In order to sustain this high level of public transportation on the modal split, the Wiener Linien, carrier of the Viennese public transport system, need to expand their service, as Vienna is expected to increase in population by 300.000 to about 2 Million residents in 2035. To assess the sustainability of expanding a public transport system, we developed three scenarios and compared the effects on the Ecological Footprint and the direct CO₂-emissions of the Wiener Linien, including the effect of the source of electricity on these indicators. Additionally, resource consumption to implement each of these scenarios was calculated.

We used real inventory and energy-consumption data directly from the Wiener Linien. These data were supplemented with production and consumption emissions as well as life-cycle inventory data from the ecoinvent-database (Ecoinvent Centre, 2007).

Cases studied

The scenarios cover three development-strategies for the public transport layout of Vienna. The traffic-calculations, including number of vehicles, transport capacity vehicle-kilometres and others, were done by the Austrian Institute for Spatial Planning (Deußner, 2007).

The first scenario does not include large expansions. Current projects are finished, and then service expansion is halted. This scenario would lead to a drop in modal split, due to increased population without then adequate transport capacities. The resulting increase in motorised individual vehicle-traffic (MIV) would slow down the trams and busses, creating a self-sustaining negative feedback-loop.

In the second scenario, a major subway line is added to the existing system. Accompanied by adaptations in tram service and a slight reduction on the busses this scenario is the logical

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continuation of the public transport strategy of the City of Vienna. Although building a new subway line through the heart of the city poses a technical and financial challenge, this scenario is to be implemented in reality.

The third scenario massively expands tram-service, resulting in a traffic system of equivalent capacity as in scenario 2. This scenario would require substantial political back-up, as the required tram-lines would further compete with the motorized individual traffic for space on the surface. Additionally, tram-lines would be built through areas with high building density, causing conflict with the local communities.

Results

The **Ecological Footprint** of the Wiener Linien in the year 2012 amounts to 71.000 ha, Figure 1 shows the contribution to the Ecological Footprint for each scenario. Per seat-km the Ecological Footprint amounts to 0.04 m²/a. The specific Footprint of the modes “subway” and “tram” are about the same 0.04 m²/a, while the mode “bus” has a specific Footprint of about 0.08 m²/a. With regard to passenger-km the Ecological Footprint is 0.8 m²/a. (subway 0.9 m²/a; tram 0.5 m²/a; bus 0.9 m²/a).

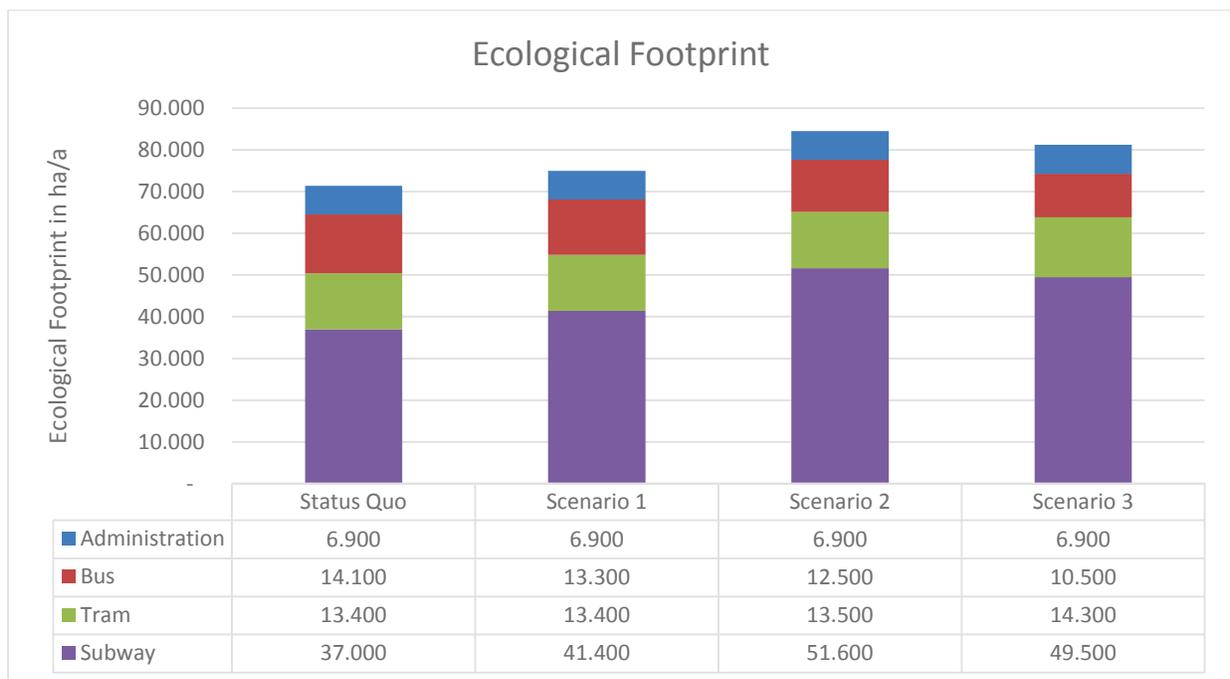


Figure 1 Ecological Footprint of the Wiener Linien – Scenarios and sectors

Depending on the development scenario until 2025 the **consumption of resources** varies considerably, from 1.1 to 4.6 million tons of material. The biggest demand (1-4,3 Mt) consists of concrete, followed by steel (65-250 kt) and aluminium (2.2-6.4 kt). The specific resource consumption for the scenarios – normalized for duration of use phase and seat-km is shown in Figure 2. Scenario 2 consumes most resources, as would be expected. The difference for plastics is not as large, due to subway trains being larger, thus using less material per

passenger. This measure for resource consumption does not distinguish, whether resources could be recovered or not. The concrete and steel built in subway tunnels is most likely not being recovered, whereas tram infrastructure can be reused (cf. Lederer et al. 2014)

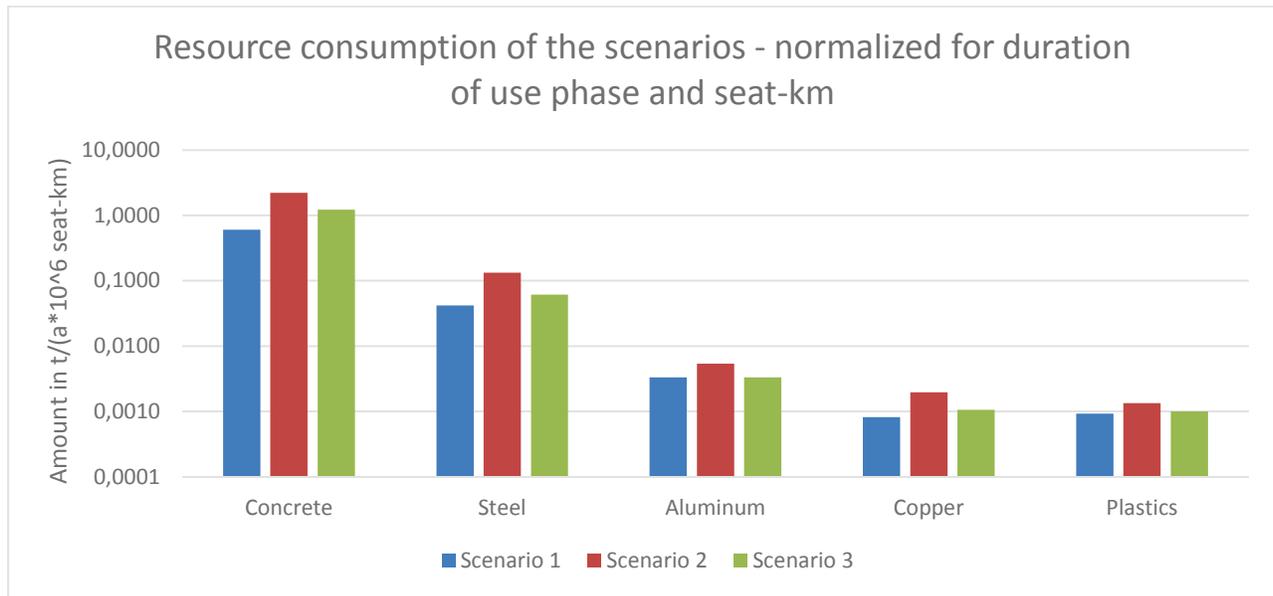


Figure 2 Resource consumption of the scenarios - normalized for duration of use phase and seat-km

For **direct CO₂-Emissions**, Scenario 3 yields the best results. This is mainly due to the expanded tram service substituting busses, which are fossil fuel driven. That effect holds true for the two more ecologically friendly sources of electricity – namely Wien Energie and certified green-energy – see Figure 3. If electricity with average UTCE is used, Scenario 1 pulls slightly ahead.

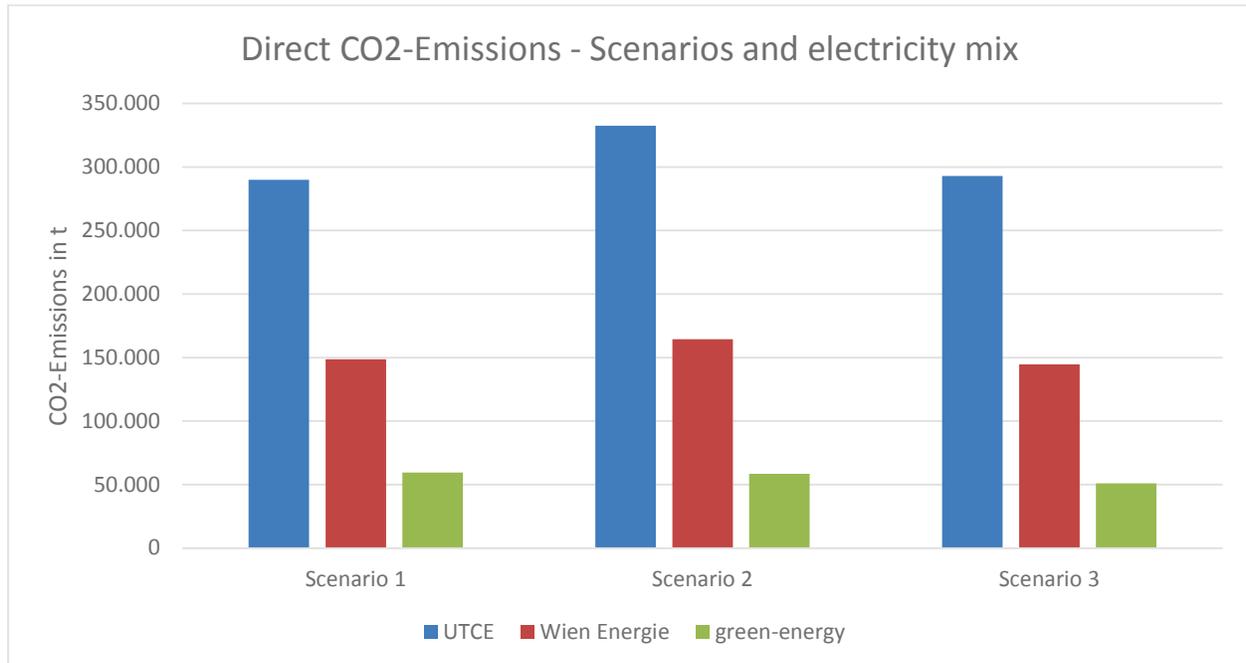


Figure 3 Direct CO₂-Emissions, for Status Quo and each Scenario as well as three sources of electricity

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DEVELOPMENT OF AN INTEGRATED RAW MATERIAL MODEL FOR URBAN SYSTEMS

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Introduction

According to estimates of the German Federal Ministry for the Environment, the German building stock contains around 10.5 billion tonnes of mineral building materials, around 220 million tonnes of timber products and around 100 million tonnes of metals. Due to continuous building activities, especially renovation and retrofit measures, it is estimated that this raw material stock will grow by a further 20% until 2050 [1]. The building sector is one of the most resource intensive economic sectors in Germany. The German National Strategy for Sustainable Development sets targets of doubling the raw material productivity until 2020 based on 1994 levels [2]. The developed raw material flow model for the building industry that is described within this paper can aid in reaching these proposed targets.

