Dissertation

Evaluierung gekoppelter Ressourcensysteme am Beispiel des österreichischen Phosphor- und Stickstoff Haushaltes

ausgeführt zum Zwecke der Erlangung des akademischen Grades eines
Doktors der technischen Wissenschaften
eingereicht an der Technischen Universität Wien, Fakultät für Bauingenieurwesen

als Teil des
Doktoratskolleg für Water Resource Systems

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Wien, im September 2019

Vienna Doctoral Programme on Water Resource Systems
www.waterresources.at
FWF Der Wissenschaftsfonds.

Doctoral Thesis

Evaluation of coupled resource systems using the Austrian phosphorus and nitrogen budget as an example

submitted in satisfaction of the requirements for the degree of
Doctor of Science in Civil Engineering
of the TU Wien, Faculty of Civil Engineering

as part of the
Vienna Doctoral Programme on Water Resource Systems

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Kurzfassung

Der beträchtliche Anstieg des Ressourcenverbrauchs im Laufe des vergangenen Jahrhunderts und die damit verbundenen Umweltprobleme, haben sich zur einer der größten globalen Herausforderungen unseres Zeitalters entwickelt. Die entscheidende Rolle, die Wechselwirkungen zwischen Ressourcen für die Bemühungen um nachhaltigere Produktions- und Konsummuster spielen, wird zunehmend erkannt. Trotzdem fokussieren Materialflussanalysen (MFAs) derzeit häufig auf eine einzelne Substanz oder differenzieren nur sehr grob zwischen verschiedenen Materialien.

In der vorliegenden Dissertation wird eine Methode zur simultanen MFA mehrerer Substanzen entwickelt, wobei der Güterlayer, der die Gesamtmasse jedes im System befindlichen Flusses angibt, als verbindendes Element dient. Die Methode wird anhand einer Fallstudie des österreichischen Phosphor- (P) und Stickstoff- (N) Systems getestet. P und N sind ein hervorragendes Beispiel für zwei stark miteinander verwobene Stoffe und spiegeln die Herausforderungen nachhaltigen Ressourcenmanagements gut wider: Einerseits sind sowohl P als auch N essenzielle Nährstoffe und deshalb für die Sicherstellung globaler Ernährungssicherheit unerlässlich. Emissionen von P und N sind jedoch einer der Hauptgründe für aquatische und terrestrische Eutrophierung und können, im Fall von N auch zu Luftverschmutzung und Klimawandel beitragen. Während Bedenken zur Ressourcenknappheit und Qualität von P zunehmen, weist die Gewinnung von N-Dünger aus atmosphärischem N_2 mittels Haber-Bosch Prozess einen äußerst hohen Energieverbrauch auf.

Die gekoppelte MFA bestätigt die engen Verbindungen zwischen dem P und N System in Österreich. Darüber hinaus zeigt eine Analyse verschiedener Maßnahmen, die von der derzeitige weitgehend lineare Nutzung hin zu mehr Kreislaufwirtschaft und Nachhaltigkeit führen sollen, zahlreiche Synergien und Zielkonflikte auf. Diese treten sowohl zwischen den beiden Nährstoffen, als auch zwischen einzelnen Maßnahmen auf, wobei Synergien deutlich überwiegen. Durch die Kopplung steigt die Modellkomplexität jedoch stark an; die Detailliertheit in der natürliche und industrielle Prozesse dargestellt werden, gewährleistet zwar eine gute Repräsentativität der Wirklichkeit, beeinträchtigt aber die Vergleichbarkeit zu anderen Studien, sowie die Erweiterung des Modells auf weitere Substanzen.

Deshalb wird in einem zweiten Schritt eine mehrschichtige MFA-Struktur entwickelt, die auf der obersten Ebene einen gemeinsamen Referenzrahmen für die Analyse verschiedener Regionen, Substanzen und Skalen bereitstellt und weitergehende Detaillierung in Subsystemen ermöglicht. Der größte Vorteil dieser Struktur liegt in dem hohen Grad an Transparenz. Die Zuordnung von Flüssen und Prozessen realer, komplexer Systeme zu den aggregierten Flüssen und Systemen des generischen Systems erfordert immer subjektive Entscheidungen. Besonders in Systemen wie dem in dieser Dissertation untersuchten P-N Haushalt in Österreich, in denen Biomasseflüsse eine bedeutende Rolle spielen, können mehrere gleichwertige Interpretationen bezüglich der Unterscheidung von Kuppelprodukten und Nebenprodukten, sowie zwischen natürlichen und industriellen Prozessen bestehen. Solche unvermeidbaren Mehrdeutigkeiten treten auch bei anderen Evaluierungsmethoden wie der Ökobilanz oder Kreislaufwirtschafts-Indikatoren auf, in der mehrschichtigen MFA-Struktur treten sie jedoch explizit zu Tage, was die Systemvergleichbarkeit erhöht.

MFA stellt nur den ersten Schritt in der Evaluierung von Ressourcensystemen dar; um den Grad der Nachhaltigkeit der derzeitigen Ressourcennutzung zu beurteilen oder adäquate Maßnahmen für künftiges Management zu entwickeln, muss sie von einer Auswertungs- oder Designphase

begleitet werden. Dafür stehen bereits ausgereifte Methoden wie die Ökobilanzierung oder Umweltverträglichkeitsstudien zur Verfügung und werden beständig weiterentwickelt. Diese Methoden bedürfen jedoch großer Mengen an Daten und Ressourcen, weshalb einfache und anschauliche Indikatoren für überblicksmäßige Bewertungen und öffentliche Kommunikation oft bevorzugt werden. Zwei solcher Indikatoren werden im dritten Teil der Dissertation eingehender untersucht: Circularity (C), die den Anteil des Recycling am gesamten Systemdurchsatz quantifiziert, und Substance Concentration Efficiency (SCE) ein auf der statistischen Entropie (H) beruhender Indikator, der den Grad, zu dem eine Substanz im System konzentriert oder verdünnt wird, misst. Die Analyse wird wieder am Fallbeispiel des österreichischen P-N Systems durchgeführt.

Auch in dieser Untersuchung stellt die Definition einer gemeinsamen Referenzgrundlage einen entscheidenden Schritt dar. Beide Indikatoren weisen die höchsten Verbesserungspotentiale bei einer Kombination verschiedener Maßnahmen auf. Während jedoch Maßnahmen, die auf veränderte Konsumgewohnheiten und Emissionsreduktionen abzielen, in Bezug auf SCE effektiver erscheinen, wird bei C Recycling stärker gewichtet. Aufgrund der besseren Erfassung von Dematerialisierung und Änderungen in der Prozesseffizienz, scheint SCE für die meisten Anwendungsfälle der aussagekräftigere Indikator zu sein. Trotzdem sollten die Grenzen eines einfachen, überblicksmäßigen Indikators darin, die Nachhaltigkeit des Systems ganzheitlich widerzuspiegeln, nie außer Acht gelassen werden.

Abschließend kann festgehalten werden, dass es für die in dieser Dissertation vorgestellte gekoppelte MFA auf Basis eines generischen Referenzrahmens zwei Anwendungsbereiche gibt: Einerseits kann sie in eine ganzheitliche Bewertung, die neben physischen Masseflüssen auch Umweltauswirkungen, sowie ökonomische und soziale Aspekte miteinbezieht, integriert werden. Andererseits kann sie für erste, indikative Bewertungen ohne hohen Daten- und Ressourcenaufwand genutzt werden. In beiden Fällen kann sie wertvolle Einblicke in die Wechselwirkungen zwischen verschiedenen Ressourcen liefern und zum besseres Verständnis nachhaltiger Managementpraktiken beitragen.

Abstract

The tremendous increase in resource consumption over the past century and the environmental problems it entails has become one of the main global challenges of our times. In the efforts towards more sustainable production and consumption patterns the crucial role of resource interactions is increasingly recognized. Nevertheless, to date Material flow analyses (MFAs) still often focus on a specific substance or distinguish between different materials on a very coarse basis only.

In the current PhD thesis a method for simultaneous MFA of several substances is developed using the goods layer, which indicates the total mass of each flow within the system, as the coupling vehicle. This method is tested on a case study of the Austrian phosphorus (P) and nitrogen (N) system. P and N constitute an excellent example for two tightly interwoven substances and reflect the challenges of sustainable resource management well: On the one hand both are essential nutrients and therefore crucial for ensuring global food security. However, emissions of P and N are a main cause for aquatic and terrestrial eutrophication and, in the case of N, may also contribute to air pollution and climate change. While P is a scarce resource and concerns about future availability and quality are rising, the production of N fertilizer from atmospheric N_2 via the Haber-Bosch process is very energy intensive.

The coupled MFA confirms the close links between the Austrian P and N systems. Furthermore an analysis of several policy actions to move from the current largely linear to a more circular and sustainable resource use reveals numerous synergies and trade-offs both between the two nutrients and between the individual measures, where the former clearly outweigh the latter. However, coupling significantly raises model complexity and while the level of detail on which natural and industrial processes are shown ensures good representation of reality, it hampers comparability to other studies and model extension to additional substances.

Therefore, in a second step, a multi-level MFA structure is developed that provides a common reference framework for analysis of different regions, substances and scales on the top-layer and enables detailed representation of processes in sub-systems. Its high level of transparency proves to be the main advantage of the generic structure. Allocating flows and processes of a real, complex system to the aggregated flows and processes of the generic one is always requires subjective choices. Especially in systems like the Austrian P-N system studied in this thesis, in which biomass flows play an important role, multiple equally valid interpretations regarding the distinction between co-products and by-products or between natural and industrial processes may be possible. Such unavoidable ambiguities also exist in other assessment method such as life cycle assessment or circular economy indicators; however, the multi-level MFA structure developed in this thesis makes them more explicit and thus enhances system comparability.

MFA only constitutes the first step in resource system assessment; in order to evaluate the state of sustainability of current resource use or to develop adequate policy measures for future management, it has to be followed by an evaluation or design step. Several sophisticated methods such as life cycle assessment or environmental impact assessment exist for this purpose and are constantly being developed further. These methods are highly data and resource intensive though, which is why more simple and illustrative indicators are often preferred for preliminary assessments and public communication. Two such indicators are analyzed in the third part of this thesis: Circularity (C), which quantifies the share of recycling on total system throughput

and Substance Concentration Efficiency, an indicator based on statistical entropy (H), which measures the extent to which a substance is concentrated or diluted within a system. Once again, the Austrian P-N system is used as a case study.

Also in this analysis, the definition of a common reference basis constitutes a crucial step. Both indicators show highest improvement potentials for a combination of different measures; however, while for SCE measures aiming at changed consumption behavior and reducing emissions seem to be more effective, recycling efforts are valued higher in C. Due to the fact that it captures effects of dematerialization and process efficiency, SCE seems to be the more meaningful indicator for most applications. Nevertheless, the limits of a simple, preliminary indicator in representing full system sustainability should always be kept in mind.

In conclusion, coupled MFA based on a common reference framework as presented in this thesis can be applied in two ways: it can be integrated in more holistic assessments, which, apart from physical mass flows, also take environmental impacts, as well as economic and social aspects into account, or it can serve as a basis for coarse, indicative, low data- and resource-demanding assessments. In both cases, it can provide valuable insights in resource interactions and thus enhance understanding of sustainable management practices.

Acknowledgements

I would like to express my gratitude to the people who supported me and my work over the past years.

A special thanks goes to my supervisor Prof. Helmut Rechberger, who initiated my PhD and thus gave me the opportunity to work in the exciting and interdisciplinary environment of the Vienna Doctoral Programme on Water Resource System and who always supported me with his extensive expertise and experience, prompt help, constructive feedback and pragmatic approach to any difficulties arising in the course of the work.

I would also like to thank Ottavia Zoboli for sharing her data and experience with me and for her great support especially in the initial phase of my thesis, Prof. Matthias Zessner for his steady helpfulness and expertise in the field of water quality issues and Oliver Cencic for his prompt support and advice on all issues concerning the STAN software and uncertainty assessment.

Furthermore, I would also like to express my gratitude to my external examiners xx and xx for their time and expertise during the revision of this doctoral thesis.

The financial support for this work was provided by the Austrian Science Funds FWF (DK-plus W1219-N22), which I gratefully acknowledge.

Last but not least, many thanks go to all my colleagues of the Doctoral Programme, the Institute of Hydrology and Water Resource Management and the Institute of Water Quality and Resource Management for the great working atmosphere, the fruitful discussions and for sharing their knowledge and experience with me.

Published articles and author's contributions

Results of the present doctoral thesis have also been published in (or submitted to) peer-reviewed scientific journals in the form of the following three articles:

J. Tanzer et al. (2018). "Filling two needs with one deed: Potentials to simultaneously improve phosphorus and nitrogen management in Austria as an example for coupled resource management systems". In: *Science of the Total Environment* 640-641C, pp. 894–907. DOI: 10.1016/j.scitotenv.2018.05.177I

J. Tanzer and H. Rechberger (2019). "Setting the Common Ground : A Generic Framework for Material Flow Analysis of Complex Systems". In: *Recycling* 4.23. DOI: 10.3390/recycling4020023

J. Tanzer and H. Rechberger."Complex systems, simple indicators: assessing resource system sustainability with statistical entropy and circularity". Submitted to: *Science of the Total Environment*.

Author's contributions:
In all three papers formal analysis, investigation, visualization and writing - original draft preparation were undertaken by the author of the present thesis.
Conceptualization, methodology, data curation, validation and writing - review and editing were performed in cooperation with the respective co-authors.

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Chapter 1

Motivation and Outline

The transition of the world from an agricultural to an industrial society over the past century, which brought about a quadrupling of global population and a 5-fold increase in world GDP/cap, came at the expense of an explosion of global resource use. Extraction of natural materials has tripled over the last 50 years and growth rates are accelerating steadily (Oberle et al. 2019). This is linked to a number of problems, such as pressures on and damages to the environment, resource scarcity and distributional conflicts, and therefore poses a serious threat to global sustainability (Krausmann et al. 2009). Moreover, while the world has made progress on mitigating some of the other negative side-effects of industrialization, such as air pollution and acid rain caused by sulphur emissions (Stern 2005) or the depletion of the Antarctic ozone layer (Hand 2016), material consumption shows no signs of stabilization or reduction. Material productivity improved only in parts of the developed world and was partly achieved by outsourcing production processes to other regions, whereas globally material productivity even declined in the course of the 21st century (Oberle et al. 2019). As the impacts of excessive resource use such as climate change, biodiversity loss and water stress become more and more noticeable, concerns about future livelihood are rising and the need for a shift towards more sustainable production and consumption patterns becomes undisputable.

As a consequence, concepts to reconcile human and environmental needs such as sustainable development or circular economy (CE) have evolved. Despite their broad adoption into national, international and corporate policy around the world (e.g. Deegan and Islam 2012; United Nations General Assembly 2015; President of the People's Republic of China 2008; Ministry of the Environment Government of Japan (MEGJ) 2000; European Commission 2015), the state of the environment continues to decline though. CE for instance to date perfectly complies with the characteristics of an umbrella concept in the stage of validity challenge (Blomsma and Brennan 2017): Some praise its potential to reduce environmental impacts of consumption processes (Blok et al. 2016), spur the economy (Beasley and Georgeson 2014) and significantly contribute to the achievement of global sustainability (Schroeder et al. 2018), whereas others regard it mainly as a re-labelling of long existing practices and/or criticize its failure to account for rebound effects, quality rather than quantity of recycling, thermodynamic entropy and market mechanisms (Zink and Geyer 2017; Cullen 2017; Korhonen et al. 2018; Geyer et al. 2016). Similarly, the concept of sustainable development has been criticized as being too vague, heterogeneous and complex (Lélé 1991; Engelman 2013) and thus is claimed to have started to lose traction (Brande et al. 2011; Kirchherr et al. 2017).The main advantage of both concepts, their broad applicability on variable scales (from global economy-wide to single firm assessment), in different contexts and for numerous materials and products, is also their main drawback as it has led to the coexistence of a wide range of definitions of terms and indicators for their assessment. Different institutions and academic bodies have sometimes even assigned the same name to differently measured indicators or designated identical measures differently. For instance, the ratio of recycled end of life waste to total material demand is termed α by Cullen (2017) and Use Rate of Recovered Used Products by Hashimoto and Moriguchi (2004). Recycling rates, on the other hand, may mean something different for different materials, even if reported by the same institution (Haupt et al. 2017). This situation might tempt businesses and public organizations to “cherry-pick” weak indicators purporting great progress without requiring real structural changes (Geyer et al. 2016; Bocken et al. 2017), eventually causing the collapse of the concept.

Following the proverb "What gets measured gets managed", success towards sustainable resource management will therefore crucially depend on our ability to harmonize definitions and indicators across substances, regions and scales. Efforts in this direction in the scientific community have increased over the past years (Geissdoerfer et al. 2017; Ghisellini et al. 2016; Kalmykova et al. 2018; Kirchherr et al. 2017; Merli et al. 2018; Murray et al. 2017; Elia et al. 2017; Parchomenko et al. 2019; Pauliuk 2018; Saidani et al. 2019); however, due to the multitude of contexts under which CE can be studied, no universally applicable assessment method has evolved to date.

The fact that resource systems are not independent of each other, but, to the contrary, exhibit numerous complex interdependencies, is another main challenge in the efforts towards sustainable resource management. Changes in the management of one resource will inevitably affect other (environmental and anthropogenic) systems and only if linkages and feedback mechanisms are understood, can undesired effects be avoided and synergies exploited (Liang et al. 2019). The crucial role of such resource nexuses is increasingly recognized in scientific assessments (e.g.Giurco et al. 2014; Font Vivanco et al. 2018; Liang et al. 2019).

In the present thesis, both of these challenges will be addressed.

Using the Austrian phosphorus (P) and nitrogen (N) system as an example, interrelations, synergies and trade-offs between two highly interlinked substances are studied. Both nutrients are closely involved in the water-food-energy nexus, which is among the currently most discussed. P and N are associated with a number of environmental problems, among them eutrophication, groundwater contamination, climate change and air pollution, which are described in detail in Chapter 2. In fact, in the concept of planetary boundaries, anthropogenic perturbation of the global P and N cycles are associated with the highest risk of pushing the Earth out its safe operating space (Steffen et al. 2015).