Model Development

The major aim of this project is to analyse the construction related material flows over the whole life cycle of urban areas. The basis for this inventory sets the identification of what types of materials and how much are contained in different residential building types of the German building stock and selected infrastructure such as roads. Furthermore a developed life cycle model can give an indication at what times raw materials are required (e.g. insulation for retrofit, or new construction), and when raw materials and potential pollutants become available again, for recycling or disposal, after the end of life of individual components of the analysed area.

The developed raw material cadastre can then be integrated into GIS systems, such as the CityGML standard as an additional layer and be linked to energy information for example (e.g. heating demand), to analyse the influence of raw material flows on the energy consumption of individual buildings and the analysed area as a whole. As the individual material flows (life cycle inventory) will be identified, it will also be possible to link this information to life cycle assessment (LCA) data to identify the environmental impacts (e.g. CO₂ emissions) the continuous changes of urban systems and the anthropogenic stock may have.

This integrated approach is not only examining the life cycle of material flows of urban systems over time, but it also tries to link and provide an interface to existing systems and calculation methods, to move towards rating the overall resource efficiency over time. As stated in VDI 4800, a conclusive rating of the overall resource efficiency of systems can only be achieved if

the use of all natural resources is being quantified and then placed into relation with each other [3].

The continuous spatial-temporal model will be able to capture how urban spaces change over time, from a raw material perspective. The 3D/4D model will be feed with actual data, where 3D indicates the spatial dimension, such as building geometries obtained from 3D city models (e.g. CityGML standard) and the fourth dimension (4D) represents the temporal dimension.

The object orientated modelling language, Universal Modelling Language (UML) was chosen to link the relevant attributes and influence factors which represent the complex structure of urban systems. This object orientated approach is a transdisciplinary approach and allows room for future expansion to provide a link to other models, beyond the system boundaries of this project. These links are very important as many different factors from different disciplines need to be considered to accurately model the dynamics of changing complex systems, in this case urban spaces.

Input data for the material consumption of different building types and building ages will be taken from existing databases and case studies, such as the catalogue of the “Institut für Ökologische Raumentwicklung (IÖR)” or “Institut für Wohnen und Umwelt (IWU)” typologies and other data sources. Data for the primary energy consumption of different building types and standard building components for example can also be abstracted from these sources.

The basis for life cycle assessment data will be the Ökobau.dat, a freely available German life cycle database, which contains data on most building materials used in the German construction industry [4]. With the aid of this datasets it will be possible to calculate indicators such as CO₂ emissions and embodied energy, which is the energy that is required to provide the materials, but also resource related indicators, such as the ADPelements (APDe), which purely rates the raw material consumption based on a reference material, can also be identified. The LCA will be based on international standards (ISO 14040 and ISO 14044) and will be in line with the criteria’s of the German building rating systems DGNB (Deutsche Gesellschaft für Nachhaltiges Bauen) and BNB (Bewertungssystem Nachhaltiges Bauen).

Building geometry and the geometry of individual components (e.g. wall areas, window area, roof area and type and other information) will be taken from existing 3D/4D city models, depending on the available level of detail (LOD). For many parts of Germany a comprehensive LOD 2 model is available, which has an accuracy of about 1 meter and shows a buildings individual roof type. A higher LOD can be achieved by constructing BIM (Building Information Modelling) models for example, of individual building types. LOD 3 models have a higher accuracy, and can show individual openings of facades (e.g. windows and doors). LOD 4, also referred to as walkable architectural models can provide information such as a building’s interior. These higher abstraction levels are however very time consuming and costly to build, especially if a larger number of buildings is being analysed. For these reasons LOD 2 models will be used as input data for the model.

A closer examination of potential pollutants that may be contained within the materials and demolition debris (e.g. asbestos, PCB) and the quality and purity of materials is also a very

important factor for the integral model. Especially buildings that have been constructed in the 1960s and 70s, that are now ready for retrofitting or demolition contain a variety of potentially harmful and hazardous substances that need to be properly recycled or landfilled.

The separability of individual building components is also directly related to the quality of individual materials. As the connection between the individual layers of building components play a vital role in the way the material can be recycled or be disposed of. The ability of a building component to be dismantled in an efficient way in the end of life phase sets the path for future recycling options and raw material flows.

Further Discussion

It is important to trace resources, in this case building materials and its raw material components not only in a special dimension, but also over time. As an indication can be derived at what times materials and potential pollutants from buildings become available for further use, such as a substitute for primary raw materials, further treatment or disposal on landfill sites. In a further step, the gained information from this model can also be used to identify the radial distance where it is still feasible to use secondary raw materials as opposed to primary material resources within the individual material fractions.

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MINING FROM ANTHROPOGENIC DEPOSITS: DEVELOPING THE FOUNDATIONS TO EVALUATE THE ACCESSIBILITY OF RARE EARTH METALS FROM END OF LIFE PRODUCTS

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Introduction

An increased number of geochemically scarce metals³¹ are entering our daily lives via new technological applications (Zepf *et al.*, 2014). A reversal of this trend is not foreseeable, leading to concerns about security of supply. Today's raw material situation is considered critical for many of these scarce metals because: (i) the production of raw material is concentrated to a few countries (Simoni *et al.*, 2014), (ii) there are limited options for substitutions (Graedel *et al.*, 2013), and (iii) recycling rates for these metals are very low (UNEP, 2012). In order to close material cycles and with this potentially ease the supply situation, raw material management requires to be rethought (Ongondo *et al.*, 2015). One approach is being implemented by the Swiss ordinance on the return, take-back and disposal of electrical and electronic equipment (ORDEE), which is currently being revised. The future ORDEE will require recovery of scarce metals from technological equipment where possible. This will not only apply for waste electrical and electronic equipment (WEEE) but also for electrical and electronic equipment from buildings and vehicles, provided that this is possible with proportional effort (FOEN, 2013).

How can this be tackled?

Prerequisites for the recovery of raw materials are its occurrence, i.e. its "availability", and its "accessibility". Whilst the notion of "availability" is typically well-understood, an explicit and systematic scientific examination of the concept of "accessibility" has not yet been provided. Implicitly, individual aspects of raw materials accessibility have been included in studies of economic geology, for instance within different resource classification frameworks (Weber, 2013). A first implementation of such implicit aspects in anthropogenic deposits was conducted by Mueller *et al.* (2015). They developed a framework that allows the establishment of analogies between geological and anthropogenic processes, and applied this framework to three products

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³¹ Geochemically scarce metals are those metals, whose crustal abundance is <0.01 weight-% (Wäger *et al.*, 2012).

containing rare earth elements (REE), aimed at identifying the most concentrated deposits in the anthropogenic cycle.

Based on these implicit applications, the following research questions can be derived: How can “accessibility” be conceptualised in the context of geogenic mining approaches? How can these concepts of “accessibility” be transferred to anthropogenic deposits under explicit considerations of the requirements for a more sustainable raw material mining? How accessible are anthropogenic raw materials?

What is the PhD research addressing?

This PhD research aims to elucidate and apply the concept of accessibility to raw materials from anthropogenic deposits. Thereby, particular attention will be paid to the requirements of sustainable raw material mining. The study will involve the following steps:

1. Elucidation of the term “accessibility” and its use in the context of geogenic and anthropogenic deposit discovery.
2. Identification, selection and weighting of key factors that influence the accessibility of both geological and anthropogenic raw materials under consideration of sustainability constraints.
3. Modelling and evaluation of the accessibility of REE in anthropogenic deposits based on the developed framework. This will be applied onto electrical and electronic equipment in buildings and vehicles.
- 4.

What has been addressed so far?

The first research step has been tackled. This was split into two separate analyses; first, the elucidation of the term “accessibility” by extracting semantic relevant data; secondly, analysing the use of “accessibility” in the context of geogenic deposit discovery.

For the first analysis, a generic applicable definition for the geological and anthropogenic context was derived from the dictionaries Oxford, Cambridge and the lexical database WordNet. Following, “accessibility” can be described by synsets³² “availability” and “approachability” (WordNet, 2014). On this basis, the extraction of semantic relevant data was conducted. This included a structural, statistical and semantical analysis as suggested by Weinhofer (2010). Therefore, 161 papers and books from geogenic and anthropogenic deposit discovery literature were studied with the PDF-XChange Viewer software (2014). This analysis of “accessibility”, “availability” and “approachability” resulted in the following. The term “approachability” was not used in this context. Both terms “accessibility” and “availability” are almost not used in the heading, but in the body of the text within diverse contexts, such as geology, waste management, technology, infrastructure, society, environment, governance and economy. This extraction showed concurrence with the generic definition. This first research step was concluded with the development of our own definitions to direct future research. The working definitions comprise: **available / availability**: “exists” in the geosphere and/or anthroposphere at

³² Sets of cognitive synonyms (WordNet, 2014).

"relevant"³³ mass fractions; **approachable / approachability**: it is possible to get to the material of interest but it is unknown if the material actually exists at "relevant" mass; and **accessible / accessibility**: there are no "major"/"significant" constraints (e.g. ownership, protected areas, ...) to "access" / "get to" the material of interest in view of potentially extracting it.

For the second analysis, approaches from geological deposit discovery were critically analysed. These approaches were then transferred based on existing similar research; and further developed to evaluate the development of both the geogenic and anthropogenic deposit discovery. This was complemented through discussions with experts. On this basis, the understanding of "accessibility" to raw material was positioned at an early stage of the classification / evaluation steps of the deposit discovery approaches. This "accessibility" positioning enables a prospective early stage comparison for mining projects. This research was concluded with the development of a conceptual framework to guide future research on quantifying the degree of accessibility (Figure 1). This framework comprises of three key stages: first, sustainability constraints, which provide the basis for the stages; second, classification / evaluation steps; and third, key influencing factors. Currently, sustainability is not integrated consistently within the classification / evaluation steps. Therefore, we propose to restrictions through sustainability constraints through the entire quantification of accessibility. For a broad sustainability integration, we suggest to integrate the domains from the mineral resource landscape perspective: technology, governance, environment, society and economy (Giurco and Cooper, 2012). The second stage comprises the classification / evaluation steps. These pose a successive description for the final decision on mine development. Consequently, they are the systemic structure towards deciding on mine development. The classification / evaluation steps encompass three successive main steps: 1. "reconnaissance exploration", 2. "detailed follow-up exploration" and 3. "final evaluation". The "reconnaissance exploration" is a macro-level evaluation that aims to sort out promising "prospects"³⁴ rapidly with low costs. The "detailed follow-up exploration" is a micro level evaluation that concludes with a prefeasibility study.

³³ Raw material estimated with a high level of confidence and hardly any obstacle will prevent its exploitation (Pohl, 2011).

³⁴ Prospect is a distinct volume of ground that is considered to have the possibility of directly hosting an "ore body" and is usually a named geographical location (Marjoribanks, 2010).

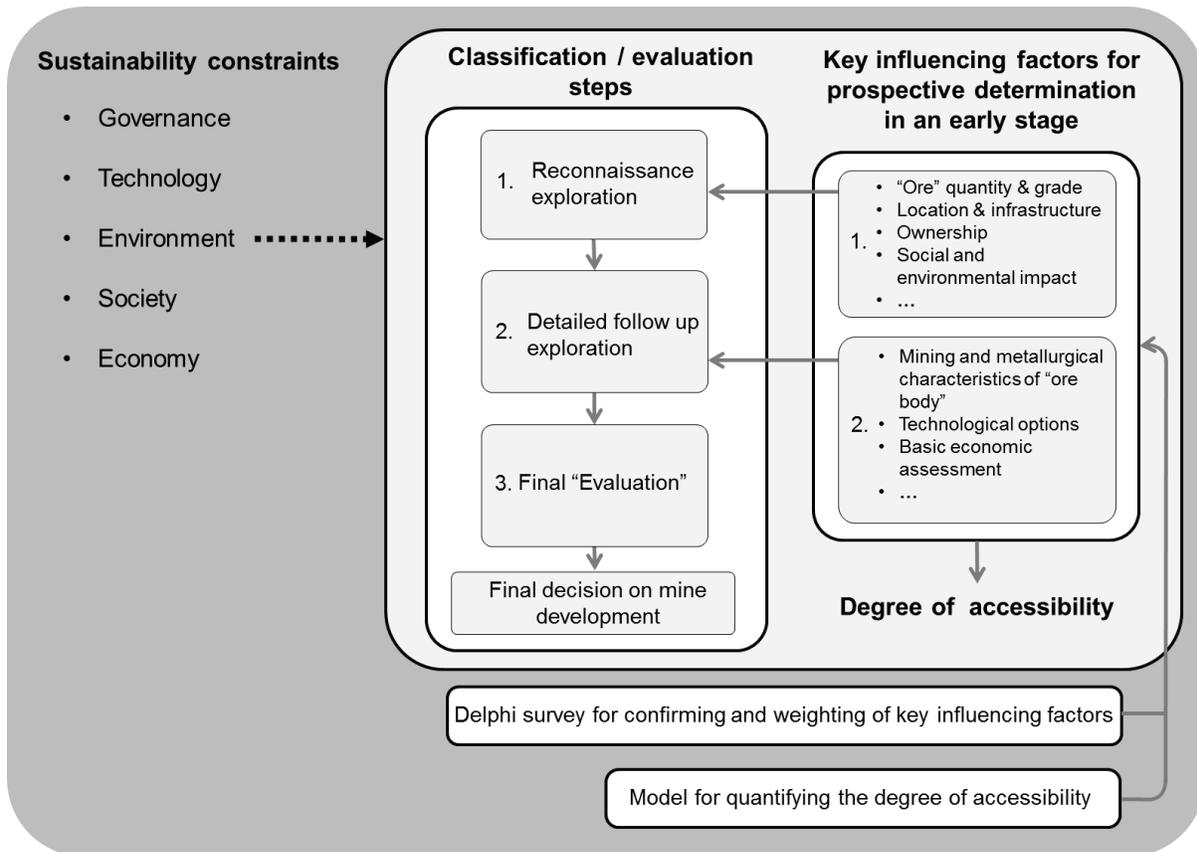


Figure 1. Approach to quantify the degree of accessibility of geogenic and anthropogenic raw materials.

The last step, i.e. final "evaluation", aims to generate comprehensive data from the "prospect" to determine the final decision on mine development and satisfy investors. This step is concluded with a feasibility study. The feasibility study is summarised by classifying the final decision in e.g. a resource classification framework. Within the classification / evaluation steps, each sequential step is influenced by different key influencing factors. In geology they are known as modifying factors or quantifying elements. Some of these key influencing factors affect the final result more than others. At present, it is unclear which of these factors the most important ones are and how strongly they influence the "degree of accessibility". By understanding the effects from selected key influencing factors at an early stage, a prospective statement on the degree of accessibility of raw materials can be derived. This applies for both geogenic and anthropogenic deposit discovery. On this basis, a foundation for establishing a common platform between geogenic and anthropogenic deposit discovery is provided. Importantly, a prospective statement on the recovery of scarce metals with a proportional effort can be formulated, which quantification is required according to the future ORDEE.

What are the next research steps?

The identified factors that influence the “accessibility” of raw materials will be confirmed and weighted using a Delphi study with independent experts. On this basis, a model will be developed and simulation experiments will be carried out to quantify the degree of accessibility of REE in buildings and vehicles.

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ENHANCING PHOSPHORUS GOVERNANCE IN AUSTRIA: POTENTIAL, PRIORITIES AND LIMITATIONS

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Abstract

The need for enhancing P (phosphorus) governance in Austria, as in most other countries, is driven by two major objectives: protecting surface waters from eutrophication and ensuring future food and energy security under scenarios of uncertain supply. Furthermore, the management of P rich flows constitutes a key element to achieve the ambitious goal of the European Commission to move towards a circular economy that enables economic growth by minimizing waste, use of raw materials and environmental damage (EC 2014).

The first step necessary towards these objectives is the analysis and understanding of the system representing the current situation, in order to identify losses, inefficiencies and processes which require action. This was initiated in Austria by Egle et al (2014a), who performed a detailed MFA study for an average year within 2004-2008. Zoboli et al (in press) carried it forward, by performing a historical analysis between 1990 and 2011 and examining more in depth the data quality and its implications in view of decision making and future monitoring.

Once the system has been thoroughly analyzed and understood, it is necessary to move forward and to assess how it can be optimized. A number of studies have investigated in detail specific issues in Austria, namely the potential for P recycling from wastewater and sewage sludge (Egle et al, 2014b), the potential impact exerted by a shift of dietary habits on nutrients flows (Thaler et al, 2015), and the performance of the national agri-environmental ÖPUL programme in reducing diffuse emissions of nutrients to water bodies (Gabriel et al, 2014). However, the holistic assessment of the improvement possibilities, which shall provide decision makers with a comprehensive perspective and with the capability of setting priorities, is still missing.

The objective of this analysis is to select fields of action aimed at optimizing national P management and to discuss their applicability to the Austrian system, their effectiveness, time frame and costs. Furthermore, the relative impact of each field of action on the overall national P management is quantified through two proposed indicators, in order to allow for an easier comparison and prioritization.

For this analysis the timespan covered by the MFA has been extended to 2012 and 2013, so to assess the optimization potential of a more updated system.

The selected fields of action are grouped in 3 categories: 1. Reduction of P demand and consumption; 2. Increase of P recovery and recycling; 3. Reduction of emissions to water bodies.

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For each selected field of action, the potential improvement with respect to the reference year 2013 is quantified and barriers and opportunities are analyzed. Furthermore, the uncertainty affecting the assessment of the status quo and the estimation of the potential is discussed and priorities for improving data collection schemes are identified.

The results are displayed in an optimized but hypothetical system that shall serve as a target state.

This study case on P governance and monitoring in Austria is an example of the maturity of the MFA approach as basis to design national materials accounting schemes and to support environmental decision-making.

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EXPLORING THE AMOUNT OF PLASTICS IN THE FEED OF AUSTRIA'S WASTE TO ENERGY PLANTS

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Introduction

Consumption of plastics and thus also the generation of waste plastics have increased tremendously during the last decades. Whereas at the beginning of the 1980s global consumption of plastics amounted to about 65 million tons (PlasticsEurope, 2013), in 2013 worldwide production has increased to almost 300 million tons (PlasticsEurope, 2015). A significant share of plastics (more than one third) is used in very short-life products, such as packaging, and are thus almost directly contributing to present waste generation (unlike plastics used e.g. in the building and construction sector) (PlasticsEurope, 2015).

In many affluent countries separate collection of specific plastic wastes (with special focus on packaging plastics) has been introduced during the last decades. The aim is to reduce the quantity of mixed household wastes, and to generate post-consumer waste streams that contain polymer types which can undergo high quality recycling.

The amount of plastic waste separately collected is usually recorded quite accurately, simply due to its economic value, but also due to its positive image for the plastics industry (e.g. contribution to circular economy). The vast majority of waste plastic generated however is not separately collected, but together with other materials (e.g. municipal solid waste (MSW), commercial waste (CW), waste electrical and electronic equipment (WEEE)), thereby resulting in "mixed wastes". Data about the plastic content in these mixed wastes is derived from literature or is based on sorting analyses. Due to the fact that waste composition may show significant variations even over time periods of some days (Morf and Brunner, 1998), a few single sorting campaigns are not sufficient to calculate a reliable annual average plastic content in mixed wastes. Furthermore, the plastic content of wastes determined via sorting analyses may be of limited significance even for the respective waste sample analysed: 1) due to the lack of visual recognisability of different materials and 2) due to the fact that sorting analyses usually aim at determining the content of different waste fractions (such as biowaste, hygienic products, composite materials,...) that do not necessarily contain only plastics or are free of plastics.

All these limitations demonstrate that figures about plastics in mixed wastes and thus also total quantities of plastic wastes generated are associated with significant uncertainties. Hence, also

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data about the recycling or thermal recovery quota of plastic waste, as published for different European countries (e.g. BIO Intelligence Service, 2013; Bogucka et al., 2008; PlasticsEurope, 2013), are rather uncertain.

The aim of the current study was to determine the total amount of plastics present in mixed wastes (collected as MSW or CW) by applying a novel approach, the so called Balance Method (Fellner et al., 2007). The investigations have been conducted for Austria for the year 2014.

Method

The Balance Method, utilized to determine the plastic content in mixed wastes, was originally developed to evaluate the ratio of energy from biogenic sources in the feed of waste to energy plants, but it also allows calculating the content of plastics in the waste feed.