Material flow analysis (MFA) is chosen as the principal method for the analysis. Building on the principle of mass conservation, MFA quantifies the mass flows and stocks of goods or substances within a given system. Thus, it provides transparency for the pathways materials take in ecosystems, human societies, and industrial processes and enables early recognition of critical depletions or accumulations of material, as well as environmental loadings. The idea of mass conservation dates back to ancient Greek philosophy and was applied in a systematic way in the 1930s by Wassily W. Leontief in form of the economic input-output analysis (Brunner and Rechberger 2017). Studies on metabolism of cities and specific industries followed in the 1960s and since the mid-1990s MFA has become one of the most popular tools in resource management assessment (Bai et al. 2015). Two main types of analysis can be distinguished: MFA (in a narrow sense) deals with the study of goods or groups of goods. Goods in this perspective are defined as tradeable assets, consisting of one or more substances (e.g. heating oil, mineral water, waste). Mass flows of goods are typically reported in national or regional statistics. For the assessment of resource criticality or environmental impacts focus on the mass flows of a particular chemical element or compound (i.e. a substance) may be more relevant, though. These studies are known as substance flow analysis (SFA). As "material" serves as an umbrella term of both substances and goods, "MFA" (in a broad sense) can be used as an umbrella term for both analyses on the level of goods and substances.

However, to date distinction of material groups in MFA is only undertaken on a very coarse basis, such as fossil fuels, metals, construction minerals and biomass (e.g. Fischer-Kowalski et al. 2011; Mayer et al. 2018) and can therefore not capture substance specific issues of resource criticality and environmental impacts. SFA on the other hand mostly consider one substance only and hence disregard resource interactions. Even if multiple substances within a system are studied, each substance is treated individually (e.g. Antikainen et al. 2005; Coppens et al. 2016; Ma et al. 2013; Thaler et al. 2015). In contrast, in Chapter 3 a method for coupled MFA for the Austrian P and N system, focusing on substance interrelations is presented. State (Egle et al. 2014), historical trends (Zoboli et al. 2015), and potentials to support sustainable management (Zoboli et al. 2016) have been thoroughly studied for the Austrian P system. Chapter 3 examines, how the measures that proofed most effective for P management affect the N system and highlights synergies and trade-offs between the nutrients, as well as when combining different measures with each other. Impacts of coupling on model structure and complexity are also investigated. Thus, the analysis should not only reveal co-benefits and conflicting goals between the two substances, but also, from a methodological viewpoint, foster understanding of whether and how a coupled, multi-substance MFA can generate more robust and meaningful results compared to a mono-substance one. In this respect the coupled MFA of P and N should serve as a case study for coupled resource systems in general.

While Chapter 3 provides valuable and detailed insights into interactions in the Austrian P-N system, it does not address the second challenge of resource management, which is harmonizing resource assessments across substances, regions and scales. The model structure is highly complex and customized for this specific case study. This may complicate the extension of the study to other resources and hamper comparability to other studies, a problem not only pertaining to the present case: The multitude of indicators to assess the level of resource efficiency and circularity that exist alongside each other (e.g. Elia et al. 2017; Parchomenko et al. 2019; Pauliuk 2018; Saidani et al. 2019; Avdiushchenko 2018) points to the fact that transferring indicators to studies with different model structure or purpose is not always straightforward. Real systems are complex and specific, whereas, usually indicators are generally defined, even under similar circumstances, and therefore room for interpretation remains or slight adjustments have to be made. For instance, when calculating the cyclical use rate for the Czech Republic, an indicator developed by the Japanese government for CE assessment (Government of Japan 2013), Kovanda (2014) had to undergo several alterations to the original definition in order to ensure compatibility with local conditions. Núñez-Cacho et al. (2018) recently developed a model for the transition of family businesses to CE, which proves that even differences in factors not predominately associated with CE, such as the ownership structure of a business, may be highly relevant for the outcome of a CE study. Avdiushchenko (2018) found that the CE monitoring framework of the European Union, which was developed for implementation at the member states level, was inappropriate for capturing CE effects at the regional and local level. Likewise, problems were noted when transferring monitoring frameworks between China and the EU and also between different regions within the EU, due to different perceptions of CE priorities and indicators which are highly adapted to local geographic, environmental, economic, and social circumstances. Ordóñez et al. (2019) encountered similar problems on a micro level, noting that "the lack of comparable inventory systems of all included (urban reuse and remanufacture) initiatives limited the ability to quantitatively analyze environmental impact and economic viability". Rahman et al. (2019) identified variable system boundaries and levels of detail, due to different management regimes and data availability, as a main challenge when comparing P recycling efficiency in the waste sector of different countries. Therefore, common reference frameworks are essential for the comparability of CE indicators under different circumstances.

Apart from ensuring comparability across substances, regions, and scales, a common reference system may facilitate the development of universally applicable indicators, increase understanding with political and economic decision makers, and therefore strengthen the concept of CE as a whole. Moreover, if CE assessments are based on a common framework, differences in focus become more visible so that mutual learning and broadening of perceptions is encouraged.

For economy-wide MFA, such schemes and indicator definitions have been available and developed for a long time (Eurostat 2001; Fischer-Kowalski et al. 2011; Haas et al. 2015; Matthews et al. 2000; Mayer et al. 2018), and recently, Pauliuk (2018) has proposed a general MFA system at the organizational and product life cycle level. Because they depict the system at a very coarse level, fitting complex systems to this structure remains difficult and would require substantial simplification and thus loss of informative value. On the other hand, comprehensive templates for SFA have, among others, been developed for national P balances (Chowdhury et al. 2014; Jedelhauser and Binder 2015; Seyhan 2009), but are not intended for model extensions. Rahman et al. (2019) also solved the problem of heterogeneous definitions of the waste sector by aggregating flows and processes to a baseline/standard system. However, when calculating one of the more general CE indicators for such a system, it was not always clear what flows it should be referred to. Moreover, the importance of resource interactions for the achievement of holistic sustainable solutions is increasingly recognized (Font Vivanco et al. 2018; Giurco et al. 2014; Liang et al. 2019) so that assessment frameworks that are applicable to a multitude of substances are gaining in importance.

The aim of Chapter 4 is to develop a framework structure that addresses all of these needs simultaneously. Particular emphasize is given to the transformation process and the implication assumptions that need to be taken therein have on assessment results.

Yet, even if conducted in a generic setting, MFA only constitutes a first step in resource assessment; in order to support the development of policy measures to enhance resource sustainability it has to be followed by an interpretation or design step (Bai et al. 2015). As implied above, a common approach is the calculation of indicators through which MFA results can be transformed into a single value and studies pertaining to different regions or substances made comparable.

Life Cycle Assessment (LCA) (International Organization for Standardization 2006) and Environmental Impact Assessment (United Nations Economic Commission for Europe 2017) are perhaps the most well-known tools for the integrative evaluation of the environmental impacts of products and processes and of projects, respectively. Both methods are continuously evolving and may, in adapted forms such as life cycle sustainability assessment (Valdivia et al. 2013), be extended to economic and social domains.

However, in national and international policy comparatively simple indicators such as circularity- or recycling rates predominate. The limits of such indicators are currently being intensively discussed within the scientific community: they are claimed to be ill-defined, neglecting quality aspects of recycling, rebound effects, or economic market mechanisms and opening the door to tokenistic action that do not require real structural changes (Bocken et al. 2017; Cullen 2017; Geyer et al. 2016; Haupt et al. 2017; Korhonen et al. 2018; Zink and Geyer 2017). Suggestions for more meaningful resource efficiency- and CE indicators are mushrooming (e.g. Ellen MacArthur Foundation and Granta Design 2015; Figge et al. 2018; Huysman et al. 2017; Linder et al. 2017; Mayer et al. 2018) and new insights have reached policy makers. For instance, measurements for recycling rates have recently been harmonized across the European Union and defined in relation to waste that enters the recycling operation (European Union 2018), making the potential for improvements in resource efficiency more explicit compared to formerly applied collection rates (Haupt et al. 2017).

Yet, policy development usually requires cooperation of multiple stakeholders from different fields and not least the general public. In the Austrian P-N system for instance, changes in consumption behavior such as a shift to a less meat-reliant diet or more efficient application of compost in private gardens were identified as one of the most effective measures to reduce import dependency and emissions of P and N (Thaler et al. 2015; Zoboli et al. 2016). Moreover, policy-making institutions are often constraint in financial, temporal, and human resources available for conducting assessments (Kaufman et al. 2010). Furthermore, lack of sufficient consistent, well-documented and reliable data is a well-known issue in MFA (Laner et al. 2013; Patrício et al. 2015; Wang and Ma 2018). All of this undermines that in addition to holistic assessment methods such as LCA, illustrative, ready-to-use indicators that can directly be calculated from MFA and are intuitively understandable without extensive background knowledge are needed for initial, general assessments and public communication.

Possibly, simple indicators and more holistic assessments may lead to similar conclusions, which makes the former an adequate basis for further in-depth analyses. For instance, Kaufman et al. (2010) showed that cumulative energy demand is representative for results of a full LCA of waste management systems. Nonetheless, careful indicator selection and awareness of its limitations is crucial when dealing with simple, ready-to-use indicators.

Two such indicators, Circularity (C), which quantifies the share of recycling on total system throughput and substance concentration efficiency (SCE), which measures the extent to which a substance is concentrated or diluted within a system, are analyzed in the context of the Austrian P-N system in Chapter 5. The indicators are compared with respect to their scope and focus as well as their potential to reflect overall system sustainability. This should reveal the circumstances under which they provide meaningful results and the potential existence of hidden trade-offs.

Finally, Chapter 6 highlights the conclusions that can be drawn from the present work and provides outlooks for further investigations.

The main aims of this doctoral thesis can thus be summarized as follows:

1. Development of a MFA-based method for coupled analysis of several resources and evaluation of the ability of this method to reveal synergies and trade-offs between management measures of different substances as well as analysis of the impacts of coupling on model complexity

2. Analysis of the potentials to harmonize MFA structures of complex systems across regions, resources and scales to ensure maximum comparability without requiring loss of information in one or several of the systems studied

3. Analysis of the potentials of C and SCE as simple, straightforward indicators to capture overall system sustainability and thus provide first insights or be integrated in more holistic resource system assessments

4. Regarding the case study on the Austrian P and N system, the study should reveal, which measures or group of measures achieve the highest improvements in the efficiency of nutrient management

Chapter 2

The challenge of phosphorus and nitrogen management

2.1 Phosphorus

P is an essential element for all living organisms and often the factor that limits plant growth (Johnston and Steén 2000). Thus, the widespread application of industrial phosphorus fertilizer was crucial to the augmentation of agricultural yields during the green revolution after World War II (Borlaug 1970). Still today growing population numbers and changes in dietary habits call for a continuous increase in agricultural production. Consequently, demand for P fertilizer is constantly growing and expected to do so in the future (Smit et al. 2009; Tenkorang and Lowenberg-Deboer 2009).

However, phosphate rock is a finite resource. Regardless of ongoing discussion about future availability and a potential peak in P production (e.g. Cordell and White 2011; Scholz and Wellmer 2013; Edixhoven et al. 2014), as reserves with pure and easily available P are depleted, extraction becomes more costly and quality of the mined product is declining. Rock phosphates usually contain minor amounts of heavy metals and radioactive substances, traces of which can be found in fertilizers (Kratz et al. 2016). Recently an augmentation in the amount of these undesired by-elements has been noticed (Egle et al. 2016a). Furthermore, P that is currently economically feasible to extract is concentrated in a few countries only, with Morocco accounting for 80% of the known reserves (Jasinski 2016). Most of the reserves are located in geopolitically unstable regions and some countries restrict extraction for strategic reasons, giving rise to strong price fluctuations on the global P markets (Ridder et al. 2012; Cordell and White 2015).

But not only is the procurement of phosphorus problematic, its disposal may be just as critical. Excess P eroded from agricultural soils and contained in untreated wastewater enters surface waters, where it can cause eutrophication: Like in terrestrial ecosystems, P is often a limiting factor in water bodies. Therefore, unintentional "fertilization" may lead to algal blooms, subsequently causing light and oxygen deficiencies or release of toxins during algal decomposition, which may affect fish and other aquatic organisms (Johnston and Steén 2000).

The EU is tackling the problem of eutrophication with the Water Framework Directive and the Common Agricultural Policy. Furthermore, it has included phosphate rock into its list of Critical Raw Materials, substances considered of high importance to the economy and of high risk associated with their supply (Bureau de Recherches Géologiques et Minières et al. 2017). Scientific, political and industrial interest in recycling P from waste and wastewater and thus reducing import dependency has been rapidly growing over the past years. Several studies exploring improvement potentials in P management have been conducted (e.g. Hamilton et al. 2017; Klinglmair et al. 2017; Zoboli et al. 2016) and efforts to harmonize national MFAs to facilitate systematic comparison and transfer of lessons learned have been made (e.g. Jedelhauser and Binder 2015; Dijk et al. 2016). Policies like the German sewage sludge ordinance (Bundesregierung Deutschland 2017) or the proposal for the EU Fertilizer Regulation revision

(European Commission 2016) set legal prerequisites to spur recovery and recycling and respective technologies have been developed (Egle et al. 2016b). As efforts move from a theoretical to a more and more implementational stage, economic and environmental impacts of different measures and technologies are moving into focus as well (e.g. Egle et al. 2016b; Ernst Basler + Partner AG 2017; Jossa and Remy 2015; Hanserud et al. 2017). Nevertheless, P management to date mainly follows a linear production path. Austria is no exception to this rule, as has been shown in a detailed analysis of the national P budget by Egle et al. (2014), Zoboli et al. (2015), and Zoboli et al. (2016).

There is no production of P in Austria; therefore 14 000 t P for domestic consumption are imported each year via phosphate ore and mineral fertilizer. Internal recycling such as composting of organic waste and sewage sludge is only taking place to a limited extent, so that 40-45% of P contained in waste ends up in landfills and cement kilns, from where it can hardly be recovered. Although mineral fertilizer application was almost halved and total P input to agricultural fields reduced by 13% over the past 20 years, the P stock in agricultural soils is still growing with 2300 t P/year. Consequently, P losses to erosion could not be reduced significantly and despite considerable reductions in the wastewater sector emissions to surface water still amount to 3600 t P/year (Zoboli et al. 2015; Zoboli et al. 2016).

The optimization study conducted by Zoboli et al. (2016) revealed some 15 strategies through which national phosphorus management could be improved, making it a prime example for the CE approach. It showed that Austria could substitute its fossil P fertilizer consumption completely with recycled products and identified a reduction potential of 89% and 28% for import dependency and emissions to water bodies respectively. For instance, large amounts of P could be recovered from sewage sludge or meat and bone meal ashes. Similarly, replacing nutrient-rich compost with other materials in landscaping would make large amounts of P available for fertilization purposes. Through changes in agricultural practices such as optimization of P content in animal feed, increasing use efficiency in crop farming and erosion control, reductions in P demand could be achieved in a particularly cost-effective manner. Finally, high reduction potentials also lie in a shift towards a balanced diet, less reliant on meat and dairy products.

2.2 Nitrogen

Being an essential constituent of, among others, amino acids, which are the building blocks of proteins, DNA, protoplasm and chlorophyll, N is also a vital element for life. N is the sixth most abundant element in our universe and constitutes 78 vol.% of the Earth's atmosphere. Nevertheless, N is often a growth limiting factor for plants. N_2 in the atmosphere is virtually inert, whereas with the exception of legumes plants can take up only ionized NH_4^+ and NO_3^- (Banks 1990; Munch and Velthof 2007). The invention of the Haber-Bosch process in the beginning of the 20th century, which enables technical NH_3 synthesis from atmospheric N_2 and H_2, was thus crucial for meeting the rising demand for agricultural products (Galloway et al. 2008).

However, as this process requires high levels of temperature and pressure, it is very energy-intensive. The production of N fertilizers accounts for 1.1% of global energy use and 0.93% of global greenhouse gas emissions (IFA 2009; Dawson and Hilton 2011). Hence, future N supply is hardly an issue of resource scarcity, but rather one of energy availability.

Moreover, N leaching from agricultural fields and discharge of wastewater into lakes and rivers is another main cause of eutrophication. Besides, excess N fertilizer can percolate to the groundwater and thus pollute drinking water resources. NO_3-contaminated drinking water can cause methaemoglobinaemia, an impairment of blood oxygen transport, which is potentially lethal to infants (Munch and Velthof 2007). The EU has therefore set a standard of 50 mg

NO_3/l for groundwater resources in the Nitrate Directive. The fact that exceedance rate of this threshold could be halved since the directive entered into force in 1991 indicates the success of measures to control agricultural NO_3 losses. Nevertheless 13% of European and 7% of Austrian measurement sites still exhibit NH_3 concentrations above 50 mg/l. (European Commission 2002; European Commission 2018; Nationale und internationale Wasserwirtschaft (Abteilung I/3) 2019).

Meanwhile, gaseous emissions of N may prove an even more serious environmental problem in the long run. Agriculture is a major source of atmospheric NH_3 and N_2O (Jenkinson 2001). While N_2O is a potent greenhouse gas with a radiative forcing 264 times higher than CO_2 and an atmospheric life time of 120 years (Myhre et al. 2013), NH_3 reacts with NO_x to form particulate matter and is considered the most important cause of air pollution (Pozzer et al. 2017). Like particulate matter, NO_x emissions on their own can lead to cardiovascular and respiratory diseases; furthermore they play a key role in the formation of tropospheric ozone. Anthropogenic NO_x emissions predominately stem from combustion processes, most notably traffic, and to a lesser extent from (agricultural) soils. Apart from their detrimental effects on human health, NH_3 and NO_x deposition is a major cause for terrestrial eutrophication and the associated decrease in biodiversity (Jenkinson 2001). Moreover, reactive forms of N "cascade", i.e. they can contribute to all of these impacts in sequence as they move through and are transformed in the environment, and have strong interactions with the global carbon cycle (Galloway et al. 2008).

To date, most action to reduce N emissions focuses on the production sector, e.g. the increase of N efficiency on farms and improvements in technology for both stationary combustion sources and vehicles. However, consumer-side based actions such as reducing food waste, lowering human consumption of animal protein and limiting road transport, as well as increasing efforts to recycle N from wastewater systems also show high potentials in increasing sustainability of N management (Shibata et al. 2017).

2.3 Interactions

Given the close connection of the P and N cycle Rockström et al. (2009) consider them as a single domain in their planetary boundary concept. In fact, most flows which play an important role for P recycling, such as manure, meat and bone meal, compost and sewage sludge, are also rich in N (Jeng et al. 2006; Fachbeirat für Bodenfruchtbarkeit und Bodenschutz 2010; Jossa and Remy 2015). This implies that the management aims of both substances can be addressed simultaneously. Similarly, Thaler et al. (2015) showed that by implementing a balanced diet, net imports of P could be reduced by 20-25%, while net imports of N sank by 27-37%. Emissions to surface waters could also be reduced by 5-6% and 11-15% for P and N respectively.

On the other hand, unlike for fossil fertilizers, the amounts of each nutrient and their ratios to each other in recycled products cannot be arbitrarily changed and may therefore not be fully compatible with the plants' requirements. The N/P ratio in meat and bone meal for instance is considerably lower than the ratio by which the elements are taken up by crops. Thus, if the amounts of meat and bone meal applied are determined by the plants' P demand, additional N fertilization will be necessary. Conversely, adjusting meat and bone meal amounts to N requirements will lead to an overfertilization with P (Jeng et al. 2006). A complete substitution of both fossil N and fossil P fertilizer may therefore not be achievable.

Chapter 3

Coupled material flow analysis for the Austrian phosphorus and nitrogen system

3.1 The coupled MFA model

Given the close connections between the P and N system (see Chapter 2.3), it seems reasonable to also address inefficiencies in their management in a simultaneous way. However, to date, MFA of both P and N focus more on the flow patterns of each substance individually than on their interactions, irrespective of whether the analysis is conducted on a sectoral (e.g. Antikainen et al. 2005; Ma et al. 2013; Thaler et al. 2015) or regional (Coppens et al. 2016) basis. To overcome this, here, a method for coupled, multi-substance MFA is developed, following the methodology described in Brunner and Rechberger (2017). The freeware STAN was used (Cencic et al. 2017) to calculate mass balances for each process in the system, perform error propagation of initial data uncertainties and data reconciliation. The later is based on least squares regression and alters a priori input data so that initial contradictions in the mass balance are eliminated. The a priori uncertainty thereby serves as a weighting factor that determines the extent of adjustment for each data element.The model builds on existing work by Zoboli et al. (2016) and Zoboli et al. (2015) who analyzed the improvement potentials of P management in Austria based on a detailed national MFA. Both the current state of P and N management in Austria and the effects of different measure scenarios are modeled.

3.1.1 Status quo

The status quo model depicts the actual situation of P and N management in 2015 and should reveal sustainability and flaws in current nutrient management, as well as serve as a reference state for the evaluation of measure scenarios.

Nine main sectors/processes, relevant for the national P and N management are depicted: Animal husbandry, crop farming, forestry, industry and trade, bioenergy, households and public establishments, wastewater management and waste management. Each of these processes is further described by one or more subsystems, to enable e.g. distinction between food-, timber- and chemical industries or between consumption and soil processes in private households and to show processes such as manure generation and handling in more detail. Exchanges of N and P between these processes in gaseous, liquid and solid form as well as import and exports across country borders are represented by flows of P and N in t/year. In the majority of cases data is present as mass flows of a good (a substances or mixtures of substances with an economic market value (Brunner and Rechberger 2017)) reported in national statistic databases or governmental reports and their respective P and N concentrations (mostly found in scientific literature). However, especially for gaseous and liquid flows, data is sometimes also directly reported as masses of N and P per time. In addition, transfer coefficients and relations are used which link data with linear equations.

The rough concept of the model can be described as follows: P and N are imported into the system as fertilizer, food and feed and are distributed (partly after refinement) by the industry and sector among agriculture and consumers. Animal husbandry uses feed from industry and pastures to produce meat and dairy products; in addition manure is generated. Crop farming consumes mineral and organic fertilizers for food production, which (again via the industry and trade sector) is transmitted to consumers or exported. Waste and wastewater from all sectors are collected in the wastewater and waste management systems respectively. Wastewater is treated in wastewater treatment plants and effluents discharged to water bodies, while emissions from agricultural and natural soils reach these water bodies directly. Depending on its composition, waste is either incinerated, deposited in landfills or treated (e.g. by composting) to generate a recyclable product.

Although model structure, flow calculation and data sources largely correspond to the ones described in Zoboli et al. (2015), the extension of the model to N and the refined consideration of data uncertainty (see Chapter 3.1.3) required several changes.

For instance, processes and flows to account for gaseous N fluxes during incineration processes, manure handling and microbial N fixation in soil, which were not relevant to the national P balance had to be added in the coupled model. Assumptions for the handling of organic waste from private households were revised according to the findings of Maier (2017) and differences in plant nutrient availability of different fertilizers included in the process “Crop farming” (see Table 3.1). The new stock “planetary boundary layer” was introduced, representing exchanges between the atmosphere and other sectors of the model. However, as the focus of the study is on the interaction between N and P, sectors and processes exclusively relevant to the N-system, such as combustion of fossil fuels for power generation and in vehicles, were left aside. Similarly, cross-border exchange of gases was not considered in the balance, due to high uncertainties in its quantification and minor relevance to the issue of this study. Therefore, the inputs to and outputs of the stock “planetary boundary layer” are incomplete, which is why it should not be considered part of the balanced system. Furthermore, a number of fluxes considered in an aggregated form by Zoboli et al. (2016), where split into their components in the present model. For instance, the flow “Crops” was split in a subsystem into 50 individual flows (wheat, potato, carrots, apples, etc.). This made it possible to enter P- and N-concentrations of the various flows into the model in the form they existed in the data sources without requiring intermediate calculation steps and thus reduced distortion during data reconciliation (see Chapter 3.1.3).

Tab. 3.1: P and N plant availability of different fertilizer products as applied in the model. Values are based on Hamilton et al. (2017) and Syers et al. (2010) for P and BMLFUW (2006) and Gutser et al. (2005) for N.

	Plant available P [t/t P input]	**Plant available N [t/t N input]**
Mineral fertilizer	0.70	0.85
Manure	0.67	0.59
Sewage sludge	0.51	0.47
Compost	0.36	0.34
Meat and bone meal	0.28	0.68
Biomass ashes	0.46	0.57
Biomass digestates	0.49	0.55

Overall, the changes described above significantly increased the complexity of the model. Total number of processes rose from 56 to 194, of stocks from eight to 15, of flows from 122 to 866 and of processes with transfer coefficients from two to 55. Figure 3.1 shows the structure of the adjusted system. Each of the processes depicted in the figure consists of one or more sub-systems, the structure of which is shown in Appendix A.1.1.1.

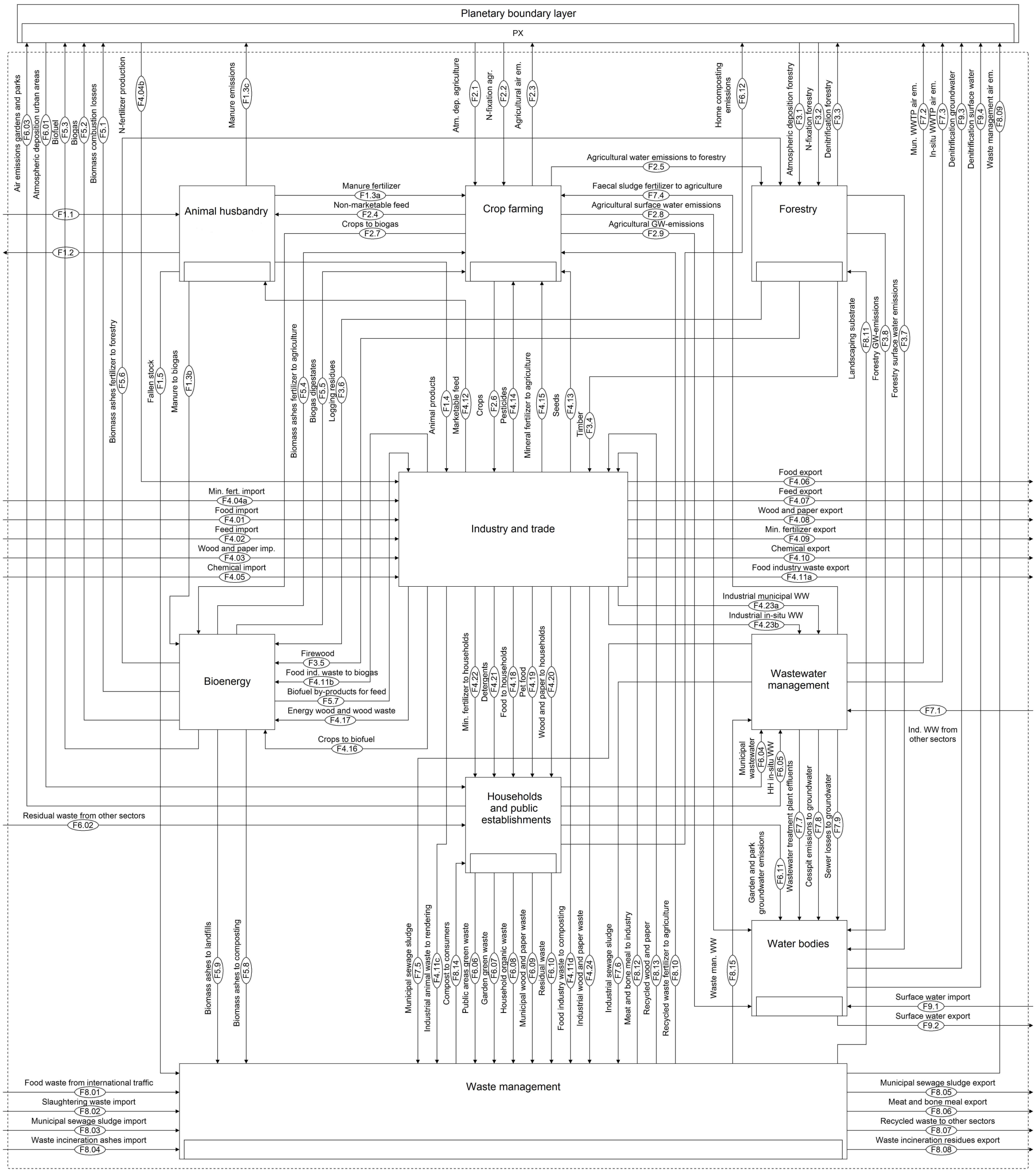

Fig. 3.1: Qualitative model of the Austrian P and N budget. Dashed line represents the system boundary, F-values in the ovals indicate the number assigned to each flow.

The coupling of the P and N balance was achieved by introducing a goods-layer into the system, which represents the total mass of a stock or flow. Where applicable, a priori data (input data of the model prior to calculation) thus consists of the mass of a flow and the respective concentrations of P and N. For some processes, especially those involving flows of air and/or water, keeping mass balance on the goods layer would have required extensions of the model that were beyond the scope of the present work, whereas data on the mass flows of N and P was readily available. For these flows masses of N and P were therefore directly used as a priori data. In addition, depending on the data source, mass is alternatively stated as dry- or fresh matter. The goods layer should therefore only be regarded as an auxiliary layer and is partly excluded from the balancing process. (For 59% of the flows and 47% of the stocks and stock changes a priori data is available on the goods layer, for flows this number rises to 84% after calculation). Linking the P and N layer takes place during data reconciliation, when masses and concentrations of flows are adjusted so that the mass balance for all processes is kept on both substance layers.

3.1.2 Measure scenarios

On basis of management strategies put forward in Shibata et al. (2017), Withers et al. (2015) and Zoboli et al. (2016) 16 individual measures to make the Austrian P and N balance more efficient were identified. According to their main aim they can be grouped into measures directed at the increase of nutrient recovery and recycling, the reduction of demand and consumption and the reduction of emissions to the environment. This classification does not mean though that measures cannot fulfill more than one of these aims simultaneously.

It should be noted that the focus of this study was on the description and functioning of the system as a whole rather than on the development of readily applicable management strategy. Therefore the measures presented should neither be regarded as exhaustive nor are they devised in full detail.

Measures aimed at increasing nutrient recovery and recycling:

- Increased recycling of food industry waste (FIW)
  In this measure it was assumed that food industry waste which is currently exported but would be suitable for composting (i.e. former foodstuff of animal origin, kitchen and food waste (BMLFUW 2017)) is composted. The surplus of compost produced was assumed to be applied as agricultural fertilizer and substitute for mineral fertilizer.

- Increased recycling of biomass ashes (BA)
  Fine fly ashes from biomass plants contain high amounts of heavy metals, which impedes their use in agriculture and composting. Coarse ashes on the other hand could be largely used as P-fertilizer (Obernberger and Supanic 2009). In this measure it was thus assumed that all coarse ashes from biomass plants are recycled, while the amounts of ashes applied in forestry and as additive in composting remain constant to the status quo and the rest is used in agriculture, where it substitutes for mineral fertilizer.

- Improved collection of organic household waste (HWC)
  As centralized composting facilities are generally more effective and private gardens and public green areas are marked by a high nutrient surplus (Maier 2017; Zoboli et al. 2016), home composting was fully replaced by separate collection of organic kitchen and garden waste in this measure. In addition, increased separation of organic material currently disposed of as residual waste was assumed. Implementing this measure would reduce N-inputs to private gardens and public green areas below plant requirements. The minimum additional amount of mineral fertilizer needed under the assumption that no N-losses to groundwater occur was therefore calculated. At the same time compost now available for agriculture substitutes for mineral fertilizer.

- Increased P-recovery from sewage sludge (SS)
  In accordance with the national strategy for future sewage sludge management (BMLFUW 2017) this measure anticipates a ban on composting and direct agricultural application of sewage sludge. Instead, it was assumed that all sewage sludge accrued is mono-incinerated and P-fertilizer recovered from the ashes. For P-recovery, LEACHPHOS® technology was studied as an example due to its high recovery potential (70-80% of the ash input) and high plant availability of recovered P in the final product (Egle et al. 2016b). LEACHPHOS® fertilizer is presumed to substitute for mineral P fertilizer.

- Increased nutrient recovery from meat and bone meal (MBM)
  This measure requires separate collection of low risk (C3) and higher risk (C1 and C2) rendering material according to EU classification (European Commission 2011; European Commission 2009). It was assumed that C3 material is directly used as fertilizer, except for a part equal to the amount in the status quo applied for feed production and other industrial purposes and that C1 and C2 material is mono-incinerated and P-fertilizer recovered from the ashes. For the latter similar recovery potentials and P-plant availability as for P-recovery from sewage sludge ashes with LEACHPHOS® technology were assumed. It is again presumed that the recovered fertilizer substitutes for mineral P-fertilizer.

- Efficient use of compost (CD)
  In order to effectively make use of nutrients contained in it, all compost produced was assumed to be applied as fertilizer in this measure, while only less valuable material is used as landscaping substrate. Amounts going to private households and public institutions remain unchanged compared to the status quo so that all former landscaping substrate is used as substitute for mineral fertilizer in agriculture.

- Increased recycling of green waste (GW)
  As ashes from green waste co-incinerated with residual waste cannot be used as fertilizer due to contamination, in this measure green waste is entirely incinerated in biomass plants. It was assumed that the surplus compared to the status quo of thus created biomass ashes is applied in agriculture as a substitute for mineral fertilizer.

Measures aimed at reducing demand and consumption:

- Full application of P- and N-optimized feed for cattle, pig and poultry (OF)
  Following the Austrian guidelines of appropriate fertilizing (BMLFUW 2006) optimized feeding practices in this measure involve a reduction of 20% of N and P content in feed for cattle and pigs and 26% for poultry. Several simplifications had to be made regarding this measure. First of all, due to lack of data, it was assumed that currently P- and N-optimized feeding plays a negligible role in animal husbandry. The effects of this measure may thus be overestimated. Furthermore, lower demand of P and N in feed were modeled as a mass reduction in production and import rates of all feed items equally, rather than as changes in concentrations or kind of feedstuff. Reduction of P and N content of feed are directly reflected in the nutrient content of manure, whereas P and N concentrations in meat and livestock stay constant.

- Shift to a healthy, less meat-reliant diet (HD)
  Nutrition according to the recommendation of the Austrian Nutrition Report (Elmadfa et al. 2009) of the total population was assumed in this measure. This corresponds to a reduction of meat consumption by two thirds and an increase in consumption of fruit, vegetables and carbon hydrates by a factor of 1.5. Percentage of food waste from industry

and households was presumed to equal status quo. Changes in consumption of different food items are reflected in changes of their import and domestic production taking into account the respective ratio in the status quo. No changes in food export compared to the status quo were assumed. Changes in animal stock numbers, animal feed requirements, nutrient concentration in waste water, etc. arising from the assumptions described above, were considered as well. Reduction potentials for agricultural land demand, agricultural water emissions and fertilizer demand were taken from a study by Thaler et al. (2015), who take similar presumptions to the ones described above. It was assumed that land no longer used for agriculture is left to natural succession.

- Food waste reduction (FWR)
  Reduction potentials of food waste from industry (ranging from 1% for meat to 45% for bakery products) and households (7% for separately collected waste and 57% for organics in residual waste) were based on estimations by Hietler and Pladerer (2017) and Pladerer et al. (2016). For changes in the system entailed by this waste reduction same assumptions as described in the measure "shift to a healthy, less meat-reliant diet" apply. However, in this case mineral fertilizer demand and agricultural emissions are not provided as a priori inputs, but calculated intrinsically in STAN.