The Balance Method combines data on the elemental composition of biogenic and fossil organic matter with routinely measured operating data of the Waste to Energy (WtE) plant. In principle the method utilizes one energy balance and five mass balance equations, whereby each balance describes a certain waste characteristic (e.g. content of organic carbon, lower calorific value). Each balance equation encompasses a theoretically derived term (left side of equations) that has to be attuned to measured data of the incineration plant (right side of equations). A simplified structure of the set of equations is illustrated in Figure 1, whereas the detailed mathematical description of each equation is given in Fellner et al. (2007). Besides the elemental composition of biogenic and fossil organic matter present in waste, the Balance Method requires the following operating data of the WtE plant: quantity of fuels incinerated (waste mass and auxiliary fuels), the amount of solid residues and steam produced, as well as data about the volume and composition of the dry flue gas (O_2 and CO_2 content).

Because the system of equations (set of constraints) used within the Balance Method is over-determined (6 equations for 4 unknowns), data reconciliation has to be performed to eliminate data contradiction and to improve the accuracy of the results. The reconciled values are subsequently used to compute the unknown quantities (mass fraction of biogenic matter m_B , of fossil organic matter m_F , of inert matter m_I and water m_W) including their uncertainties.

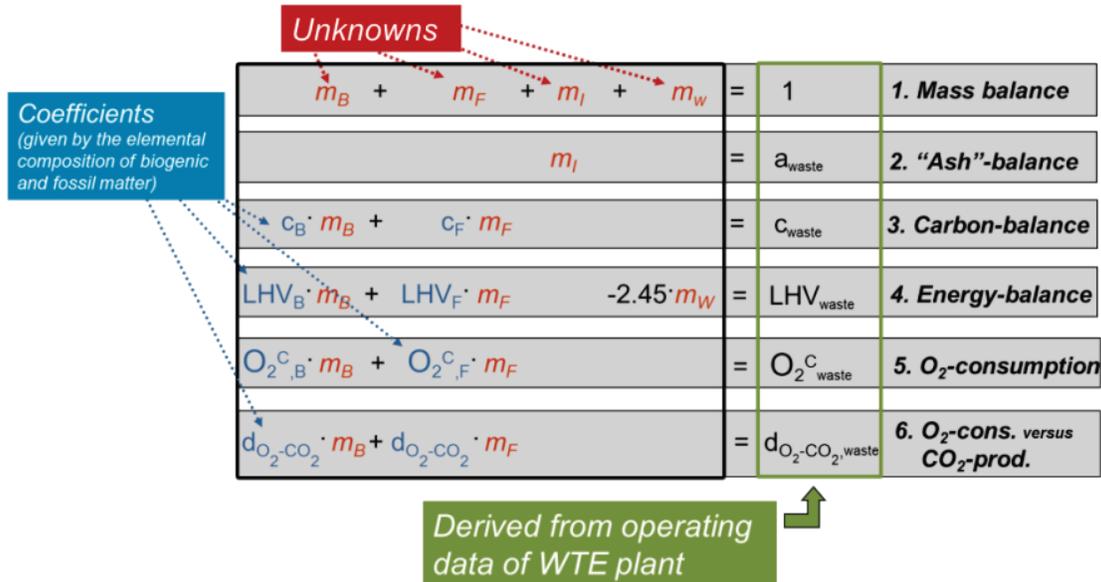


Figure 1 Simplified set of equations used by the Balance Method

The mass fraction m_F represents the content of synthetic polymers in the waste feed of the plant. By considering typical values for the ash content of plastics a_p (representing the content of inorganic additives and fillers), the fraction of plastics c_p in the waste feed can be easily determined according to the following equation:

$$c_p = \frac{m_F}{(1 - a_p)}$$

Based on national material flow studies focusing on Austrian plastics production and consumption (Bogucka and Brunner, 2007; Fehringer and Brunner, 1997) the average content of inorganic additives and fillers a_p is estimated to 90 ± 40 g/kg plastic, which is used in the present study.

Materials

The Balance Method has been applied to all Austrian WtE plants, with the exception of three facilities (two are combusting only hazardous waste and one facility was under reconstruction during the year 2014). Thus altogether the waste feeds from 10 waste incineration plants have been characterized with respect to their plastic content. Based on the calculated composition of the waste feed of each plant (using the Balance Method) and its respective annual waste throughput the total amount of waste plastics thermally recovered has been determined.

Table 1 gives an overview of the 10 WtE plants that have been investigated. The annual capacity of these facilities amounts to about 2.1 million tons of waste. The plants utilize different types of combustion technologies (grate incineration or fluidized bed combustion) and mainly incinerate municipal solid waste, commercial waste, sewage sludge and refuse derived fuels (see Table 1), whereby the share of the different wastes may vary significantly throughout a year.

Table 1: Overview of Waste-to-Energy plants in Austria

WtE plant	Combustion technology	Waste incinerated (qualitative information)
A	Grate incinerator (GI)	MSW
B	Grate incinerator (GI)	MSW and CW
C	Stationary fluidized bed combustion (FBC)	RDF and SS
D	Stationary fluidized bed combustion (FBC)	RDF and SS
E	Circulating fluidized bed combustion (FBC)	RDF and SS
F	Grate incinerator (GI)	CW
G	Stationary fluidized bed combustion (FBC)	RDF
H	Grate incinerator (GI)	MSW and CW and minor amounts of SS
I	Grate incinerator (GI)	MSW
J	Grate incinerator (GI)	MSW and CW and minor amounts of SS

MSW... Municipal solid waste, CW ... commercial waste, SS... sewage sludge, RDF... refuse derived fuels

Results and Discussion

Prior to the application of the balance method, the input data, in particular the operating data of the plants have been checked regarding their plausibility. Thereto, existing correlations between the volume & composition of the flue gas and the steam production have been used. In total almost 97 % of the overall waste feed (thus more than 2 million tons out of 2.1 million tons waste throughput) has been analysed in the study, a sample size that can hardly be achieved by any other analysis method.

Figure 2 shows the plastic contents in the feed of selected WtE facilities given as monthly averages (incl. uncertainties). The results demonstrate large temporal variations in the waste composition and also significant differences between the plants. For instance the plastic content in the feed of WtE plant E varies between 18 % and 26 % (weight percentage). The lowest plastic contents were observed for WtE plant C (annual mean of 11 %), whereas the feed of WtE plant F is characterized by the highest content of plastics (annual mean of 21 %). In general the results of the analyses indicate that the content of plastics in the feed of waste incinerators is by trend higher for plants with a higher proportion of commercial waste.

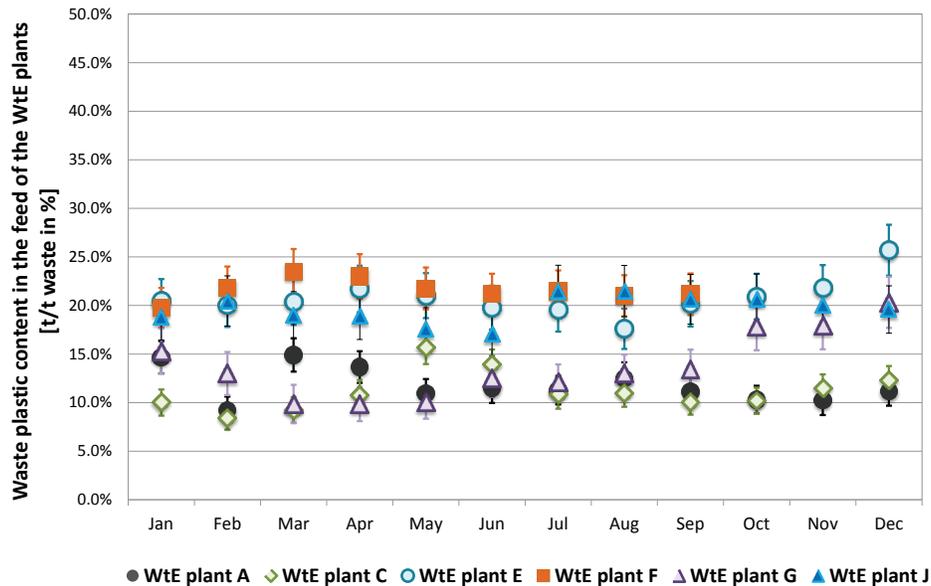


Figure 2: Plastic contents (incl. their uncertainties) in the waste feed of selected WtE plants – given as monthly averages (including statistically derived uncertainties)

In Figure 3 the annual flows of waste plastics through all 10 WtE plants are summarized. In total about 347 ± 16 kt of plastics have been thermally utilized in Austria’s waste incineration plants in 2014, which corresponds to an average plastic content of $17\% \pm 1\%$ in the waste feed. Assuming an average lower calorific value of waste plastic of 33.5 MJ/kg this equals an energy input of $11,600 \pm 500$ TJ/a via plastic waste.

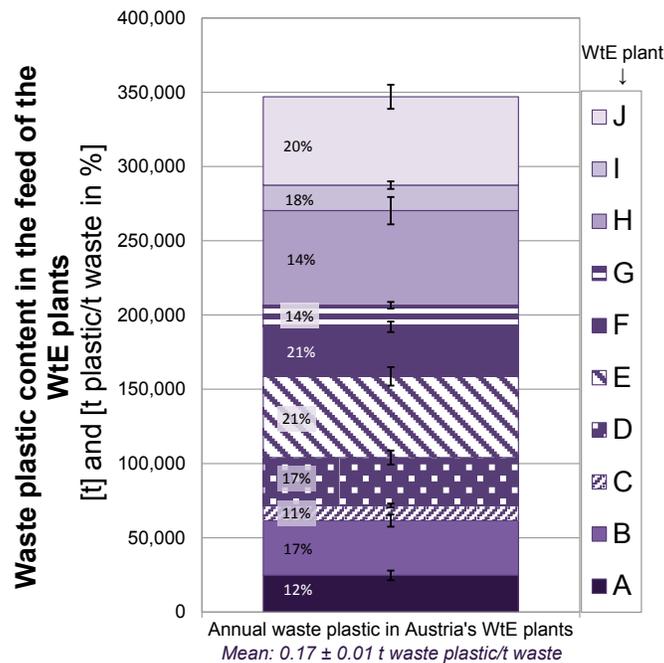


Figure 3: Total amount of waste plastics in Austria’s WtE facilities in tons (incl. their uncertainties) and average plastic contents related to the total waste feed (given in %) for the year 2014

Looking at the total plastics waste stream produced in Austria, this utilization of waste plastics in WtE plants (347 kt) represents by far the largest share of all processes, compared with mechanical (160 kt in 2010) and chemical recycling (65 kt). Noteworthy is the fact that a significant amount of mechanically and chemically recycled plastics originates from production or imported waste. The dominance of WtE plants in the overall waste plastics processing means that there is still potential for further increasing the material and chemical recycling rates by diverting waste plastics away from the municipal solid waste, by means of enhanced collection schemes or effective separation technologies. However, what kind of separation processes can be utilized, how effective these technologies are (also with respect to generate “clean” plastic wastes without hazardous additives), and to which extent these processes are feasible from an economic but also from a technical point of view, needs to be further investigated. In the meantime Austria’s WtE plants contribute to an environmental friendly and energy-efficient utilization of post-consumer waste plastics.