- Exploitation of maximum fertilizer efficiency (FE)
  In this measure application losses of all types of fertilizers, including mineral fertilizers, manure, compost, etc. are reduced to a minimum and fertilizer inputs adjusted to actual plant needs. This can for instance be achieved by widespread application of the 4R Nutrient Stewardship principles – right fertilizer source at the right rate at the right time and in the right place (Reetz 2016). Detailed analysis of individual actions especially with respect to nutrient legacy and interactions between plant, soil, water and nutrients were out of the scope of this study so that plant nutrient availability for all fertilizer products was merely shifted from average to upper boundaries of values found in literature (Hamilton et al. 2017; Syers et al. 2010; Gutser et al. 2005). An exception to this is manure: For P calcultated plant nutrient availability in the status quo model after data reconciliation exceeds the values reported in literature, which is why no further increase was assumed. For N a maximum nutrient availability of 80% was applied (Ebertseder and Gutser 2002). Fertilizer efficiency is, however, not only a determined by application practice, but also depends on local factors such as soil and climatic conditions. Maximum fertilizer efficiencies are thus unlikely to be achievable for all regions and crops and effects of this measure probably overestimated. Plant availability of P in soil on the other hand tends to be underestimated in literature (Syers et al. 2010; Hanserud et al. 2016), so that potentials for mineral P fertilizer reduction may be underestimated. Savings in nutrient losses were assumed to reduce demand for mineral fertilizer.

Measures aimed at reducing environmental emissions:

- N-emission control during manure storage (MH)
  Covering liquid and solid manure during storage, for instance with peat, can reduce N emissions to the atmosphere by 80% (Rotz 2004). Like for optimized animal feed it was assumed that this is not significantly practiced in the status quo. Lower storage losses lead to a higher N-content of manure applied to fields so that the demand for mineral N-fertilizer can be reduced.

- Reduction of P-concentration in detergents (D)
  Despite past reductions due to a ban of P in laundry cleaning products, Richards et al.

(2015) still estimate a 90% reduction of detergent P load, if only non-P-based detergents for both laundry and dishwashing were used. This assumption was the basis of this measure and P concentrations of imported, exported and nationally used detergents reduced accordingly. Lower concentration of P in wastewater ultimately leads to lower P-input to agriculture with sewage sludge. As agricultural fields currently are generally over-fertilized, it was assumed that this reduction manifests itself in a reduction of water emissions and soil accumulation of P rather than in a higher demand for mineral fertilizer.

- Agricultural erosion control (EC)
  Without looking at specific measures to implement this, it was assumed that erosion from agricultural land could be limited to a tolerable and often non-avoidable amount of 1t/ha/year (Eurostat 2015). Like above, mineral fertilizer demand was reduced in line with the decreases in nutrient losses and further decrease of emissions following the reduced nutrient input taken into account.

- Increased P-removal efficiency of waste water treatment plants (WWTP-P)
  For this measure, it was estimated that P-removal efficiency of both municipal and industrial wastewater treatment plants could be raised to 96%. This corresponds to the efficiency currently reached in the Austrian Rhine river basin district (Überreiter et al. 2016). As for fertilizer efficiency, waste water treatment plant efficiency is dependent on local conditions. Implementing 96% P-removal may hence not be achievable on a national scale or at least require extensive structural changes. All P removed from the incoming wastewater is transferred to the sewage sludge. Once again, higher P-concentration of the part of sewage sludge used in agriculture enables substitution of mineral P-fertilizer.

- Increased N-removal efficiency of waste water treatment plants (WWTP-N)
  The maximum achievable N-removal efficiency of municipal and industrial wastewater treatment plants was estimated with 90%, which is equivalent to the rate currently reached in plants with a size of 10,000 - 15,000 population equivalents (Überreiter et al. 2016). Contrary to P-removal, it is assumed that N transferred to sewage sludge remains equivalent to the status quo, while additionally removed N is emitted to the atmosphere. Considerations regarding the achievability of P-removal efficiency also apply for N.

In the agricultural sector differences in fertilizer efficiency were taken into account, whenever mineral fertilizer was substituted for by other products. With the exception of the measure “Exploitation of maximum fertilizer efficiency”, efficiency factors of each fertilizer were kept equivalent to the status quo, irrespective of changes in its mass, nutrient content or total nutrient input and output, which evidently constitutes a simplification of reality. Similarly, distribution of losses among soil, atmosphere, surface- and ground water was also kept equivalent to the status quo for all measures, except for “Agricultural erosion control”, where erosion control measures lead to a relative reduction of surface water emissions with respect to other losses.

For simplification, fertilizer efficiency was not taken into account in the forestry and household sectors. Instead changes in input and output of the processes "Forests and miscellaneous soils" and "Private gardens and public green areas" are reflected directly in a change of soil stock (P) and groundwater emissions (N). Especially for the measures "Improved collection of organic household waste", "Efficient use of compost" and "Agricultural erosion control" that involve considerable reductions of P input into these processes, this simplification may lead to an overestimation of P water emissions in favor of lower soil accumulation of P.

In addition to individual measures a scenario, in which all of the 16 measures are combined, was analyzed as well. This made it possible to also consider feedback mechanisms between measures. For instance the measures “Full application of P- and N-optimized feed for cattle, pig and poultry” involves a decrease in field nutrient input with manure and thus may lead to higher requirements of mineral fertilizer, unless measures such as “Shift to a healthy, less meat-reliant diet” and “Food waste reduction”, that lead to a reduction of agricultural production and thus total fertilizer demand, are implemented concomitantly. On the other hand applying the measure “Increased P-removal efficiency of waste water treatment plants” together with “Improved P-recovery from sewage sludge” would make both measures more efficient as the increased amount of P in sewage sludge due to better removal efficiency of waste water treatment plants is increasingly made available for agriculture by the recovery process.

Model structure for measure scenarios was essentially kept equivalent to the status quo; however, some measures required addition of new elements (like a process for the recovery of P from ashes for the measures “Improved P-recovery from sewage sludge” and “Improved nutrient recovery from meat and bone meal”) or redirection of flows (e.g. the flow “Green waste to incineration” from waste incineration plants to biomass plants in the measure “Increased recycling of green waste”). Moreover, due to differences in data sources, a priori input sometimes had to be inserted into the model as transfer coefficients rather than mass flows or vice versa. As for the status quo, these changes in the model structure are depicted in Appendix A.1.1.2.

3.1.3 Handling uncertainty

Initial uncertainty of data elements from literature were characterized using a method developed by Laner et al. (2015), combining qualitative data classification with exponential-type uncertainty characterization functions. As mentioned in Chapter 3.1.1, where possible, a priori input data to the STAN system was directly derived from the data sources and all necessary conversions from volume and concentration to mass and subsequent error propagation conducted within the software. However, in some cases, for instance, if concentrations provided in a data source referred to area rather than mass or volume, intermediate calculation steps were needed to create a priori input data. In these cases, Gaussian error propagation was used to compute a priori uncertainty. This is a main difference to the model of Zoboli et al. (2015), who equally applied the method of Laner et al. (2015) to define the uncertainty of masses and concentrations, irrespective of whether they were calculated or directly derived from literature. The new approach enables greater consistency in the initial qualitative data classification.

During data reconciliation in STAN values of uncertain data are altered in a way that contradictions in the balance disappear and uncertainty of the reconciled data is reduced (Cencic 2017). Great care was therefore paid to only use independent values as a priori input data (i.e. no a priori input value is used to calculate any other a priori input value), in order to avoid underestimation of final uncertainty after data reconciliation.

For the evaluation of the reconciliation process D-values specifying the degree by which a priori values are altered during data reconciliation with respect to a priori uncertainty are calculated as follows:

$$D = \frac{\sum \frac{|reconciled\ value - a\ priori\ value|}{a\ priori\ uncertainty}}{No.\ of\ reconciled\ values} \tag{3.1}$$

Measures are evaluated with respect to the status quo. Uncertainty of the data which changes with respect to the status quo is stated in the same way as described above. For flows, stocks, stock changes and transfer coefficients that are not affected by the measure on the other hand, reconciled values of the status quo are used as a priori input, assuming a very low uncertainty of 1‰ necessary to ensure model stability. In addition, only as much data from the status quo system was transferred to the measures scenario as is absolutely necessary for its calculation, keeping overdetermination to a minimum. The aim of both proceedings is to avoid altering of data elements that are not affected by the measure during reconciliation.

Uncertainty of evaluation indicators is derived with Gaussian error propagation from the uncertainty of model output data used to compute them. EIPs are considered significant, if their absolute value exceeds the uncertainty.

3.2 Current state and improvement potentials in nutrient management

3.2.1 Status quo

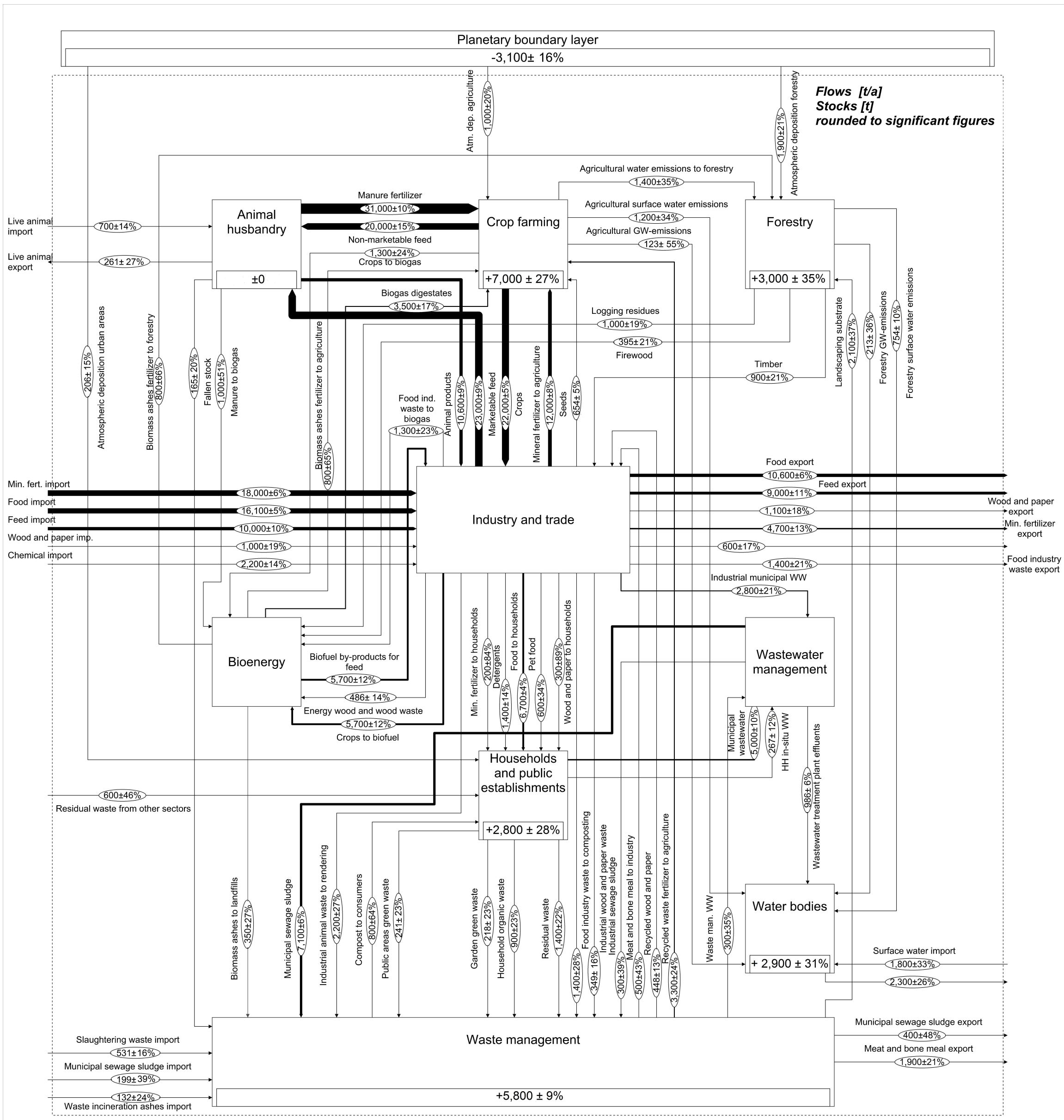

Fig. 3.2: Austrian P system for the reference year 2015. Only flows > 100t P/year are shown.

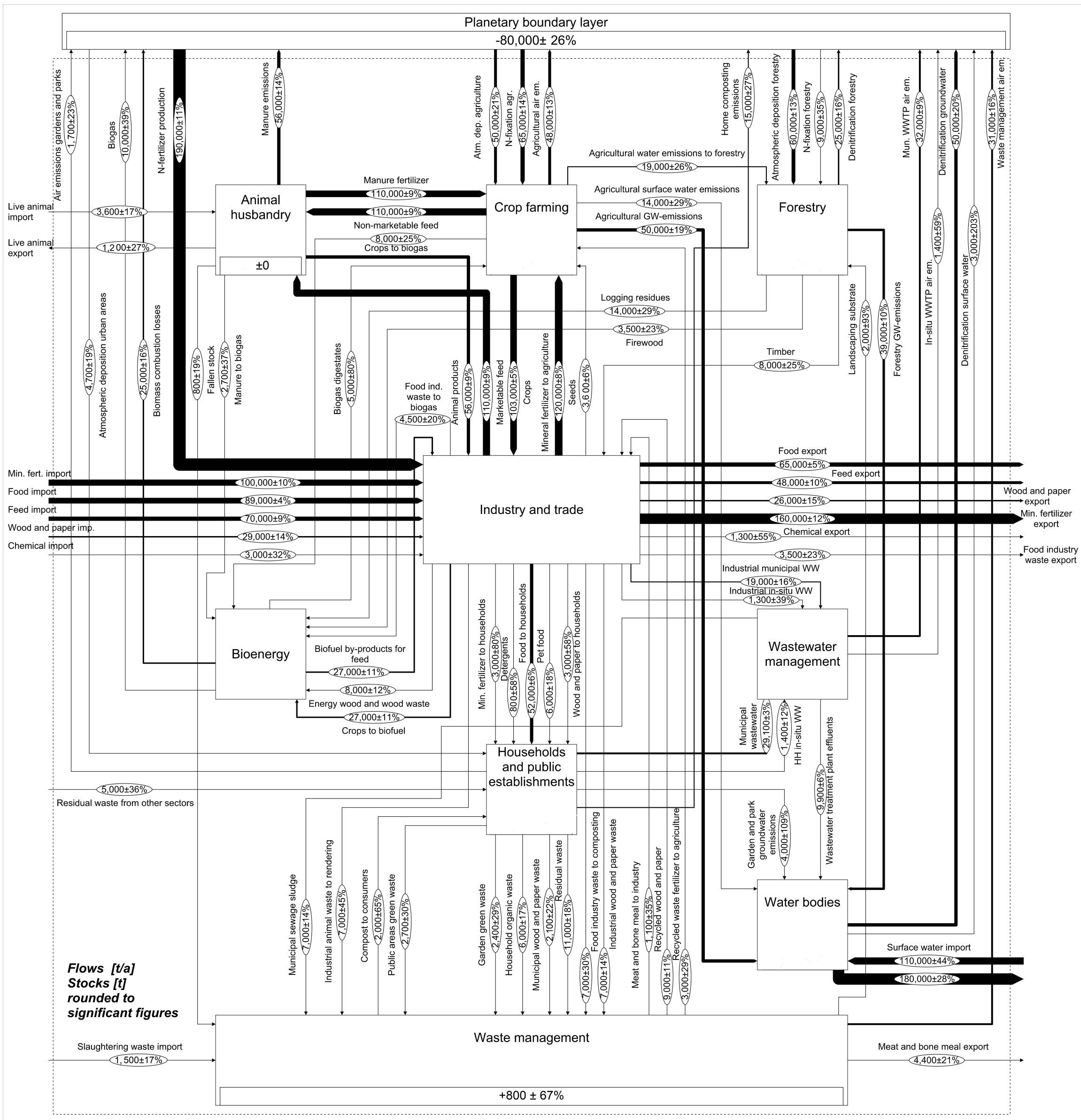

Fig. 3.3: Austrian N system for the reference year 2015. Only flows > 500t N/year are shown.

Modeling results confirm the close connection of the Austrian P- and N-systems. For both substances a linear management pattern can be observed with high nutrient imports for food production and subsequent dissipation to various emissions and the waste sector (Figure 3.2 & 3.3). However, there are differences in the predominant pathways of this dissipation: For P 50% of the net-import accumulates in agricultural and urban soils and another 37% is lost in the waste sector, whereas for N 80% of the net-imports are eventually emitted to the atmosphere (Figure 3.4). A full list of all flows in the model and their respective quantities of N and P is provided in the enclosed electronic supplementary material.