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DOMESTIC E-WASTE ESTIMATION FOR RURAL COMMUNITIES BASED ON ENERGY ACCESS DATA IN BANGLADESH

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Short Abstract

Universal access to modern energy services like electricity till 2030 is one of the objectives of the Sustainable Energy for All Initiative (SE4ALL). To track the progress towards the SE4ALL objectives a multi-tier framework was developed. Based on the collected data for testing the multi-tier framework in rural Bangladesh, a methodology is in development to approximate future domestic e-waste. One criteria of electrification is access to energy services where electrical and electronic equipment (EEE) in households are surveyed. Preliminary results show that depending on the tier of electrification the stock of EEE per household is between 0,05kg to 20,8kg. The goal is to have a dynamic model approximating e-waste flows based on energy access data.

E-Waste generation in Energy Access Deficit Countries

The 20 countries with the biggest access to electricity deficit (CBAED) have a population of almost 2.7 billion and 889 million still have an access to electricity deficit. The CBAED are either low- or lower-middle income countries (LMI). The per capita E-Waste generation in the CBAED is currently on average 0,84kg per year which is little compared to the 15,4kg in Europe (Worldbank 2015, Baldé et. al. 2015, own calculations). Nevertheless the CBAED already account for 8% of the global E-Waste. The average E-Waste generation for LMI is 2.6kg per capita and therefore almost three times the one of CBAED. Furthermore global E-Waste flows are expected to grow by 4 to 5 % annually. Preliminary results show that if the CBAED will improve their access to electricity to the standard of LMI they will contribute more than 34% of the global E-Waste in 2030. The SE4ALL objectives are even higher than lifting the access to energy of the CBAED to the one of LMI.

Access to modern Energy Services and Sustainable Development Goals

In 2011 Ban-Ki Moon announced the SE4ALL-Initiative. One objective is to provide universal access to modern energy services like electricity till 2030. In 2015 still 1.1 billion people lack access to electricity (Worldbank 2015). Access to energy was not included in the Millennium Development Goals (MDG) but is likely to become the seventh Sustainable Development Goal (United Nations, 2015) and should be adopted in the post-MDG agenda in September this year.

Off-grid electrification based on photovoltaic systems has already proven to contribute significantly to rural electrification especially on a household level. So called solar home systems (SHS) play a crucial role in rural electrification and are sold on market terms in countries like Bangladesh, India or Tanzania (Palit, 2013). Therefore two questions shall be answered. How can energy access data contribute to E-Waste estimation in developing countries, especially for the least developed? How does electrification based on SHS influence estimated future domestic E-Waste flows?

Measuring Access to Electricity

Electrification used to be measured in a binary way by international institutions like the International Energy Agency or the World Bank. Either a household is electrified, meaning legally connected to the grid, or not. The datasets were comparatively easy to collect and part of regular household surveys; nevertheless some developing countries with officially high electrification rates are falling short on providing energy services to their population, due to insufficient power supply. Therefore a new methodology – the global tracking framework - was developed to track the progress towards the SE4ALL objectives (Worldbank 2015). For access to energy a multi-tier matrix with different frameworks for households, communities, institutions and companies was developed. There are five tiers of electrification. For households tier 0 means no access to electricity, while tier 5 is close to the standard of electricity supply in industrialized countries. Table 1 illustrates the parameters to qualify households into different tiers of electrification. In the questionnaire a list of 27 EEE is included and therefore data about the stock of household’s EEE is collected. By the end of 2015 data for the 20 biggest energy access deficit countries shall be available.

Table 1 Multi-tier Matrix for Access to Household Electricity Supply combined with the Multi-tier Matrix for Access to Household Electricity Services (Based on ESMAP 2015)

		TIER 0	TIER 1	TIER 2	TIER 3	TIER 4	TIER 5
Capacity	Power		Very Low Power Min 3W	Low Power Min 50W	Medium Power Min 200W	High Power Min 800W	Very High Power Min 2kW
	AND Daily Capacity		Min 12Wh	Min 200Wh	Min 1.0kWh	Min 2.4kWh	Min 8.2kWh
	OR Services	Not applicable	Task lighting Phone charging	General lighting Television Fan (if needed)	Tier 2 AND Any medium-power appliances	Tier 3 AND Any high-power appliances	Tier 4 AND Any very high-power appliances

Data Collection, Methods and Sample

The dataset being used in this research is the one Groh et al. (2015) collected for testing the multi-tier framework in Bangladesh. In the following methods and samples are presented shortly. A sample of 231 rural households was surveyed. The data field selection was performed in a top down way. One random district was drawn from the Northern, Central and Southern part of the Bangladesh. Three types of households were identified and a random selection within the

groups was drawn. Data was collected based on the generic underlying questionnaire of the multi-tier framework. Table 2 shows the access types of the interviewed households of the dataset. The dataset is not representative for Bangladesh, since only rural households were surveyed. Nevertheless this dataset is useful for demonstrating the methodology.

E-Waste estimation based on Energy Access Data

The data of EEE per household were connected to a data being published in the Global E-Waste Monitor (Baldé et. al. 2015). All 27 appliances of the multi-tier framework were linked to the representing UNU-Codes. Within the Global E-Waste Monitor average weights and lifespans for EEE based on data mainly from European countries are published. Neither in the dataset of Groh et al. 2013 nor in future datasets of the global tracking framework information about sales or age of the appliances will be surveyed. Only the age and the capacity of SHS are available to determine the tier of electrification of a household. The next step (ongoing) for this research is to approximate an E-Waste flow out of the EEE-Stock. Wang et. al. (2013) described various types of E-Waste estimation models. For this research a combination of two E-Waste estimation models will be used. For households having a SHS it was assumed that all related appliances (TV, Fan, LEDs) were sold in the same year as the SHS. Therefore a market-supply model can be applied. Due to the scarcity of data the model for all other appliances will be a leaching model. Within the next four weeks the model for the E-Waste estimation will be finished. The results are average stocks and flows of EEE and E-Waste on different tiers of electrification for a year. For the dynamic modelling until 2030 the electrification plan for Bangladesh of the SE4ALL-Initiative will be used. In that plan the objectives for the different tier groups until 2030 are defined.

Preliminary Results:

Since the data was collected in rural Bangladesh, a major energy access deficit within the households was observed. None of the households reach Tier 4 or Tier 5 electrification level, meaning that none of the households has any high or very high consuming power appliances, like a washing machine or an electric cooking system. Within the dataset the average stock per capita of EEE was 2,0 kg and 3,1kg including the SHS. Tier 3 households have obviously the highest stock of EEE and all households are connected to the electricity grid. For 95% of Tier 2 and 46% of the Tier 1 households a SHS is the primary source of energy. The average weight of all components of SHS excluding the lead-acid battery in the dataset is 8,6kg (own calculations). The average weight of the lead-acid battery is 10,5 kg³⁷. Including the SHS Tier 2 households have a stock of EEE of 4,5 kg per capita, which is almost double the stock without the SHS.

³⁷ Batteries are not considered as E-Waste in this abstract. Nevertheless the recycling of used lead-acid batteries of SHS is seen as the biggest negative environmental impact of SHS.

Table 2 Tier Status of electrification and stock of EEE per household (HH) and per capita (Groh et al. 2015, own calculations)

	# of households	Stock of EEE in kg/HH (including SHS)	Person per HH	Stock of EEE in kg per capita (including SHS)
Tier 0	57	0,05 (0,05)	3,5	0,02 (0,02)
Tier 1	23	4,9 (9,2)	2,6	1,9 (3,5)
Tier 2	134	8,7 (17,2)	3,8	2,3 (4,5)
Tier 3	17	20,8 (21,0)	3,2	6,6 (6,7)
Tier 4 and Tier 5	0	n.a.	n.a.	n.a.

Discussion:

It is absolutely clear that with a higher tier status, a significant higher stock of EEE is observed. Electrification based on SHS is more material intensive on a household level, because the power supply is added to the stock of EEE of a household. Unelectrified households neither contribute significantly to the stock of EEE nor to E-Waste generation. In Bangladesh 43% of the population have no access to electricity (Worldbank 2015). Most of them will be electrified with decentralized solutions like SHS.

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URBAN MINING AT A REGIONAL LEVEL IN FRANCE: THE CASE STUDY OF PARIS REGION

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Paris region: a mature and developing urban area

Paris administrative region (*région Ile-de-France*) is an 11.3 million inhabitants highly urbanized area which is about to experience an important development. Indeed, a regional development plan for 2030 called *Grand Paris* has been launched recently. Its objectives are the construction of 205 km of mainly underground metro lines, 69 stations, and 22 urban development projects (70 000 housings/year and 600 000 new jobs expected until 2030). As both metro lines and urban projects are mainly located in the center of the region, the implementation of this plan will have a great impact on the volume and the location of construction material input and output flows. Indeed whereas today's aggregates consumption is 33.2 Mt/year, estimates show that approximately additional 7.2 Mt/year will be needed for the *Grand Paris* (2.5 Mt/year for the metro lines and 4.7 Mt/year for buildings). Concerning the output flows, approximately 30 Mt/year of C&D waste are generated today in the region when estimates show that 4.6Mt/year will be generated in addition by the plan (2.6 Mt/year of excavated material for the underground metro lines³⁹ and 2 Mt/year for the urban projects) (Commission du développement durable, 2015).

Only 45% of the minerals consumed by the region today are extracted locally: 28% are imported from nearby regions located between 50 and 120 km from Paris and 17% from farthest areas (UNICEM et al., 2008). Moreover, as in Paris region, urban sprawl in some of those supplying regions generates land use conflicts which lead to a growing scarcity. On the output side, only 24.5% of the 24.1Mt of minerals waste generated each year are up-cycled: 34% are sent to landfills and 41.5% used for quarries reclamation⁴⁰, all those sites being located in the outskirts of Paris region (Conseil Régional 2013). This low recycling rate leads to a saturation level of landfill sites.

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³⁹ Excavated material can also be considered as unused domestic extraction.

⁴⁰ The use of minerals waste for quarries reclamation is generally considered as recycling in France.

Construction materials make up 40% of the region's Domestic Material Consumption (DMC) (Barles, 2009). In 2003, this consumption was very different in the center of Paris region (Paris and 3 *départements*⁴¹ called *Petite Couronne*), characterized by a high residential density, and in the originally agricultural and industrial outskirts (4 *départements* called *Grande Couronne*) which experienced a low housing density urban sprawl. Indeed, whereas the DMC in the center area remained below 1 t/capita (5.7 kt), it reached 4.8 t/capita in the outskirts (23.5 kt). This difference seems to point out the importance of linear infrastructure development in the material consumption, as the DMC per new built gross floor area remained at 2 t/m² in the region's center whereas it reached 4.5 t/m² in the outskirts (Barles, 2014). Hence the center of the region can be considered as a "mature" urban area where the outputs "represent more than a marginal contribution to the resource needs" (Brunner, 2011).