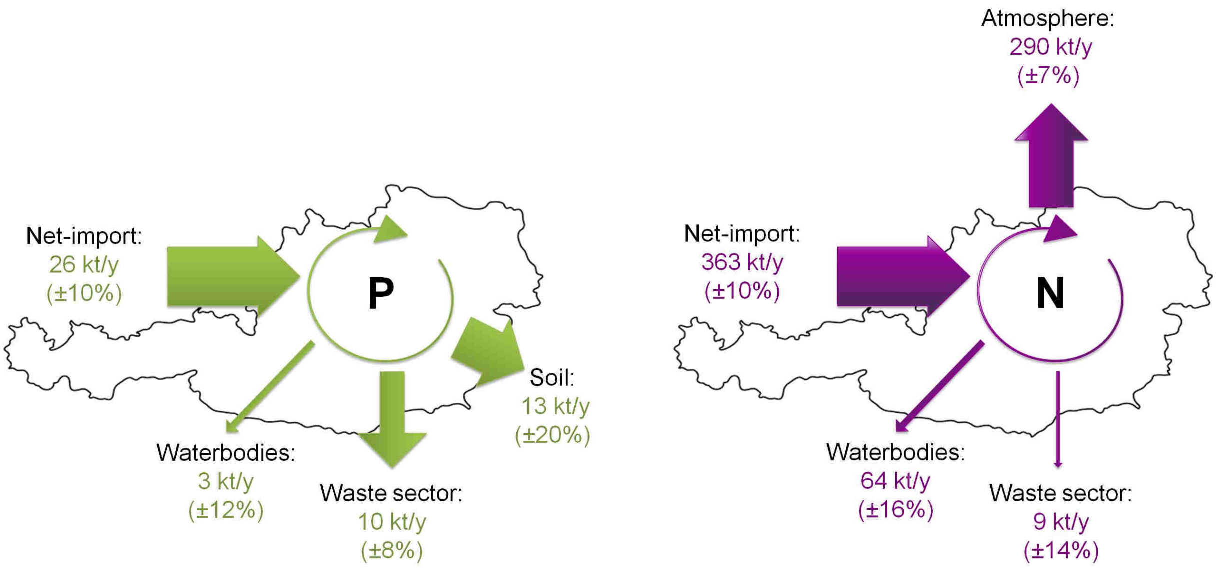

Fig. 3.4: Aggregated P and N balance for the reference year 2015.

3.2.2 Measure scenarios

Efficiency improvement potentials of each measure as well as of the combination of all measures with respect to the status quo are assessed using the following indicators:

- Demand of mineral P-fertilizer for domestic use
- Demand of mineral N-fertilizer for domestic use
- P accumulation in soil (including agricultural, urban as well as forested and natural soils)
- P losses in the waste sector (including accumulation in landfills, waste exports and P contained in cement and clinker after co-incineration of P containing waste as secondary fuel)
- N losses in the waste sector (including accumulation in landfills and waste exports)
- P emissions to ground- and surface water (including surface runoff and percolation from agricultural, urban as well as forested and natural soil and effluents from waste water treatment)
- N emissions to ground- and surface water (including surface runoff and percolation from agricultural, urban as well as forested and natural soil and effluents from waste water treatment; in order to avoid double accounting atmospheric N emissions from denitrification processes in ground- and surface water were subtracted)
  Emissions occur almost exclusively in the form of NH_3; NH_4 emissions from waste water are negligible due to the well-developed secondary treatment stage (Überreiter et al. 2016.)
- N emissions to atmosphere (including denitrification of agricultural, natural and urban soils, emissions during storage and spreading of fertilizer and emissions due to incineration, composting, rendering and waste water treatment processes)
  No distinction between the different forms of N-emissions (N_2, NO_x, N_2O, NH_3) is made.

A detailed description of the flows used to compute each indicator can be found in Appendix B.1. For each indicator x and measure y the efficiency improvement potential (EIP) is calculated as

$$EIP_{x,y} = 1 - \frac{Indicatorvalue_{x,y}}{Indicatorvalue_{x,statusquo}} \tag{3.2}$$

The total efficiency improvement potential (EIP_{tot}) of each measure is

$$EIP_{tot} = \frac{1}{8} * \sum_{y=1}^{8} EIP_y \tag{3.3}$$

An EIP_{tot} of 1 thus signifies a situation, in which all losses and inefficiencies are completely eliminated; negative side-effects of measures exist, if indicator values < 0 occur.

Table 3.2 shows the EIPs of all 16 efficiency improvement measures studied as well as for the combined measures scenario, while in Figure 3.5 their EIP_{tot} is depicted. The respective quantities of P and N for each flow and measure can be found in the enclosed electronic supplementary material.

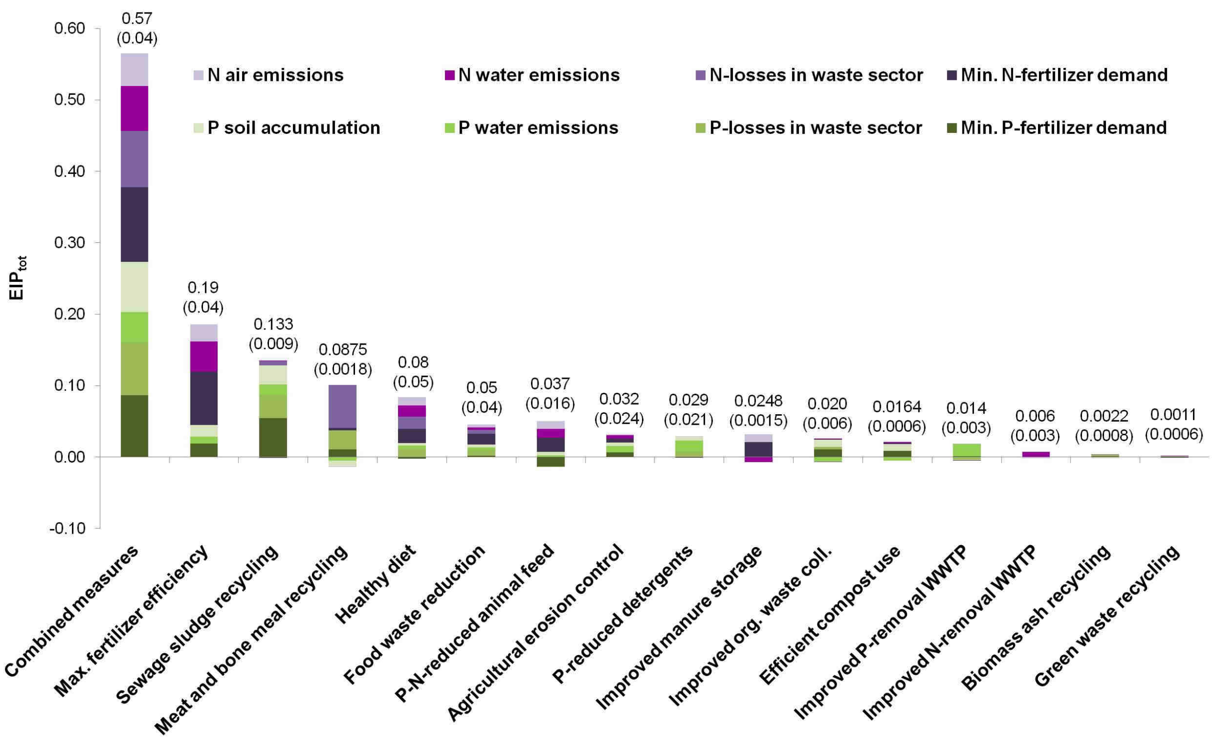

Fig. 3.5: EIP_{tot} of analyzed nutrient efficiency improvement measures. The measure "Increased recycling of food industry waste" was excluded from this depiction as it does not yield significant changes compared to the status quo.

Tab. 3.2: EIPs of analyzed nutrient efficiency improvement measures. MFD: mineral fertilizer demand, LWS: losses in waste sector, WE: water emissions, SA: soil accumulation, AE: air emissions, WWTP: waste water treatment plants. All values rounded to significant digits. Values in brackets indicate uncertainty. Highest total and single-measure values for each EIP are marked in bold, for values in italics change is insignificant.

	MFD-P	LWS-P	WE-P	SA-P	MFD-N	LWS-N	WE-N	AE-N
Combined measures	**0.69 (0.13)**	**0.59 (0.05)**	**0.34 (0.06)**	**0.56 (0.10)**	**0.84 (0.08)**	**0.63 (0.11)**	**0.50 (0.26)**	**0.37 (0.04)**
Max. fertilizer efficiency	*0.15 (0.20)*	*0.003 (0.003)*	*0.08 (0.09)*	*0.13 (0.15)*	**0.60 (0.09)**	*0.0001 (0.0009)*	**0.34 (0.10)**	**0.190 (0.019)**
Sewage sludge recycling	**0.44 (0.06)**	**0.255 (0.023)**	0.119 (0.024)	**0.22 (0.04)**	-0.008 (0.004)	0.0486 (0.0009)	0.0045 (0.0021)	-0.0059 (0.0006)
Meat and bone meal recycling	0.084 (0.010)	0.215 (0.003)	-0.037 (0.004)	-0.067 (0.007)	0.026 (0.004)	**0.4833 (0.0005)**	-0.0022 (0.0021)	-0.0030 (0.0007)
Healthy diet	*-0.02 (0.07)*	0.09 (0.06)	*0.04 (0.15)*	*0.03 (0.10)*	0.16 (0.08)	0.14 (0.11)	*0.12 (0.34)*	0.10 (0.05)
Food waste reduction	*0.01 (0.04)*	0.06 (0.06)	0.028 (0.024)	*0.03 (0.04)*	0.12 (0.07)	*0.04 (0.13)*	*0.03 (0.23)*	0.03 (0.03)
P-N-reduced animal feed	-0.11 (0.07)	*0 (0.03)*	*0.02 (0.03)*	0.039 (0.023)	0.16 (0.08)	*0.0001 (0.0009)*	0.10 (0.04)	0.089 (0.018)
Agricultural erosion control	*0.05 (0.06)*	*0.0009 (0.0014)*	0.073 (0.024)	*0.03 (0.05)*	*0.05 (0.09)*	*0.0001 (0.0009)*	*0.04 (0.15)*	*0.011 (0.017)*
P-reduced detergents	*-0.0001 (0.0018)*	0.057 (0.012)	0.1281 (0.0018)	0.044 (0.010)	*0 (0.003)*	*0.0001 (0.0009)*	*0 (0.004)*	*0.0007 (0.0014)*
Improved manure storage	*-0.0001 (0.0017)*	*-0.0001 (0.0009)*	*0 (0.0007)*	*0 (0.0009)*	0.169 (0.009)	*0.0001 (0.0009)*	-0.057 (0.005)	0.087 (0.005)
Improved org. waste coll.	0.088 (0.006)	0.023 (0.015)	-0.048 (0.003)	0.084 (0.005)	0.004 (0.004)	*0 (0.05)*	0.0067 (0.0022)	*0.0004 (0.0018)*
Efficient compost use	0.0689 (0.0025)	*0.0003 (0.0008)*	-0.0387 (0.0008)	0.0777 (0.0008)	0.010 (0.004)	*0.0001 (0.0009)*	0.0106 (0.0020)	0.0022 (0.0006)
Increased P-removal WWTP	0.009 (0.003)	-0.026 (0.004)	**0.138 (0.023)**	-0.0068 (0.0013)	*0 (0.003)*	*0.0001 (0.0009)*	*0 (0.004)*	*0 (0.0014)*
Improved N-removal WWTP	*-0.0001 (0.0018)*	*-0.0001 (0.0014)*	*0 (0.0008)*	*0 (0.0010)*	*0 (0.003)*	*0.0001 (0.0009)*	0.061 (0.022)	-0.014 (0.005)
Biomass ash recycling	0.007 (0.003)	0.0165 (0.0011)	-0.0024 (0.0010)	-0.0044 (0.0009)	*0 (0.003)*	*0.0005 (0.0009)*	*0 (0.004)*	*0 (0.0014)*
Food industry waste recycling	*0 (0.003)*	*0.0007 (0.0011)*	*-0.0002 (0.0010)*	*-0.0003 (0.0009)*	*0 (0.13)*	0.0030 (0.0009)	*0 (0.014)*	*0 (0.03)*
Green waste recycling	0.004 (0.003)	0.0083 (0.0011)	-0.0012 (0.0010)	-0.0022 (0.0009)	*0 (0.003)*	*0 (0.0009)*	*0 (0.0024)*	*0 (0.0012)*

Like in the study by Ma et al. (2013) on different management options in the Chinese food chain, also in the present case the combined measures scenario shows the highest potentials both for each indicator and in total. Positive feedbacks between measures hence clearly outweigh negative ones. Only water emissions of N may be equally well reduced by implementing the measure "Exploitation of maximum fertilizer efficiency" on its own as by the combination of measures, considering the respective uncertainty levels. (The improvement potentials for this measure may have been overestimated though, as mentioned in Chapter 3.1.2)

Nevertheless, trade-offs between different goals within one measure exist. For example in the measure "Increased P-recovery from sewage sludge" sewage sludge that was formerly directly applied to agricultural land or composted is incinerated. The N present in the sewage sludge is lost to the air in the process, causing thus both an increase in air emissions of N and in mineral N-fertilizer demand. The same is true for the measure “Increased nutrient recovery from meat and bone meal”; however, here the lack of N in meat and bone meal ashes is compensated by increased application rates of meat and bone meal that was formerly exported. As the fertilizer efficiency of meat and bone meal is lower than of mineral fertilizer, this measure is accompanied by higher losses of both P and N to water bodies and a higher soil P accumulation.

The highest negative impacts of this analysis occur for the mineral P-fertilizer demand in the measure “Full application of P- and N-optimized feed for cattle, pig and poultry”, where an increase of 11% compared to the status quo would be needed to compensate for the lower P input with manure. For N, this effect is not visible, because contrary to P, the fertilizer efficiency of N in manure is much lower than of mineral fertilizer, so that the reduction of N taken up by plants is smaller than the decrease in demand following the lower need for fodder production in this scenario. On the other hand, the low N-fertilizer efficiency of manure is the reason why N water emissions rise in the measure “N-emission control during manure storage”.

Similarly, the low P-fertilizer efficiency of compost causes P water emissions to rise in the measures “Improved collection of organic household waste” and “Efficient use of compost”. Water emissions of N from agriculture and P soil accumulation in agricultural soils also rise in these scenarios, however, as at the same time the amount of compost applied to private gardens and public green areas and forest and miscellaneous soils is reduced respectively, overall effects on these indicators remain positive. As mentioned in Chapter 3.1.2, results may be distored by the simplified assumptions on fertilizer efficiency in the later two processes though. Lower fertilizer efficiency of biomass ashes compared to mineral P-fertilizer is also the reason, why water emissions and soil accumulation of P rise in the measures “Increased recycling of biomass ashes” and “Increased recycling of green waste”.

The measures “Increased P-removal efficiency of waste water treatment plants” and “Increased N-removal efficiency of waste water treatment plants” focus on reducing emissions to water bodies without further consideration of the fate of the removed nutrients. Consequently, these measures are marked by an increase of P losses in the waste sector as well as P soil accumulation and air emissions of N respectively.

In all cases negative impacts are however outweighed by positive effects in other indicators. Moreover, they can be mitigated by combining for instance measures that would cause an increase in water emissions with measures aimed at improving fertilizer efficiency or by implementing measures that would reduce nutrient input to agriculture together with measures aimed at reducing the need for agricultural products, as can be seen in the combined measures scenario. In total, the annual amount of P and N making its way through the system can be significantly reduced: from 26±2 kt/a to 12±2 kt/a for P and 363±36 kt/a to 219±23 kt/a for N (Figure 3.6) Note that Figure 3.4 and Figure 3.6 show the full MFA balances in aggregated form. The flow "Net-import" thus comprises not only P and N in the form of mineral fertilizer, but also with food, feed and other products, while the flow "Atmosphere" stands for the net-export of deposition and emissions of N. This is why relative reductions in these flows between Figure 3.4 and Figure 3.6 differ from the EIPs "Demand for mineral P/N-fertilizer for domestic use" and "N emissions to atmosphere" listed in Table 3.2.

In general, measures aimed at reducing demand for P and/or N score higher than those directed at emission reduction, as the later are often end-of-pipe solutions that tackle the system at points, where interaction with other sectors is limited. An exception is the measure “Exploitation of maximum fertilizer efficiency” because here the reduction of nutrient losses to water bodies is directly reflected in a lower fertilizer demand. The effect of measures directed at increased nutrient recycling largely depends on the mass of recyclable product possible to obtain and on its efficiency when used as fertilizer in agriculture. As mentioned in Chapter 3.1.2 EIPs of the measure "Exploitation of maximum fertilizer efficiency" are likely to have been overestimated. However, the focus of this study was not on the exact quantification of improvement potentials, but rather on interactions of the two studied substances and the behavior of the system as a

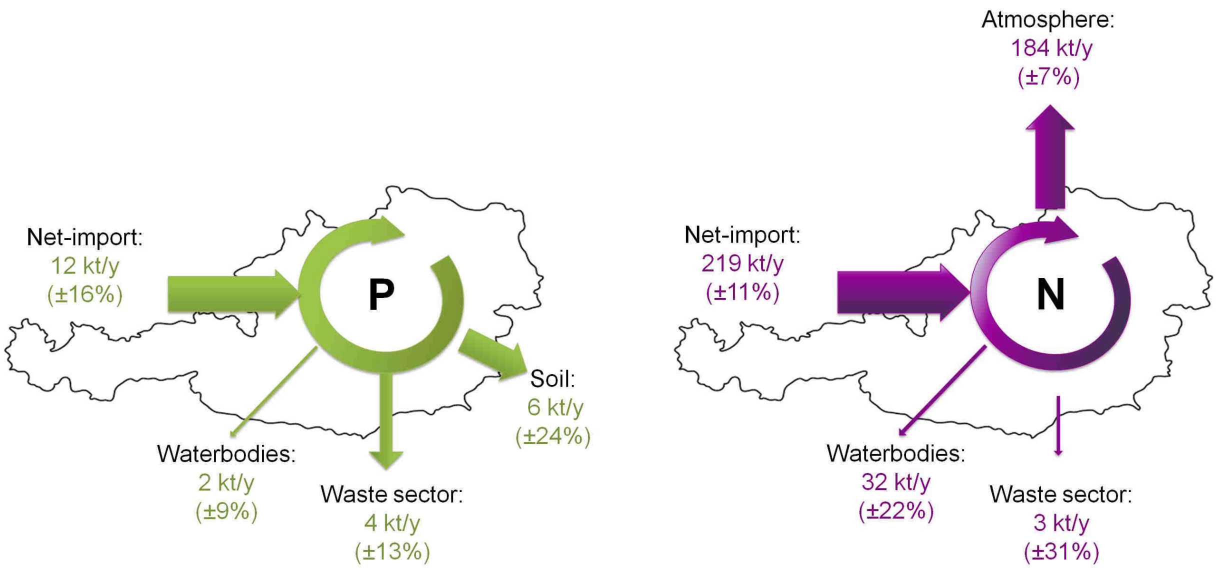

Fig. 3.6: Aggregated P and N balance for the combined measures scenario.

whole. High effectiveness of fertilizer efficiency increases are in line with the findings of Abalos et al. (2016), Ahrens et al. (2010) and Ma et al. (2013).