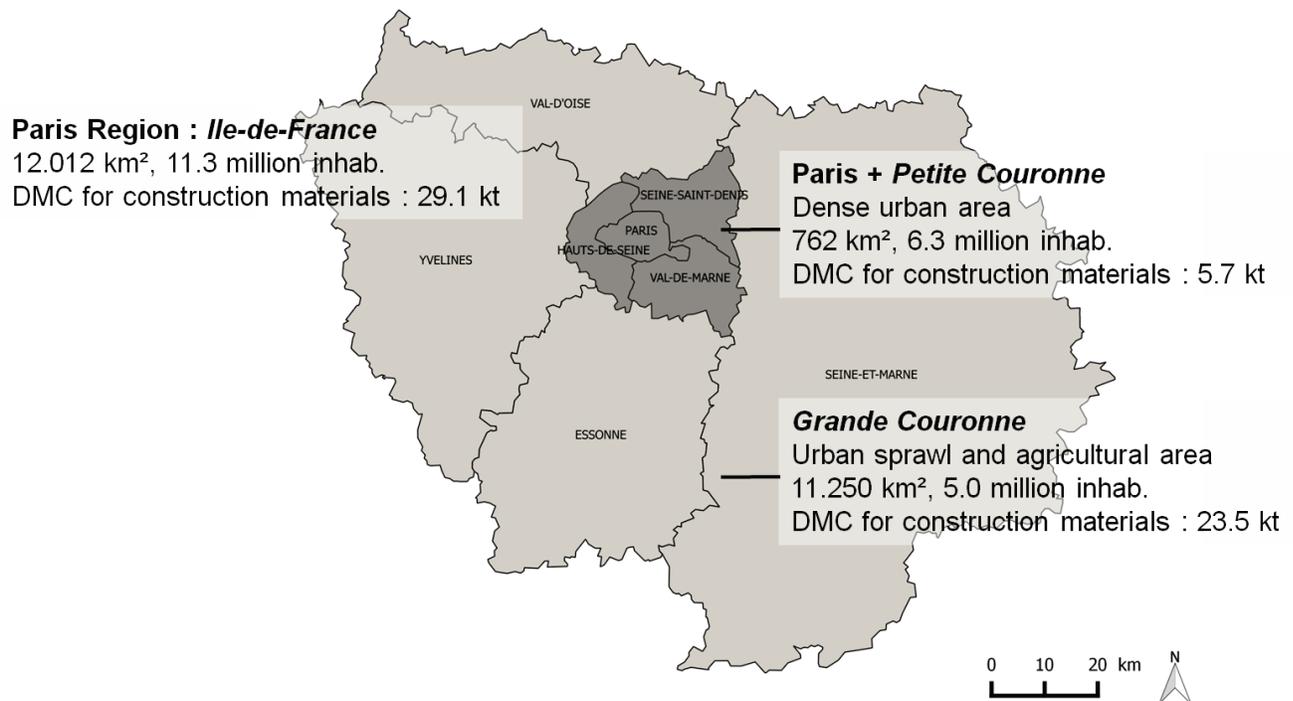


Figure 1 Domestic Material Consumption of construction materials in 2003 (after Barles, 2009) (DMC for Paris + *Petite Couronne* assuming that all rail traffic is within this area)

How could urban mining be set up in Paris region ?

Considering those issues, the development of construction materials recycling at a regional scale is defined as a priority target by the local authorities. Up-cycling of today's in-use stock would indeed enable the reduction of the amounts and lengths of input and output flows. A research project launched recently by Géographie-Cités laboratory with the regional council (*Région Ile-de-France*) in charge of the regional C&D waste management plan, and the regional

⁴¹ France is divided into administrative regions that are themselves divided into *départements*. Paris is an exception as it is both a municipality and a *département*.

office for environment (DRIEE) responsible of the regional quarries regulation plan, aims at helping those two authorities to set future urban mining objectives.

First a detailed assessment of the input and output flows of construction materials (including metal, glass, plastic and wood) in 1990, 2000 and 2010 for all Paris Region, its center area (*Paris and Petite Couronne*) and its outskirts (*Grande Couronne*) will bring a better understanding of the dynamic of those flows both in time and space. The Eurostat MFA method adapted to urban and regional scales by Barles (Barles, 2009) will be used for this analysis. The results will be compared with other case studies including Vienna (Daxbeck et al., 1997).

Secondly a “preview” assessment of the in-use stock in terms of quantity and location at a regional level and including the infrastructures (roads, networks) will give a first idea of the mining potential. Geographical information from a database called BD Topo® available for all France matched with census data give the volume and surface of buildings and infrastructures and 3 age periods for housings. These data were used at a city scale in France with a bottom-up method in ASURET research project lead by BRGM which brought interesting results (Rouvreau et al., 2012). Calculated per volume material contents from literature review including LCA studies and business expertise will be used to calculate the stock weight.

An analysis of demolition statistics in the region will bring a better understanding of the dynamic of the stock: which buildings in terms of type (apartment buildings, single family houses, offices...), age and location are demolished today? Which factor seems to be the most important? Though the public database Sit@del2 available for all France does not give detailed information about demolition, one for Paris city (GERCO) and another developed by the demolition professionals syndicate (SNED) could help us to take into account the stock dynamic. As the three regional plans (*Grand Paris*, quarries regulation and the C&D waste management) have short horizons (from 6 to 15 years), a static bottom-up assessment of the stock seems appropriate. However a dynamic modeling (Müller, 2006) could be very interesting in future works.

In a third step a field survey of construction material and waste management in two Grand Paris urban development projects will help us to better characterize the quality of the stock and the “realistic” short term mining potential. This work with project managers will show what can be done in the 15 coming years to reduce construction material flows and recycle considering the difficulties, whether they relate to technique, cost, organization or building codes and standards. Though many professionals show interest for recycling in France, they consider those factors as limits today. Finally, flows and stock assessment results will be used to set urban mining objectives with the regional authorities.

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POSSIBILITIES OF ALLOY DETECTION AS A NECESSITY FOR RECYCLING OF CRITICAL RAW MATERIALS

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Introduction and Background

With the product development in different product categories and changes and trends in technology, such as waste electronic and electrical equipment (WEEE), batteries, and end-of-life vehicles (ELV), more and more different metal alloys are used to improve material characteristics. For this, materials with an increasing content of precious and special alloying elements are used. The EU set up an ad hoc working group to assess elements regarding their criticality. To be characterised as critical an element has to fulfil two criteria: economic importance for the EU and supply risk [EU COMM, 2014]. A group of 20 elements and elemental groups (such as chromium, magnesium, cobalt, and gallium) were identified as critical for the EU. Considering this, an increased recycling of these elements is crucial to mitigate the effects of a possible supply risk.

Metal recycling bears great challenges regarding other than the mere recycling of base and carrier metals. The fate of alloying metals of end-of-life (EOL) products in pyrometallurgical processes is defined by their thermodynamic phase distributions. This defines in which phase alloying metals accumulate: metal phase, slag phase, or off-gas [UNEP, 2013]. When focusing on e.g. aluminium as a carrier metal alloying and disturbing metals are mostly dissolved in the metal phase and, if not used in the recycling product, are lost due to non-functional recycling. To shift these effects to a recycling of all alloyed metals a detection of the composition of processed material streams is crucial. One approach is the use of X-ray fluorescence analysis (XRF). This technique is already in use on scrap yards for sorting e.g. wrought from cast aluminium alloys or high-grade steels from low-grade steels and iron alloys.

To test the applicability of XRF-based alloy detection pre-trials were conducted. The main focus of this study lies on aluminium alloys from a WEEE processing plant, but also iron and steel alloys were included in the measurements. The objectives of the pre-trials are:

- to investigate the applicability of XRF for alloy detection on the basis of aluminium alloys
- to identify aluminium alloys used in a specific waste fraction (WEEE)
- to investigate the limit of detection for detected critical metals

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Future steps will extend the amount of measured steel alloys and extend the scope of the study to other metals, such as magnesium. Also additional waste fractions other than WEEE will be analysed, testing a universal application of the method.

Materials and methods

In pre-trials a mobile XRF device (NITON XL3t, analyticon GmbH) was used to analyse alloys with known composition (reference material; accessible at the TU Berlin) and sorted metal scrap from a WEEE processing plant. The mobile device has a pre-set analysing software tool for alloys holding a database with around 500 alloys. Using the database the software gives suggestions for alloys in addition to element mass fractions of 28 metals. The accessible samples analysed were:

- magnesium alloys
- nickel alloys
- copper alloys
- steel alloys
- non-ferrous scrap from a WEEE processing plant
- ferrous scrap from a WEEE processing plant

Results

As an example for light metal alloys figure 1 depicts the deviation of measured elements from given mass fractions for magnesium alloys (the values represent the difference between percentages of given and measured mass fractions).

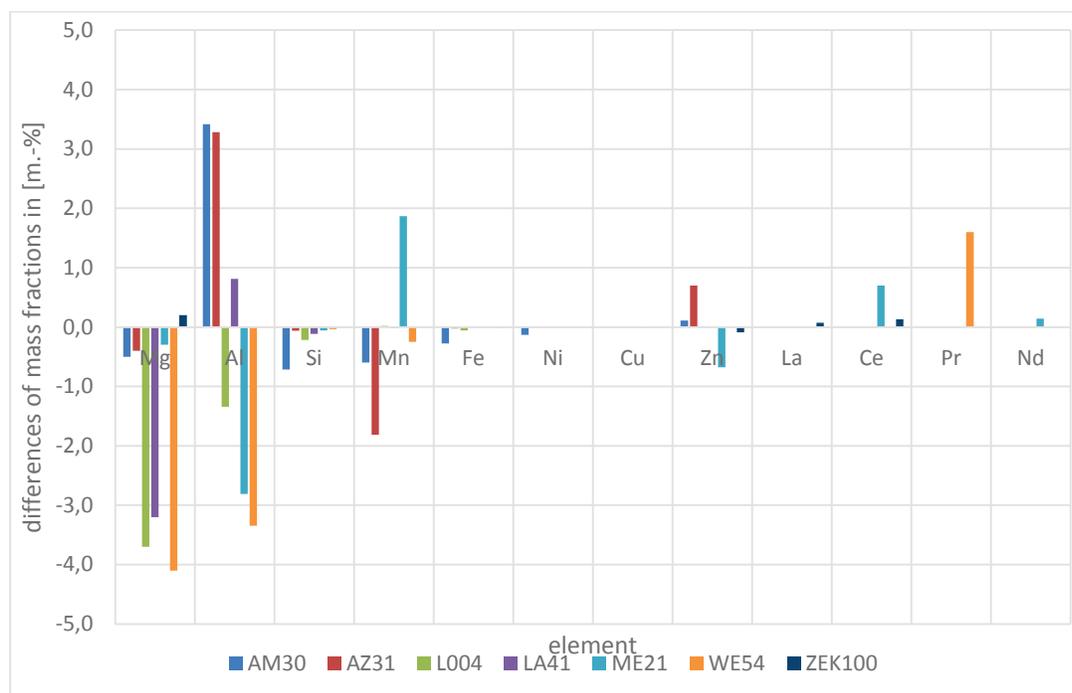


Figure 1 Comparison of measured and given composition for analysed magnesium alloys

Figure 1 shows that there are elements not always detected correctly, such as light elements like magnesium or aluminium. In general the light elements are hard to detect with X-ray fluorescence, because of the low energy emission of the nuclei (Beckhoff et al., 2006). But when used for alloying elements of these metals the focus is on the alloying elements and thus the results of the measurements are not impeded in a major way.