Furthermore, what has already been observed by Zoboli et al. (2016) for P is also true for the present study: To date, we seem to be far more successful in environmental protection than in resource protection. This is reflected in the fact that improvement potentials for mineral fertilizer demand are about twice as high as for reduction of emissions to water bodies and air, both when looking at the combined measures scenario (69% lower fertilizer demand compared to 34% lower water emission for P and 84% lower fertilizer demand compared to 50% and 37% reductions in water and air emissions respectively for N) and average scores of individual measures (5% vs 3% for P, 8% vs 4%/3% for N). This is partly an effect of measures taken in the past. Municipal waste water treatment plant removal efficiencies for instance rose from 64% to 90% for P and from 51% to 82% for N since 1999 and are now considered close to their maximal achievable limits (Überreiter et al. 2016). Nevertheless, even slight further improvements of removal efficiencies to 90% for N and 96% for P, as assumed in the present study, could reduce water emissions by 6% and 14% respectively. Although smaller than the reduction potentials achievable for mineral fertilizer demand they thus should not be omitted.

For air emissions of N another factor comes into play as well. As mentioned in Chapter 2, N-emissions to the atmosphere can take various forms with very different consequences for the environment and human health. Current measures to reduce N-emissions to the atmosphere are predominately concerned with transforming reactive forms of N into N_2 prior to emission (e.g. Campos et al. 2016; Grosso et al. 2009). Indeed, N_2 can be considered as unproblematic from an environmental perspective; in terms of resource efficiency keeping reactive N in the system as long as possible would be preferable though, because it would reduce the need for energy intensive artificial conversion of N_2 into NH_3 in the Haber-Bosch process.

It has to be noted that the combined measures scenario only represents the best case in the context of the present analysis. Although interaction between different measures was taken into account, only a case in which all measures are fully implemented was studied. Even higher increases in efficiency improvement potentials may be reached by implementing measures only up to a level in which co-benefits with other measures are maximized and negative trade-offs minimized. Creating such a true optimization model could be the issue of further studies.

Moreover, measures are specifically fitted to the reference situation of 2015, which only represents a snapshot in time. While dynamic MFA has evolved into a frequently applied method for metals over the past 20 years (Müller et al. 2014) it has not yet been applied much to nutrients. Zoboli et al. (2015) and Keil et al. (2018) have conducted time-continuous studies on P flows in Austria and India, efforts that should be increased to gain better understanding of the dynamics in P resource management. The availability of an extensive data set on national P flows since 1990 from the work of Zoboli et al. (2015) provides an excellent basis for developing the present model into a dynamic MFA. A top-down approach, where stocks are derived from the difference of inflows and outflows (Müller et al. 2014), possibly using bottom-up estimates for calibration and verification as in Buchner et al. (2015), seems to be the most suitable approach. Furthermore, system dynamics models as in Treadwell et al. (2018) could be used to study effects of implementation time of the different management scenarios evaluated in the present study. Finally, results are highly dependent on the weights assigned to the different efficiency improvement indicators. In the present study, equal weights for all indicators were assumed; it could be argued though that, considering the small absolute amounts of N ending up in the waste sector (about half of this being exported meat and bone meal), relative reductions in this field should be valued less than for mineral fertilizer, where high total quantities are involved. In aquatic ecosystems on the other hand even comparatively small increases in nutrient inputs might cause ecological collapse, thus justifying higher weights assigned to this indicator. However, goal weighting is always a subjective process and highly dependent on the specific aim of the problem at hand. It was therefore not considered an issue of the present work.

3.2.3 Quality of data reconciliation

Table 3.3 gives an overview of the results of data reconciliation in STAN. The system has a low degree of overdetermination so that reconciliation was only possible for 40% of flow output data, 25% of TC output data and 84% and 86% of stock and stock delta output data respectively.

The average D-value (degree by which a priori values are altered during data reconciliation with respect to a priori uncertainty) of 4% could indicate high compatibility of a priori data from different sources; however, it may also be a result of the generally high uncertainty (34% on average for all a priori input data). Uncertainties of such magnitude are not uncommon in MFA: A priori mean uncertainty in the model of Zoboli et al. (2015) is 29.7%, while data reported by Cooper and Carliell-Marquet (2013) and Antikainen et al. (2005) on P flows in the UK- and P and N flows in the Finnish food production and consumption systems respectively both exhibit mean a priori uncertainties of 31%. Volatility of N during fertilization, manure management and waste water treatment, variability in soil leaching processes depending on soil type, slope and cultivation method as well as insufficient reporting of consumption and waste management processes were identified as the main sources of uncertainty, which matches the experiences gained in the present study. In particular increased tracking of flows of wastes and by-products, additional information on the amounts of fertilizer and compost consumed in private households and on nature and amounts of home composted waste, as well as improved methods to upscale soil leaching, erosion and N-emissions of a specific site or process to the country scale would be needed in order to reduce a priori uncertainty. The generation of such knowledge was, however, beyond the scope of the present study.

Tab. 3.3: Overview of data reconciliation results. D: degree by which a priori values are altered during data reconciliation with respect to a priori uncertainty.

	Reconciled data [% of output data]	**D** [%]	**Mean a priori uncertainty** [%]
Flows	40.3	3.8	17.6
Stocks	85.7	2.8	14.1
Stock deltas	84.0	3.8	2835.9*
Transfer coefficients	25.2	4.4	39.8
Overall	**36.8**	**3.9**	**34.1**

* Mean uncertainty > 100% is a result of Gaussian error propagation of a difference.

After calculation and data reconciliation, mean uncertainty of mass flows, stocks and stock changes of P and N rises to 125%. However, if uncertainty is scaled to flow mass, weighted mean reconciled uncertainty is reduced to 47% for P and 19% for N, which is again in line with the range commonly found in MFAs on P and N. The high mean uncertainty is hence caused by a small number of mass flows and stock changes with extraordinary high uncertainties, but small absolute values and was not deemed to restrict the overall outcome of the study. Furthermore, values, for which a priori input data was obtained from a difference of source data elements or which are computed in STAN from a difference of other flows in and out of the respective process, may exhibit uncertainties >100% due to Gaussian error propagation. The uncertainty of a value C for which C = A - B is calculated as:

$$Uncertainty\ C = \sqrt{Uncertainty\ A^2 + Uncertainty\ B^2} \tag{3.4}$$

Thus, if A and B have high uncertainties and their difference is small the uncertainty of C will be larger than its value. This also explains why weighted mean reconciled uncertainty for P is considerably higher than for N: Soil stock deltas are calculated in STAN as the difference of input and output flows of the respective process and therefore disproportionally affected by the abovementioned effect; moreover, they are relatively large in terms of mass. A main assumption of the N system is the absence of soil stocks and consequently uncertainties >100% occur less often. Probabilistic MFA via Monte Carlo simulation may avoid such problems and has recently been promoted as the most adequate method for uncertainty assessment (Müller et al. 2014; Wang and Ma 2018). However, its application in the present study is hampered by the amounts of data required (according to Wang and Ma (2018) at least 30 records for each data element should be available for a valid analysis). At any rate, further analysis of the roots of data uncertainties and their propagation in the system might provide further insights into the handling of data uncertainties in MFA.

3.3 Advantages of coupled MFA

Although coupling significantly raises model complexity, it could be shown that material flows of more than one substance can be simultaneously analyzed in a rather complex system. Transition from mono-substance to multi-substance MFAs with the tools at hand (i.e. the MFA-software STAN) should thus be possible also for other resources. Coupled MFA reveals interrelations, co-benefits and trade-offs of different resource systems that might have been omitted in a mono-substance analysis and thus improve judgment of sustainability and viability of different management strategies.

In the case of P and N, coupled MFA highlighted the close connection between the two nutrients and showed that measures aimed at improving P management are overall beneficial also for the N system and vice versa. Even where trade-offs exist, for instance where nutrient losses to water bodies increase due to a substitution of mineral fertilizers by organic fertilizers with lower nutrient efficiency, the negative effects are always outweighed by positive ones and can be compensated by combining different measures with each other. The best results in the context of the present study can thus be achieved by a simultaneous implementation of all the individual measures analyzed.

The high level of uncertainty encountered in the present model, even though Austrian P management was among others chosen as a case study for its high level of reporting and generally good quality of available data compared to other substances (Zoboli et al. 2015), stresses the need to continue the ongoing research on dealing with uncertainty in MFA.

Chapter 4

Generic framework for material flow analysis of complex systems

4.1 Principles for framework design

The aim of the generic MFA system described in this chapter was to provide a common reference for studies on various substances and materials, spatial scales, and regions on the basis of which indicators for CE assessment, as well as for related concepts such as resource efficiency, can be calculated. To overcome drawbacks of both broad and substance- or sector-specific MFA schemes described in Chapter 1, a generic, universally applicable structure could be combined with flexible subsystems.

The characteristics that the framework should exhibit are similar to the ones described by Saidani et al. (2017), i.e. systemic, yet adaptive and flexible design, intuitive user interface, and use of a commonly available software. However, while the guidelines for CE frameworks suggested by Saidani et al. (2017) apply for the assessment of product performance in the context of the three pillars of sustainable development (economic, environmental, and social), the present framework was solemnly based on MFA. Not all of the aspects of sustainability are contained in the concept of CE and not all aspects of CE can be covered by MFA, as will be further discussed in Chapter 4.5. Therefore, MFA-based indicators should be regarded as an integral, yet not sufficient part of a holistic sustainability assessment.

The study on the Austrian P-N system in Chapter 3 provides an excellent case study for the development of a generic MFA structure not only because two substances are studied simultaneously, but also because with 194 stocks and 866 flows the model is particularly complex. Moreover, the management and consumption patterns differ in several ways from the "classical" extraction-manufacture-use-disposal scheme. For instance, unlike most other resources, the majority of N is found in the atmosphere, which is why mining activities are usually negligible. On the other hand, exchanges with natural compartments play an important role in the N system because N takes up multiple chemical forms as it circles through air, water, and terrestrial ecosystems, some of which act as pollutants and/or greenhouse gases (Sutton et al. 2011).

To ensure applicability of the framework in a variety of contexts, two additional case studies apart from the Austrian P-N systems were selected. For that purpose a literature research using "material flow analysis" and "case study" as search terms was conducted. The criteria for selection were treatment of a different material, region and scale other than the case studies already selected, authorship outside the working environment of the authors of the present study, actuality and availability of a closed mass balance depicted in a MFA scheme or sufficient data in tables and/or the main text to compile such a scheme. These case studies are briefly described in Chapter 4.2.1 and Chapter 4.2.2.

The generic MFA system presented in Chapter 4.3 is the optimal variant of an iterative process of transforming these three heterogeneous, real MFA systems into a common framework. The system was built in the freeware STAN (Cencic et al. 2017), a balancing and visualization tool for MFA of goods, substances, and energy, which considered data uncertainties through error propagation, data reconciliation, and gross error detection. STAN can be used for both static- and quasi-dynamic analysis and allows studying a system on several levels of detail via the introduction of subsystems.

4.2 Additional case studies used for framework design

4.2.1 European copper system 2014

The study on copper (Cu) flows in the EU 28 (Soulier et al. 2018) constitutes a representation of a “classical” MFA of a highly-recyclable metal resource on a super-national level. Figure 4.1 shows the MFA flow chart as depicted by Soulier et al. (2018).

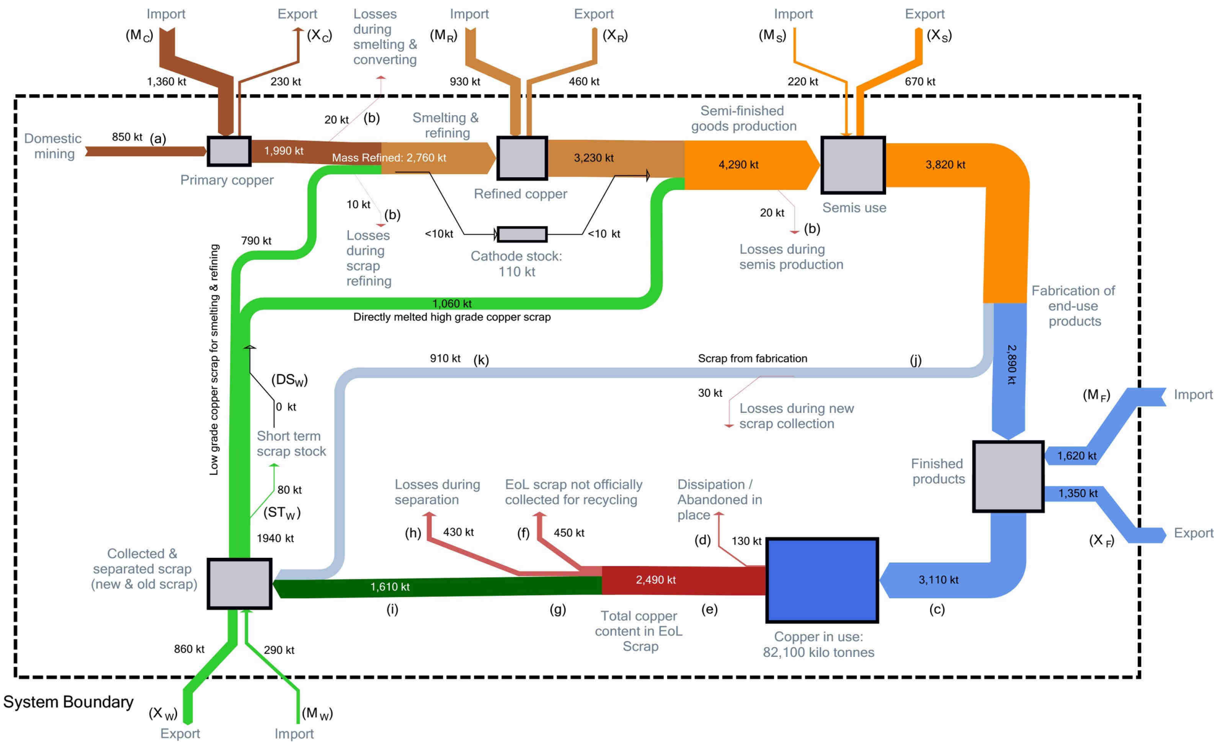

Fig. 4.1: Copper stocks and flows in the EU 28 2014 (Soulier et al. 2018).

4.2.2 Energy analysis of Karlskoga combined heat and power plant in Sweden 2016

To further test universal applicability of the generic structure, a plant-scale analysis of energy flows in a Swedish combined heat and power plant (CHP) was chosen as the third case study (Karlsson et al. 2018). The structure of this system is shown in Figure 4.2. As the mass balance is incomplete, only energy flows were further considered in the present study. When compiling the generic system, numbers given in Figure 4.2 were supplemented with information from the main text of the article.

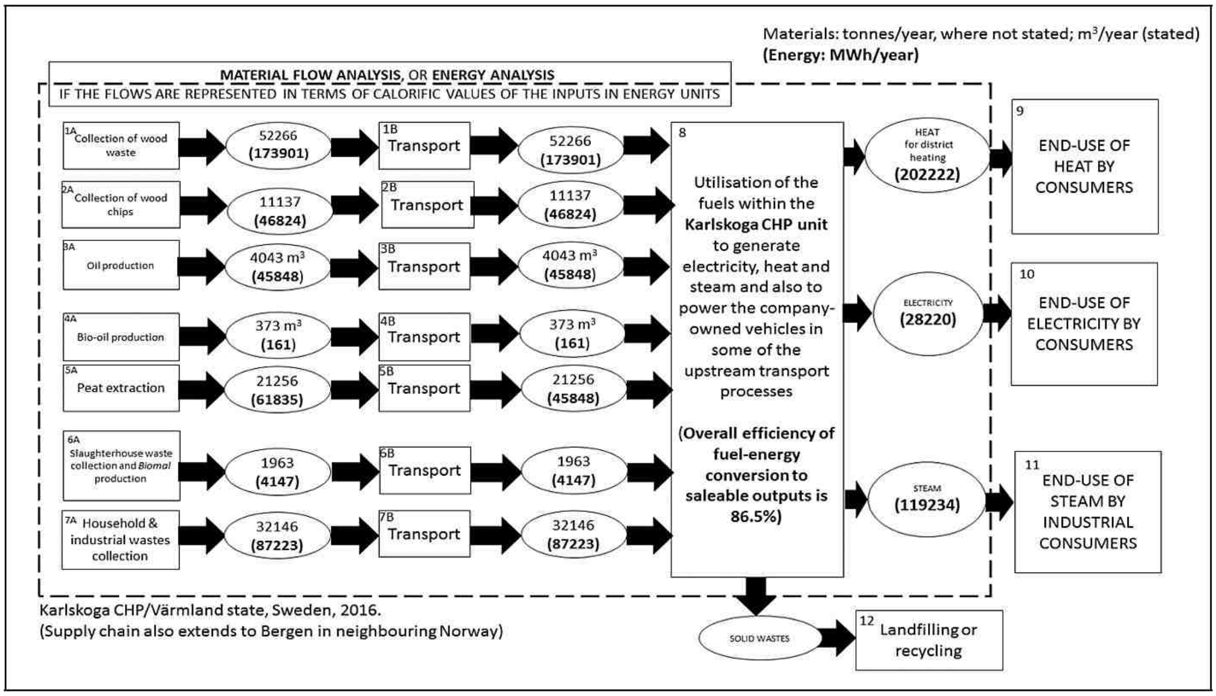

Fig. 4.2: Material and energy flows of Karlskoga combined heat and power plant 2016 (Karlsson et al. 2018).

4.3 The generic MFA structure

Figure 4.3 shows the generic structure of a resource system that evolved from the present work. It consists of five processes (one of them situated outside the system boundaries) and 14 flows.