Another preliminary result that has to be verified in further measurements is that critical elements contained in the analysed alloys show only minor deviations between the given and measured mass fraction. This is a crucial prerequisite for a subsequent recycling of critical raw materials. Although it has to be mentioned that a value of zero in figure 1 can mean a difference of 0 when measuring elements with a given mass fraction of either > 0 m.-% or equal to 0 m.-%. It is important to note that Th is only contained in the material ME21, Nd in WE 54, Ce in ME21 and ZEK100, and La in ZEK100. These metals are not contained in the other alloys, i.e. the zero values are not an indicator for compliance with given mass fractions. In comparison with aluminium alloys, high-grade steel alloys have a higher content of critical alloying elements, such as chromium, cobalt, niobium, and vanadium. To further investigate detection limits and precision of analysis for critical metals using XRF these steel alloys have to be measured.

After testing the reference materials WEEE scrap samples containing non-ferrous alloys (mainly aluminium but also magnesium alloys) were analysed. The results of first measurements are shown in figure 2, depicting the mass fractions for aluminium and nine alloying metals.

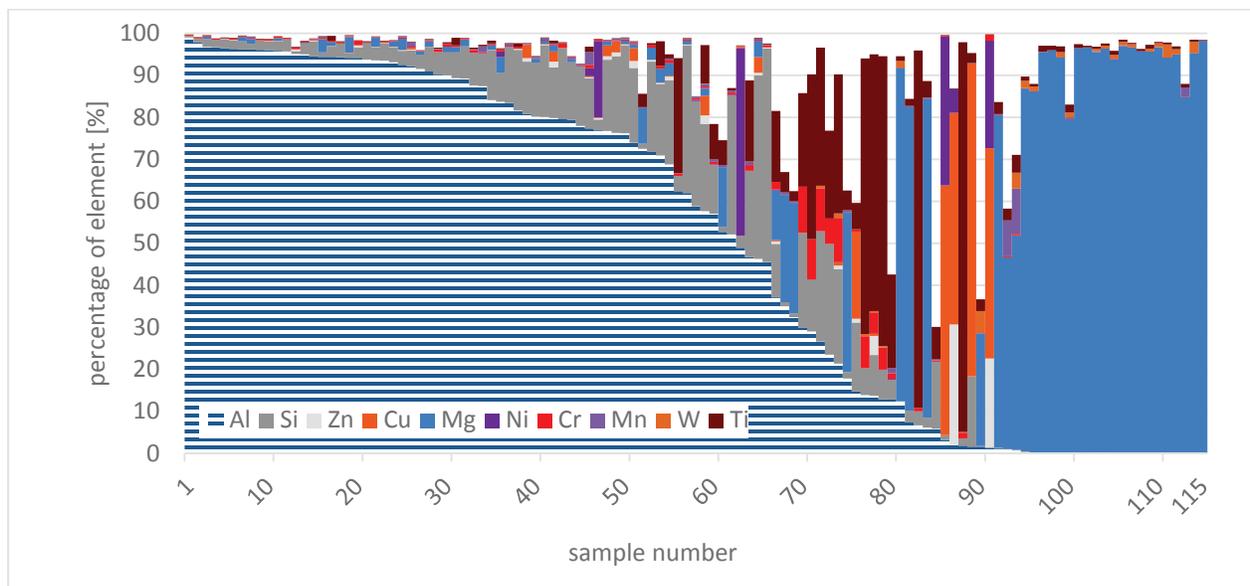


Figure 2 Results of measurements of non-ferrous scrap

In general different groups of scrap pieces can be identified in figure 2:

- aluminium content > 90 m.-% and low alloy content
- aluminium content > 80 m.-% and medium alloy content, esp. silicon
- mix of various elements, no alloy identifiable
- high magnesium content, suggesting magnesium alloys

Outlook

In a second pre-trial around 500 additional samples of non-ferrous scrap were measured analysing the mass fractions of the containing metals. The next steps are interpretation of the measured mass fractions (for non-ferrous as well as ferrous scrap samples), attempt to identify alloy series and detailed alloys (for aluminium and steel alloys), and further investigation of detection limits for critical metals. These results will be presented in the poster.

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QUANTIFYING PLASTIC FLOWS AND STOCKS IN AUSTRIA

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Introduction

Plastics have been the most used man-made material in the world since 1900, being applied in virtually all consumption sectors, both in applications with very short lifespans (such as packaging), and very long lifespans (such as piping in building infrastructure) (Hopewell et al., 2009; Ryntz, 2006). In 2013, the global plastics production reached 299 million tonnes (PlasticsEurope, 2015). The big success of plastics in a wide range of applications is explained by the various unique properties they possess: they can be used at a very broad temperature range, are resistant to chemicals, are light but still strong and tough, although they can be worked easily as a hot melt.

The use of many different polymers, combined with various additives, makes that plastics in general form a highly complex material stream. This leads to large and highly diverse material flows into the anthropogenic stocks, which needs to be managed well, as the production and use of plastics result in a number of environmental concerns. 90% of plastics are completely derived from non-renewable fossil fuels, making that around 4% of the annual petroleum production is converted into plastics, with an additional 3-4% needed for energy consumption during production. On the other end of the life-cycle, waste plastics form a significant and complex waste stream, part of which accumulates in the natural environment due to the high durability of the polymers. Recycling is thus a method to reduce both the environmental impact and resource consumption of the life cycle of plastics products (Al-Salem et al., 2009; Hopewell et al., 2009).

To achieve this recycling efficiently, detailed knowledge about the different plastics streams flowing through society should be established. However, only fragmented data of plastic waste generation are usually available. To overcome this, a Material Flow Analysis (MFA) can be performed, which balances all material inputs and outputs into society. This allows the anthropogenic resources to be identified and characterized, in order to optimize the management of existing plastic flows, and predict future waste streams. Therefore, in this study an MFA is performed for the Austrian plastics budget of 2010, to identify the key processes, stocks and flows, and to serve as a basis for further research into the dynamic modelling of this plastics budget.

Materials and Methods

The MFA method aims to connect the sources (e.g. imports), the pathways (e.g. transfer coefficients from manufacturing to consumption) and the intermediate (e.g. consumption) and final sinks (e.g. waste management) of materials, based on the law of conservation of matter. In this study, the method is applied according to Brunner and Rechberger (2004), and the software

STAN (freely available under <http://www.stan2web.net>) is used for the calculations (Cencic & Rechberger, 2008).

For this study, to model the total material budget of plastics in Austria different processes were used. First, in the chemical industry process, primary polymers are synthesized and traded. In Austria, the main producer of polymers is Borealis, and they directly provided production data for this project. Furthermore, production quantities of other production plants are derived from annual reports and other publicly available information. Data on the import and export of primary polymers are acquired from the Austrian statistical office (Statistik Austria, 2010).

Next, the polymers coming from the chemical industry process are moulded into semi-finished products such as sheets and films, and are then further converted into final products in the manufacturing and preparation process. During this process, additional additives are used for achieving the required properties of the product. Data on imports and exports of semi-finished products are again obtained from the Austrian statistical office (Statistik Austria, 2010).

The final products are subsequently used by consumers in the consumption process. This process is subdivided into ten sub-processes, representing the various consumption sectors: packaging, building and construction, transport, electronics, furniture, agriculture, medicine, household goods, others, and non-plastic applications.

No data were available on the distribution of the plastic products into the various consumption sectors in Austria, so data from a German study were used (Lindner, 2102), assuming that the distribution pattern is similar for both countries. For the transport sector, the number of vehicles entering the consumption market was obtained from WKO (WKO, 2012), while the number of tyres imported (Statistik Austria, 2010) was added to the sector as well. Data from the Austrian statistical office (Statistik Austria, 2010) were used again for the import and export of plastic products. In contrast to the processes discussed previously, the consumption processes have stocks that are built up during the use phase of the respective products, due to the longer lifespans of many products.

After the use phase of the products, they are disposed of to the waste management process. Different data sources are available to determine the size of the waste flows from the respective consumption sectors. Information on sectors with extensive regulation is generally available in reports from public authorities or related organizations (Amt der NÖ-Landesregierung, 2011; BMLFUW, 2011; Elektroaltgeräte Koordinierungsstelle Austria, 2011; Pladerer et al., 2002; TB Hauer, 2012), while data on smaller waste streams can be obtained from specialized companies or stakeholders (Ebner, 2013; Kletzmayer, 2012; Obersteiner & Scherhauser, 2008; Stadtschreiber, 2005; Vinyl2010, 2011).

In MFA, data from various and potentially very different sources (such as measurements, reports, and scientific papers) are gathered to complete the total balance of the studied system. These sources can have varying qualities, and thus differing associated uncertainties, arising because of e.g. statistical variation, variability, subjective judgement, and approximations (Laner et al., 2014). In this study, the approach by Laner et al. (2015) is used to quantify this uncertainty transparently and uniformly.

Results and Discussion

The material flow analysis is computed using STAN, and a Sankey flow scheme of the plastics budget in Austria for 2010 is produced. This flow scheme is presented in Figure 1.

In total, the chemical industry in Austria produced 1,112 kilotonne (kt, 1×10^6 kg) of primary plastics in 2010, consisting of 877 kt polyolefins, 175 kt polystyrenes and 60 kt resins. An extra net import of 24 kt polymers, 47 kt rubbers, an export of 17 kt additives and the addition of 160 kt regranulates results in a total of 1,326 kt primary plastics available for the manufacturing of semi-finished and final products.

This amount is reduced by a net export of 66 kt semi-finished products and a production waste of 127 kt, while 35 kt additives are added to achieve the desired properties of the final product. A total of 1,167 kt final plastics products is thus brought onto the consumption market.

The relatively low uncertainty ranges for these values indicate that the data sources are relatively reliable. The exception are the additives, for which rather little information is available due to confidentiality, as these substances have a large impact on the properties of the product. Furthermore, wide ranges of plastics contents for the semi-finished products increase the associated uncertainty as well.

The domestic production of plastic products is supplemented by imports and exports. Two consumption sectors report a net export balance: packaging with 74 kt and non-plastics with 49 kt. The other sectors have a net import, amounting in total to 296 kt. The resulting total amount of plastic products brought on the Austrian market is distributed to the consumption sectors according to Lindner (2102). Here, packaging (24%), non-plastics products (20%), building & construction (18%), and others (13%), are the main sectors, amounting to around 75% in total.

The packaging sector consists of plastic applications with a short lifetime, as opposed to the building and construction sector, where products such as piping and isolation materials have lifetimes of several decades, as reflected by the high increase in the stock of this sector. The sector of non-plastic products contains among others lacquers, adhesives and colouring pigments on a polymer basis. These are not recyclable, and are thus not relevant with respect to waste management, so this sector is not further considered and modelled as an export flow.

The outputs of the consumption sectors are waste flows covered by the aggregated collection and sorting process. Due to the large input into the consumption process and the short lifetimes, the packaging sector produces by far the largest waste stream of all consumption sectors. It is clear as well that sectors with an established strong regulation, such as packaging, transport and electronics, have the smallest associated uncertainty, as substantial documentation of the waste flows are required to be collected by the waste management stakeholders.

For the treatment of the plastics waste stream, six main categories of waste management processes can be distinguished. First, mechanical recycling produces regranulate, which is returned to the manufacturing and preparation stage to be used to produce new plastic products. This recovery accounts for 21% of the total treated waste stream. Next, about 5 kt (1%) of waste plastic products (mainly textiles) is reused. Around 10% of the waste flow is used as a reducing

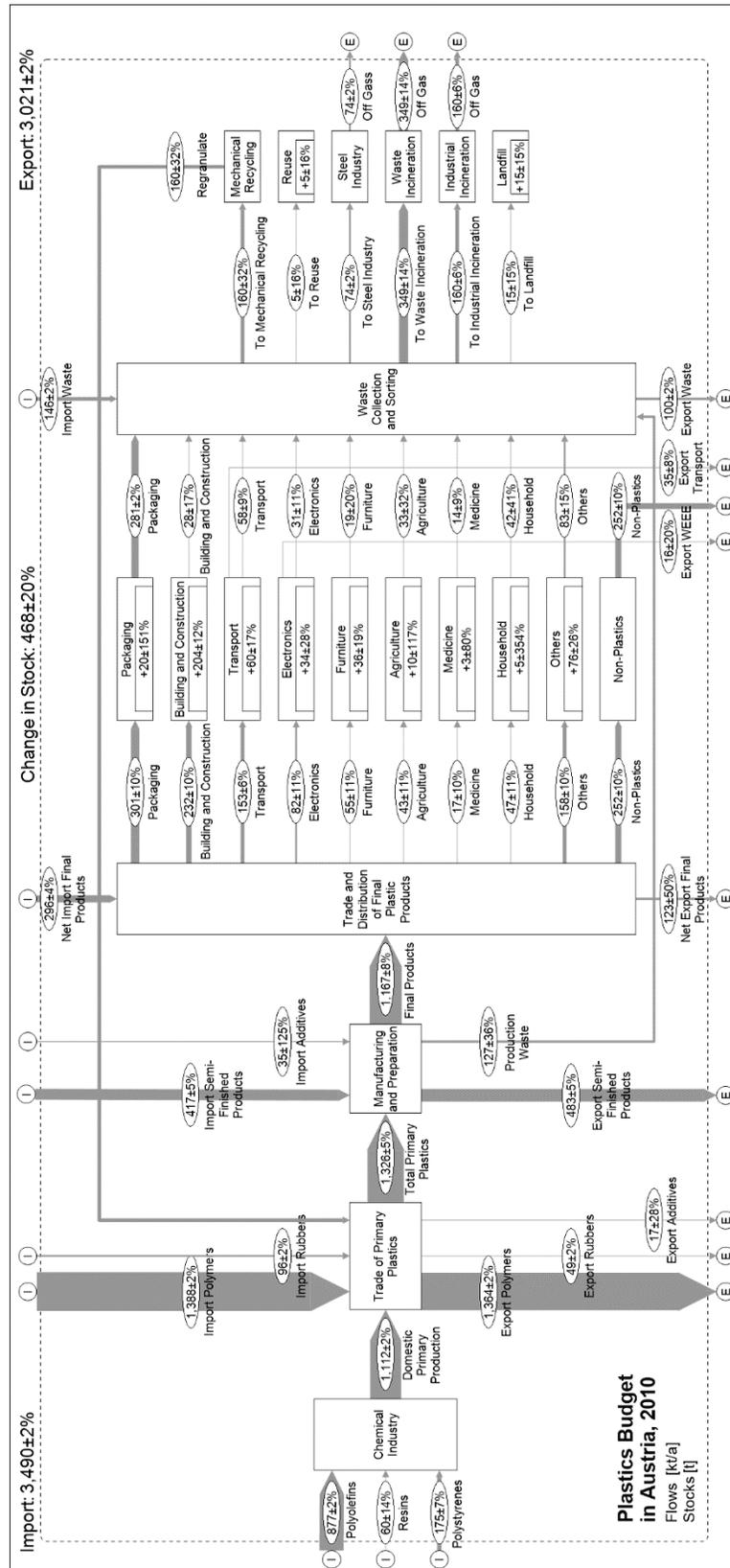


Figure 5: Flow scheme of the plastics budget in Austria for 2010

agent in the steel industry, thus replacing conventional fossil fuels, so this process can be considered as feedstock recycling. Roughly 46% of the waste plastics ends up in the municipal solid waste and is incinerated with energy recovery, whereas around 21% is used in industrial incineration as refuse-derived fuel in the cement industry. Finally, the remaining 2% is landfilled.

These kinds of results can be used for multiple objectives. Policy changes can be highlighted with successive investigations. One illustration is the share of waste plastics which is landfilled. Similar studies on the plastics household in Austria were made by Fehringer and Brunner (1997) and Bogucka and Brunner (2007) for the years 1994 and 2004 respectively. From these results, the effect of the landfill ban in 2004 is clear, as the amount of waste plastics landfilled went down from around 80% in 1994 (Fehringer & Brunner, 1997), over around 30% in 2004 (Bogucka & Brunner, 2007), to the 2% obtained for 2010 in this study.

Furthermore, making a full overview of the total flows of plastics in a region highlights the most relevant streams, which can help to focus time and resources on the main processes or sectors. Especially for waste management, it is crucial to know what kind of waste streams are generated, on a qualitative and quantitative basis. In connection with the stocks and lifespans of the various products, predictions for future waste quantities can be made, and efforts can thus be focused on the largest and most relevant (ecologically and economically) waste streams.

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PLASTIC ADDITIVES TO BE CONSIDERED WITH RESPECT TO THEIR APPROPRIATE FINAL SINKS IN SUSTAINABLE RESOURCE MANAGEMENT - THE CASE OF PBDES

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Introduction

Waste management has gone through a tremendous transition in the last decade. Today, one of the key questions is which valuable materials are contained in waste. This question should never stand alone, but should be accompanied by a second question: which substances are present in the material of interest that pose a threat to human health, to the environment, or to a recycling process and the resulting products?

The concern about contaminants applies in particular to waste plastic. This waste category grows the fastest (40 kg/c. in year 1980, 104 kg/c. in 2006, and 136 kg/c. in 2015 in Western Europe, calculated based on BASF 2007) which results in large stocks (>1000 kg/c.y) of long living goods in constructions, automobiles, consumer products and the like that will turn into waste 10 to 40 years after production.

This stock contains on one hand valuable polymers which are at present in the centre of public attention. On the other hand, it comprises organic and inorganic additives such as stabilizers, flame retardants, plasticizers and pigments that – in part – are hazardous and already outlawed when they turn into waste. Concentrations exceeding allowed values were observed in various streams: Plastics from European markets such as food-contact articles, children toys, or video tapes (see e.g. Samsonek and Puype 2013; Chen et al. 2009; Tønning et al. 2010).

This poster focuses at a single group of organic additives – polybrominated dipnelyl ethers (PBDEs). It discusses the flows of selected PBDEs and problems associated with these flows in a central European context. Possible next steps are also listed.

PBDEs: Identified problems and possible solutions

- (1) Various cases have been documented, where PBDEs are brought back to the consumption phase via polymer recycling.

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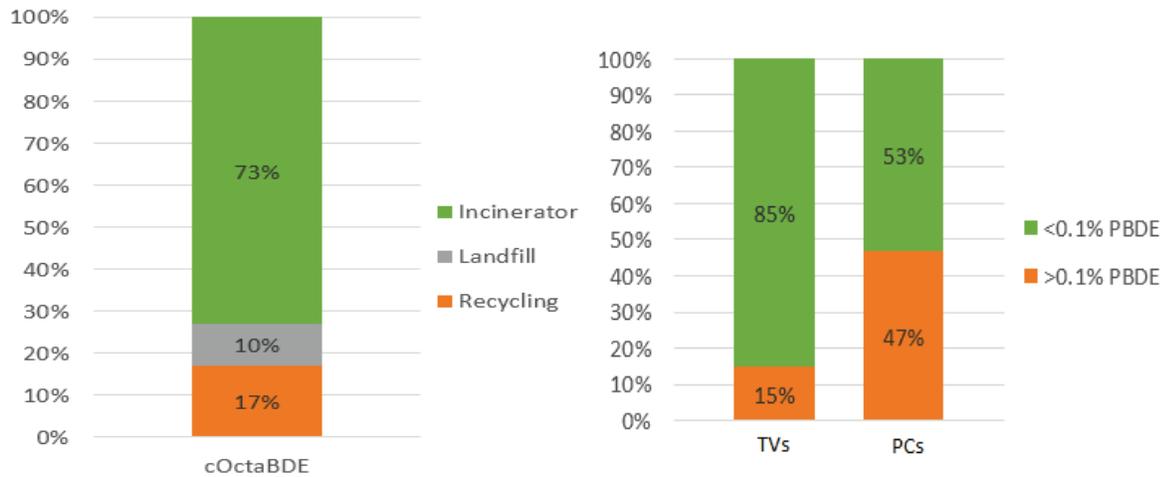


Figure 1 Acceptable and not acceptable flows of PBDEs in Austria. Based on and Vyzinkarova and Brunner, 2013 (1a - left) and Aldrian and Plomberger, 2012 (1b - right)

The Fig. 1 gives some indices of to which extent PBDEs return into the use phase but a clear and holistic overview of PBDE flows in plastic recycling is missing. In a sampling campaign of polymers originating from 3000 TVs and 1600 PCs, 15% of the polymers from TVs and 47% of the polymers from PCs contained significantly higher concentrations of PBDEs than the allowed threshold value of 0.1% defined in the EU's RoHS Directive (1a). Using a substance flow analysis (SFA) to determine sources, pathways and sinks of PBDEs in the city of Vienna, it was calculated that about 17% of octabromodiphenyl ether returned from waste management back to the consumption (1b).

(2) Separation of the contaminated from the clean plastics

The uncontaminated plastics should be recycled and the contaminated plastics should be removed from the cycle.

(3) Scarce information about the fate of PBDEs in European recycling plants

One exception is a mechanical recycling plant in Switzerland, in which the transfers of PBDEs into the output fractions were determined (Morf et al. 2005). There is a need for regular monitoring of PBDE flows in recycling plants (BMLFUW 2013).

(4) Direct the contaminated plastics into an appropriate final sink

In a developed region the appropriate final sink is a state-of-the-art municipal solid waste incinerator, where PBDEs are transformed into harmless substances.

(5) Need for better data about PBDE concentrations in wastes and various other matrices

Material and substance flow analyses (MFA/SFA) are valuable tools to track PBDEs from their source into sink. Currently, the concentration range of PBDEs in various polymer materials is very broad. More chemical analyses are needed in order to narrow down this range. This will allow for more accurate MFA/SFAs.

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