Material can enter the system either through imports (F01, F11) or deposition from the environment (F03, F08). F01 includes the import of raw material, semi-finished products, and goods ready for consumption. However, a distinction with respect to the import of waste material from outside the system (F11) is made as the latter does not enter the production process, unless it is recycled. Input to the production (F03) and use processes (F08) can, for instance, occur in the form of atmospheric deposition or freshwater abstraction. The process “environment” (P5) includes the atmosphere, biosphere, hydrosphere and pedosphere. P5 is situated outside the system boundaries because, while the magnitude of emissions to the environment from production (F06), consumption (F09), and waste management processes (F14) is an essential characteristic of a resource management system, these emissions are commonly regarded as losses or even act as pollutants in the compartment they enter. Furthermore, exchanges of air and water masses across system boundaries are usually difficult to quantify, but only play a minor role for the purpose of the analysis. Although the lithosphere can be regarded as part of the environment, geological reserves are considered separately in P1. This accounts for their importance for the management of resources.

Each process contains a stock (S_{Px}) representing, e.g., geological deposits in P1, production storage in P2 or landfills in P4. During each period of analysis material can either be extracted from the stock ($\Delta S_{Px}\uparrow$) or deposited to the stock ($\Delta S_{Px}\downarrow$).

The production sector P2 comprises all production steps such as primary production, manufacture, and trade so that “use” in P3 should be considered as end-use only. Hence, F09 and F10 only refer to emissions and wastes that occur during or after, but not prior to consumption.

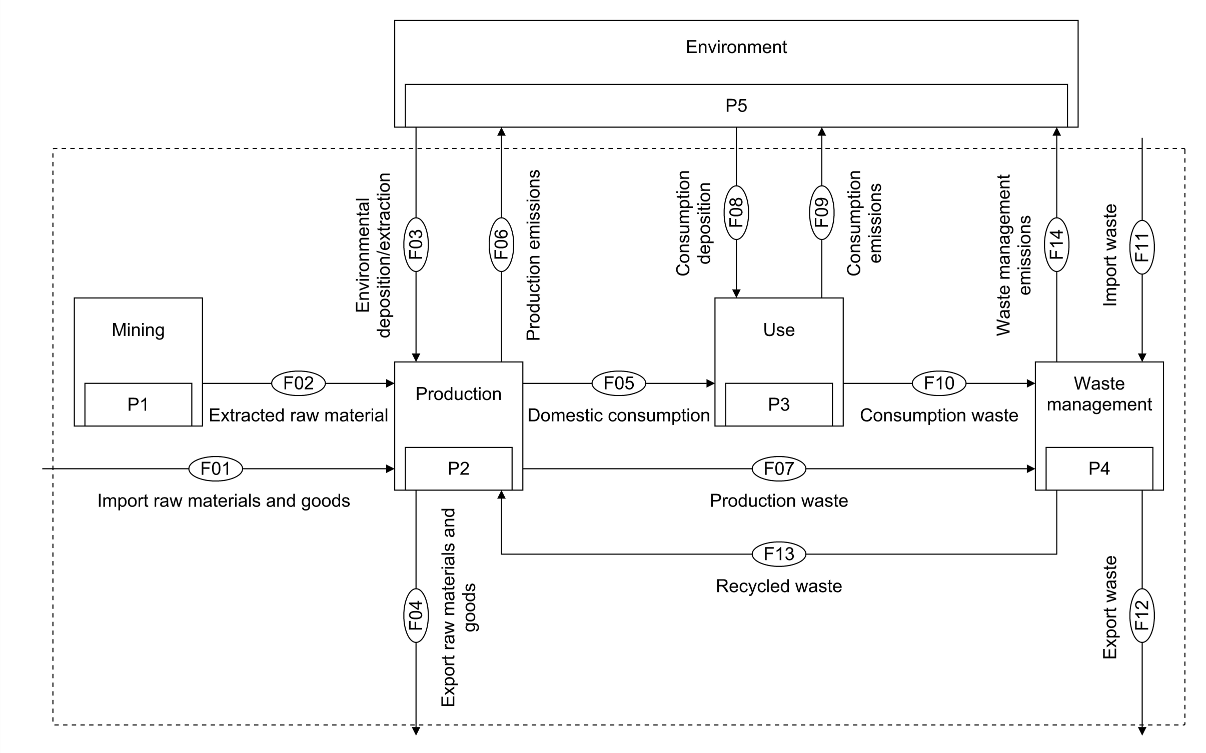

Fig. 4.3: Generic MFA system. The dashed line marks the system boundaries.

Apart from being consumed, products can be exported to sectors or regions outside the system boundaries. Again, a distinction is made between the export of raw materials and semi-finished and finished goods (F04) and the export of waste (F12). In the case of F04, export, or rather the value created from it, is an explicit reason for production, whereas waste export is mainly a way of disposal. A special case occurs if products or material stemming from recycling are exported. In this case they should first be directed to the production process, where trade is located, as part of F13 and subsequently included in F04. This has for instance been done for wood-, slaughterhouse-, household- and industrial-waste entering the power plant in the Karlskoga case study. It should be noted that depending on the purpose of the study output products may not always be visible in the MFA. This would, for instance, apply to energy and heat production in a mass-based MFA, or in the case of SFA, to any product not containing the substance analyzed.

Disposal in landfills is represented in the stock of the waste management process P4. It is assumed that all recycling takes place in P4, and therefore even if production step B makes direct use of waste from production step A, this waste should both be included in F07 and F13. Similarly, secondhand consumer products are taken into account in F10, F13, and F05.

For example, the aforementioned cyclical use rate, as a CE indicator that could be derived from the generic system, is defined as the amount of cyclical use divided by the sum of cyclical use and natural resources input and could be calculated as in Equation (4.1).

$$CUR = \frac{F13}{F01 + F02 + F03 + F08 + F11 + \Delta S_{P2}^{\uparrow} + \Delta S_{P3}^{\uparrow} + \Delta S_{P4}^{\uparrow}} \tag{4.1}$$

where Fx and $\Delta S_{Px}{}^{\uparrow}$ correspond to the mass flows and stock extractions as depicted in Figure 4.3 (depositions to stocks $\Delta S_{Px}{}^{\downarrow}$ are regarded as system outputs, and therefore not included in the total mass input). Similarly, socioeconomic cycling rates, quantifying the share of secondary material at system input (ISCr) and system output (OSCr), respectively (Mayer et al. 2018), could be applied as in Equations (4.2) and (4.3).

$$ISCr = \frac{F13}{F01 + F02 + F03 - F04 + F08 + F11 + \Delta S^{\uparrow}_{P2} + \Delta S^{\uparrow}_{P3} + \Delta S^{\uparrow}_{P4}} \tag{4.2}$$

$$OSCr = \frac{F13}{F06 + F07 + F09 + F10 + F11 + \Delta S^{\uparrow}_{P4}} \tag{4.3}$$

A "classical" recycling rate could be defined as

$$RR = \frac{F13}{F07 + F10 + F11 + \Delta S^{\uparrow}_{P4}} \tag{4.4}$$

The idea behind the generic system is that the structure depicted in Figure 4.3 forms the topmost level, on which comparisons across materials, regions, and scales can be undertaken, whereas, detailed study-specific flows and processes are included as subsystems. An example of such a cascade of subsystems for the Austrian P system is depicted in Figure 4.4.

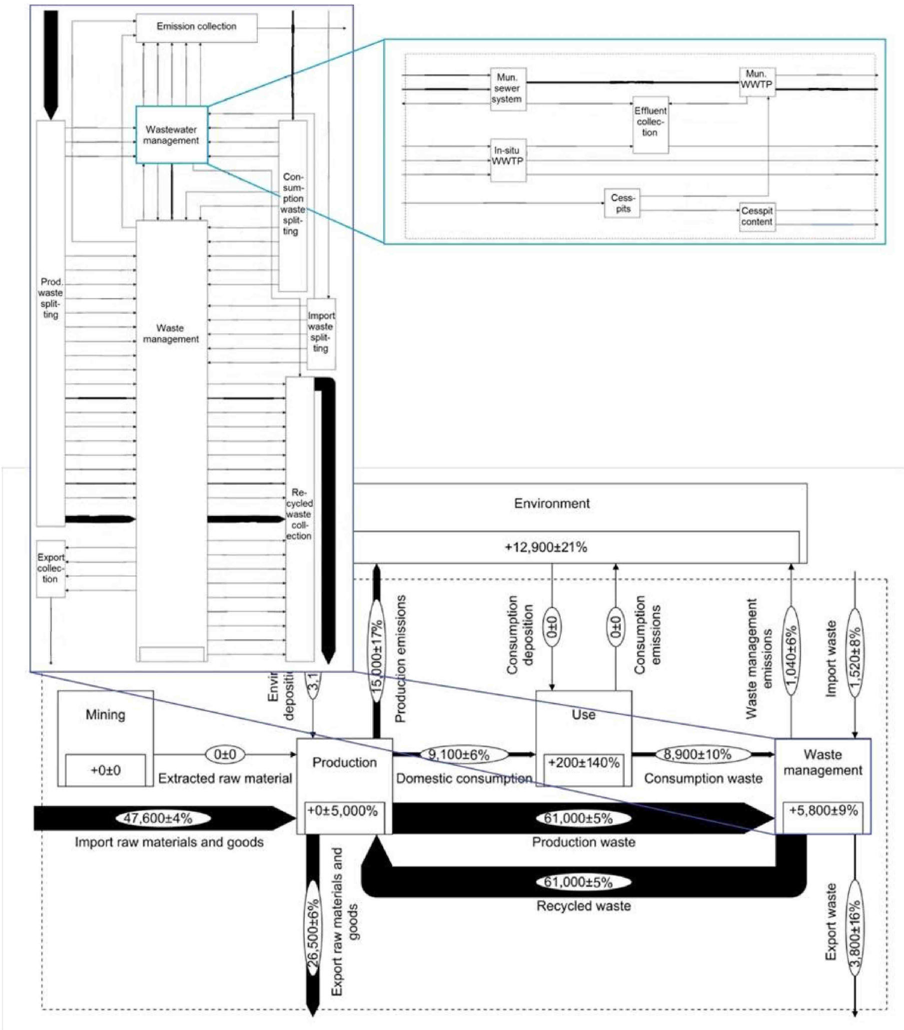

Fig. 4.4: Cascade of subsystems. Example from the Austrian P system 2015.

4.4 Results of case study transformation

Results of the transformation of the three case studies described in Chapter 3 and 4.2 into the generic system can be found in Figure 4.5. A full inventory of subsystems is provided in Appendix A. The schemes depicted in Figure 4.5 already reveal some of the particularities and management problems of a specific material or setting. For instance, the biomass dominated P-N system is characterized by high exchanges and the environment and recycling seems to play an even larger role than for Cu, which is considered one of the most recycled metals (International Copper Study Group 2018). On the other hand, production appears not to be very efficient as only a small part of input P/N ends up in F05 (domestic consumption). The system boundary in Figure 4.5d is the Karlskoga plant, thus all electricity, heat, and steam produced leave the system as export, and the use sector is virtually inexistent. According to Karlsson et al. (2018), part of the electricity produced is used to power company-owned vehicles, and therefore could be regarded as end-use consumption. However, as no further information on the magnitude and nature of this flow is given, it was neglected in the current study.

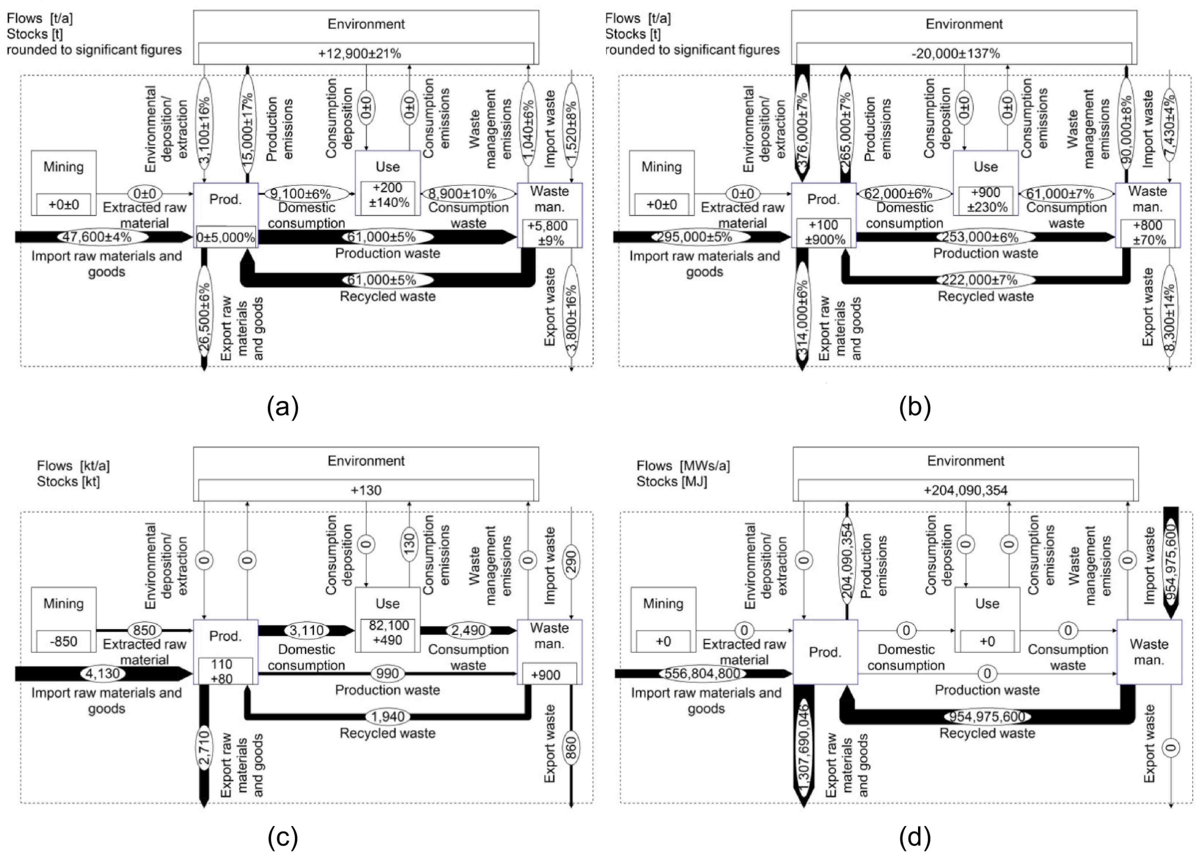

Fig. 4.5: Generic MFA system of the three case studies: (**a**) Austrian P-N system–P; (**b**) Austrian P-N system–N; (**c**) EU28 copper system; (**d**) Energy flows in Karlskoga heat and power plant.

4.5 Suitability of the framework structure as a generic evaluation basis

4.5.1 Transformation challenges

Although all three case studies could be transformed successfully, flow allocation to one of the aggregate flows of the generic system was not always straightforward and it became clear that to some extent results will always depend on subjective judgments. Therefore, Figure 4.5 only represents one out of several possible outcomes. The main challenges encountered are described below.

4.5.1.1 Definition of by-products

The distinctions among products, by-products, and waste are an issue which is much discussed in MFA. Hashimoto and Moriguchi (2004) define by-products as "outputs other than the main product, which are produced during the course of producing the main product, [...] ,regardless of their value or of whether they are solid, liquid, or gaseous." However, they noted that several high-value products can emerge from a production process, which should then be called co-products. The boundary between by-products and co-products is not always clear, especially if the outputs of one process are directly used as input to another production process. Moreover, if recovered products are used within the same production process, they may not be disclosed as separate flows, but "hidden" in improved process efficiency.

Another approach is to differentiate between waste and products according to economic value. However, if waste is perceived as a resource for material or energy and traded on recycling markets, it has an economic value and the distinction becomes once again unclear (Korhonen et al. 2018).

For the purpose of the present study, a distinction between co-products and products, which constitute the reason a production process is undertaken, emissions which are directly discarded to the environment, and waste is undertaken. Therefore, all by-products are included first in F07 (production waste) and subsequently F13 (recycled waste), even if they are directly utilized in the same or a subsequent production process without physically entering the waste sector. However, the problem of ambiguity between co-products and by-products remains.

This is especially relevant for studies in which biomass flows play an important role, such as manure in the Austrian P-N system. On the one hand, manure constitutes a waste product in animal husbandry which has as its main outputs meat, dairy, and eggs and is considered as recycled waste in this study, according to the assumptions outlined above. On the other hand, due to its widespread use as agricultural fertilizer, and to a lesser extent for biogas production, it may also be classified as a co-product. To complicate matters in the absence of legislative regulation excess manure is often "dumped" on fields irrespective of fertilization needs (Melse and Timmerman 2009), which may even justify regarding it as an emission to agricultural soil. The high share of recycling in the Austrian P-N system in Figure 4.5a,b can mainly be attributed to manure classification, as manure makes up half of the mass of F13 for both substances. This may prove problematic because it might conceal the effect of measures to improve nutrient management. For instance, even if the P-recovery from meat and bone meal was raised from a current level of 26% to 100% and a total of 3000 t, which is equivalent to approximately 25% of the current mineral fertilizer demand (see Chapter 3.2.2), F13 would merely increase by 5%. Similarly, Kovanda (2014) yielded a cyclical use rate for the Czech Republic of 9% if manure was considered in the recycling flow, but only 5% if it was not considered in the recycling flow. This may explain why several authors decided to exclude manure from recycling (Haas et al.

2015; Matthews et al. 2000; Mehr et al. 2018; Rahman et al. 2019. If manure was defined as a co-product, it would only be featured on the subsystem level of the generic structure, as can be seen in Figure 4.6.

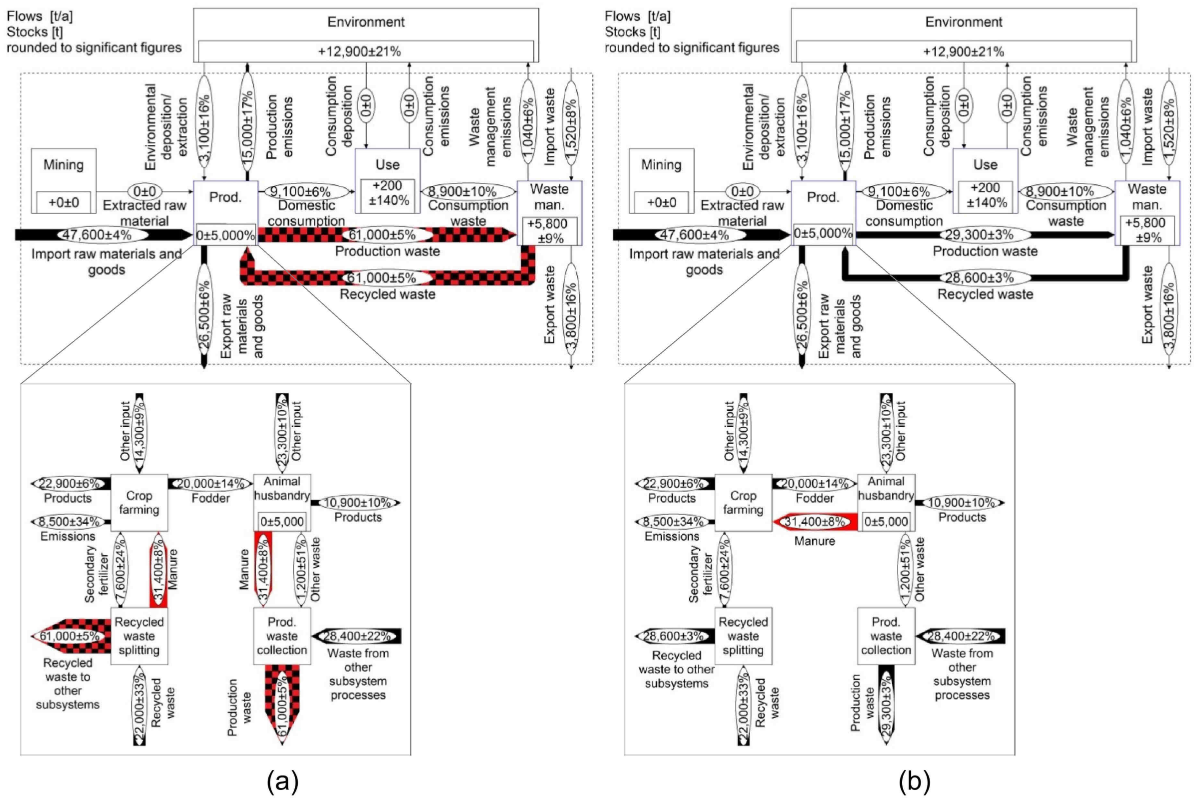

Fig. 4.6: Comparison of alternate forms of manure definition in the Austrian P system 2015. Generic topmost layer and schematic detail of the subsystem production if (**a**) manure is defined as recycled waste and (**b**) as a co-product. Manure flows are marked in red and checkered pattern indicates inclusion of manure.

Although manure is clearly the most relevant example in the present study, room for interpretation in the distinction between co-products and recycled waste also occurs for parts of the output flows from the timber and food industries, such as logging residues and wood chips burned for energy production, whey fed on-farm to cattle, and offal used for pet food production. Like manure, pet excrements on public areas can be regarded as fertilizer and thus recycled waste, which is the perception adopted in the present study. However, as the application is unintentional, they may also be viewed as an emission to soil.

Following a definition by Braungart et al. (2007), the UNEP even suggested that all biological nutrients are per se circular flows (Ajero et al. 2012). Actually, emissions of N_2, which constitute the vast majority of atmospheric N emissions, could be viewed as a way of replenishing stocks for future (natural or technical) N-fixation. On the other hand, N and P emitted to water bodies are not readily re-extractable and may even cause severe damage in the form of eutrophication. Thus, substance specification and location within the environment is particularly important for flows of biological nutrients. Differentiating emissions, waste, and recycling can partly reflect this, even if environmental impacts are not explicitly accounted for.

4.5.1.2 Allocation of processes to subsystems

Aggregation ambiguities do not only apply to flows but also to processes of the generic system. For example, the generation of district heat is regarded as a side effect of waste incineration (the main purpose of the process being the disposal of waste), and therefore was included in the waste management sector, while energy production in biomass plants was included in the production sector. However, from a plant perspective, energy generation in the Karlskoga CHP, which is powered by both primary fuels and waste, is regarded as a production process. Similarly, the use of meat and bone meal and sewage sludge as additives in cement and brick production could be considered as recycling from a goods perspective. Nevertheless, this application is treated as an endpoint of waste, similar to landfills, in the case study on the Austrian P-N system, because P incorporated in construction material cannot fulfill its purpose as a nutrient any longer (N is mainly emitted into the atmosphere during the combustion process).

Furthermore, the EU 28 copper system includes a short-term scrap stock, which could alternately be located in the production or the waste management sector. In this study, it was decided to include it in the production sector, thus regarding recycling as completed when copper enters the short-term scrap stock. This influences C in a temporal manner. In the present case, recycling is higher in the years when material is added to the stock and amounts to 27% of the total material input, whereas, if the scrap stock was considered as a part of the waste management sector, recycling would be higher when material leaves the stock, and in 2014 it would only be 26%.

Once again, allocation proves particularly challenging with regards to biomass flows. Soils used for food and timber production can be regarded either as a production stock, where nutrients are stored for subsequent growing seasons, or as an environmental compartment. The latter view was adopted in the case study on the Austrian P-N system, because the time series of P flows revealed a long-term accumulation in soil (Zoboli et al. 2015), which points to a storage in forms that are to a large extent unavailable to plants (Hamilton et al. 2017). If soils were regarded as a stock in the production sector, a considerable shift of 13,000 t P from F06 (production emissions) to SP2 (production stock) would occur. The N system does not change because soil stocks of N are neglected in this study. Private gardens and public parks form a part of the production process in the Austrian P-N system, the main output being home-grown vegetables. However, nowadays vegetable production constitutes only a minor purpose in gardens and parks, and therefore it may also be allocated to the consumption sector. This would entail a shift of mass from F03, F06, and F07 (deposition, emissions, and waste of the production sector) to F08, F09, and F10 (deposition, emissions, and waste of the consumption sector).

4.5.1.3 Insufficient flow definition

Although the generic system was formulated in a very general and coarse way, sometimes information given in the case studies was not detailed enough to allocate flows properly. For instance, in the European copper system Soulier et al. (2018)) do not differentiate between environmental emissions and landfilled waste. Therefore, additional information had to be obtained from other sources. Bertram et al. (2002) estimated that less than 0.5% of copper entering the waste system was lost to the environment, and therefore it was assumed that all losses, except for the flow named “dissipation / abandoned in place” which corresponded to F09, ended up in landfills. Likewise, further information on the source and nature of the flow of wood chips in the Karlskoga CHP (Karlsson et al. 2018) might have facilitated the decision whether it should be considered as a primary fuel or waste input. Moreover, in this study, mass flows had to be excluded altogether from the analysis, as they were insufficiently defined.

4.5.1.4 Implications of allocation choices

The significant influence that different interpretations of flow and process allocation have on CE assessment results is demonstrated by the wide range of values that the circularity indicators mentioned in Chapter 4.3 may take (see Table 4.1). This is especially true for the Austrian P-N system, where, on the one hand, greater flow differentiation allows for a wider variety of combinations, and on the other hand, large mass flows (i.e., manure and environmental emissions) are particularly ambivalent. By considering all types of soil as part of the environment and consequently inputs to soil as emissions the lowest values of circularity are produced, while at the opposite end of the range all environmental emissions are considered to be a form of recycling. The lower bound in the column “range under common assumption” refers to a case where process allocation is not changed with respect to the assumptions described in Chapter 4.3, but where only flows that physically enter the waste management sector are regarded as recycling. In the European copper system, circularity is lower if fabrication scrap is regarded as a co-product, and to a lesser extent, if the scrap stock is located in the waste management sector. High values of circularity result if all flows indicated as “losses” in the paper by Soulier et al. (2018) were allocated to emissions, a perception which is not very likely to reflect reality, as mentioned in Chapter 4.5.1.3. For the Karlskoga CHP, room for interpretation was only encountered for the input of wood chips, which could either be regarded as product or waste import, and for steam, which could alternately be defined as a co-product or a (recycled) by-product.

Table 4.1 also empahsizes the importance of careful indicator definitions. Although all four indicators measure circularity, in terms of the degree to which the system can be considered to be a closed loop, results vary considerably, depending on whether the mass of recycled materials is related to input of virgin material, waste and emission output, or total system material throughput, as well as whether gross or net inputs are considered. The informative value of an indicator may vary in different circumstances. For instance, in a system where products are quickly transformed into waste, such as the Austrian P-N system, differentiation between CUR and OSCr may be less important than in a system with a considerable buildup of consumption stock. At the level of a production site, where hardly any consumption within the system boundaries occurs, subtraction of export flows, as is the case for ISCr, may not seem reasonable, and for energy flows in incineration plants, where energy is dissipated to the environment rather than collected by the waste system, the calculation of RR is not expedient. Nevertheless, in order to facilitate communication of the CE concept to a broader public, circularity indicators should be harmonized as much as possible. Efforts to improve the meaningfulness of CE indicators are ongoing (Geyer et al. 2016; Hashimoto and Moriguchi 2004; Haupt et al. 2017) and should be continued in the future. However, they will not be explored further in this chapter as the focus is on comparability rather than harmonization.

Tab. 4.1: Range under all potential and under commonly made assumptions as well as results for the assumptions made in the present analysis for the circularity indicators cyclical use rate (CUR), input socioeconomic cycling rate (ISCr), output socioeconomic cycling rate (OSCr), and recycling rate (RR). Indicator definitions according to Equations (4.1)–(4.4).

Indicator	Range	Range under common assumptions	Assumptions of the present study
Austrian P-N System			
CUR P	7-0%	12-54%	54%
CUR N	2-46%	3-25%	25%
ISCr P	11-301%	28-236%	236%
ISCr N	4-158%	5-61%	61%
OSCr P	10-76%	22-70%	70%
OSCr N	4-61%	5-38%	38%
RR P	41-90%	41-85%	85%
RR N	17-98%	17-69%	69%
EU 28 copper system			
CUR	16-27%	26-27%	27%
ISCr	40-76%	72-76%	76%
OSCr	34-50%	48-50%	50%
RR	34-69%	48-51%	51%
Karlskoga CHP			
CUR	39-51%		39%
ISCr	468-770%		468%
OSCr	82-89%		82%
RR	100%		100%

4.5.2 Advantages of the generic structure

The ability to refer complex, differently constructed systems to a common structure facilitates comparability of MFA-based indicators and may thus contribute to clarifying the definition of the CE concept.

Experience gained in the course of this study has shown that each system comes with specific predicaments, and therefore it is deemed impossible to formulate a generic structure that fits all and that eliminates the need for any subjective assumptions. Even if all potential ambiguities could be envisioned in advance, it is not desirable to base assessment on a rather arbitrary selection of one out of several equally valid conventions. This is not only a problem pertaining to MFA. Due to its high computational and data demand, life cycle assessment is often applied in a simplified manner. Simplifications, however, are unique to each situation and need to be based on assumptions. The weighting of different goals against each other is another highly subjective matter (Jayal et al. 2010). Similarly, Saidani et al. (2017) observed that users of CE tools tend to favor middle options in trinary-based questionnaires, while binary scoring systems are often too reductive to represent system complexity adequately. In fact, finding the one "true" model may not even be necessary. As Ashby previously pointed out in 1958 information quantities in complex systems may exceed human capacity (Ashby 1958). However, even when a system is not fully understood, valuable conclusions can be drawn from its study, if scientists accept that solutions are never final, and only valid until they become obsolete. Under such circumstances it is clear that the transparency of subjective choices is crucial to ensure system comparability. The indicator values in Table 4.1 do not disclose the assumptions taken to produce them and require further explanation. However, intuitive understanding of whether a flow should be regarded as a co-product or a by-product or where to place a certain process in the generic system often exists, and therefore authors might be tempted to not explicitly state what seems obvious to them. Nevertheless, as Korhonen et al. (2018) pointed out, perceptions as to the point at which material with economic value becomes waste with no or negative economic value vary, depending on culture, society, community, history, and level of societal development, and may change over time. Here, the concept of a common structural basis and specification in subsystems, where aggregation choices are automatically disclosed, provides a clear advantage. An account of the underlying system structures and flow values, for the numbers shown in Table 4.1, is given in the Appendices A and in the enclosed electronic supplementary material.

Arguably, transparency of choices could also be achieved by the meticulous documentation of all calculation and aggregation steps, for instance, as has been done by Mayer et al. (2018). However, a multilevel structure is preferable because it facilitates uncertainty assessment. STAN, the software applied in the present study, uses weighted least square minimization for data reconciliation and error propagation (Cencic 2017), which means that flows of the balanced system, respectively their uncertainties, are correlated (Cencic 2016). Therefore, propagation of data uncertainties for manual flow aggregation is very complex, as Equation (4.5), where $\mathrm{s_{agg.}}$ and $\mathrm{s_i}$ stand for the standard uncertainties of the aggregated, and individual flows, respectively, is only applicable to independent values of $\mathrm{s_i}$.

$$s_{agg} = \sqrt{\sum_{i=1}^{n} s_i^2} \tag{4.5}$$

This problem can be avoided if calculation, data reconciliation, and flow aggregation are performed in a single step, as is the case for the generic system. To illustrate this, Table 4.2 summarizes independent input data and reconciled values for F03 “environmental deposition/extraction” of the Austrian N system in two circumstances. In the first, reconciled flows are manually aggregated from the original structure presented in Chapter 3, using Equation (4.5) as a proxy for the standard uncertainty, whereas, in the second the generic structure and subsystems are applied. As both systems use the same input data, results after data reconciliation for individual flows are identical. However, manual flow aggregation overestimates true uncertainty of F03.

The importance of resource interactions is increasingly recognized in the scientific community (Font Vivanco et al. 2018; Giurco et al. 2014; Liang et al. 2019). The study of such interactions is encouraged in the system structure presented here since goods and multiple substances can be added as different layers of the same system and balanced simultaneously. Therefore, co-benefits and trade-offs can be directly accounted for. Contrary to economy-wide MFA, where resource differentiation is usually carried out at a rather coarse level, such as that of biomass, metals, construction minerals, and fossil fuels (Kovanda et al. 2012; Mayer et al. 2018; Raupova et al. 2014), the generic system described here enables the study of total mass as well as a focus on individual resources of particular interest.

Furthermore, awareness for the necessity of an integrated supply chain management, and therefore joint consideration of product, process, and system level over multiple life cycles is growing (Jayal et al. 2010). Both these aspects are supported by the multilevel structure of the generic framework and the possibility for multiyear assessment, respectively.

Finally, the consistent application of a common reference framework during the system design may help to avoid incomplete balances, as was the case for mass flows in the Karlskoga CHP.

Tab. 4.2: Input data and reconciled values for environmental deposition/extraction in the Austrian N system when manually aggregating individual flows, using Equation (4.5) as a proxy for standard uncertainty and when computing value and uncertainty of the aggregated flow in the generic structure using subsystems.

Flow	**A priori input data**		**Reconciled values manual aggregation**		**Reconciled values generic system**	
	N[t]	**s [t]**	**N[t]**	**s [t]**	**N[t]**	**s [t]**
Atmospheric deposition agriculture	49,114	±14,210	47,449	±12,407	47,499	±12,407
N-fixation agriculture	66,159	±9,599	65,320	±8,899	65,320	±8,899
Atmospheric deposition forestry	58,370	±11,744	59,951	±7,585	59,951	±7,585
N-fixation forestry	8,569	±2,718	8,654	±2,657	8,654	±2,657
Atmospheric deposition urban areas	4,719	±1,449	4,720	±864	4,720	±864
Technical N-fixation	193,400	±36,305	190,026	±20,787	190,026	±20,787
Environmental deposition/extraction*			**376,120**	**±33,132**	**376,120**	**±19,162**

* The flow Environmental deposition/ extraction (F03 in the generic system) is calculated as the sum of the flows Atmospheric deposition agriculture, N-fixation agriculture, Atmospheric deposition forestry, Atmospheric deposition urban areas and Technical N-fixation, as can be seen in Figure A.47 in Appendix A.1.2.

4.5.3 Limitations

As previously mentioned in Chapter 4.1, the generic MFA structure presented in this chapter is intended to support CE, resource efficiency, and sustainability assessment, while it cannot fully abolish the need to take other factors into account.

CE indicators increasingly recognize aspects of product lifetime and environmental impacts (Ellen MacArthur Foundation and Granta Design 2015; Figge et al. 2018; Huysman et al. 2017), especially at a product and company level. Mayer et al. (2018) argue that strategies like extended product lifetime, reuse, and sharing find expression in lower in-use stock growth and are thus indirectly measurable by means of MFA. Similarly, MFA-based extended statistical entropy analysis can provide insights into the environmental impacts of emissions of different materials (Sobańtka et al. 2012). Nevertheless, both in-use stock growth and statistical entropy should only be used as the first estimates on the basis of which further assessment for in-depth understanding may be conducted.

Other issues that are currently much discussed in CE are entropy and rebound effects. Recycling technologies may require high input of energy so that the environmental net benefits of material recovery may be low (Korhonen et al. 2018). For example, 1.1% of global energy use can be attributed to technical N-fixation via the Haber–Bosch process (Dawson and Hilton 2011), which should be taken into consideration if atmospheric N_2 emissions were to be regarded as a form of circular use. Moreover, cheap secondary materials may increase demand for materials not covered by the analysis or in regions outside its system boundaries or lead to decreased recycling of such materials or in such regions. Market mechanisms play another crucial role, particularly in scenario analysis, because they define the degree to which recycled material is actually used to decrease primary input (Geyer et al. 2016; Zink and Geyer 2017). Furthermore, spatial planning and consideration of transitional costs, such as sinking employment rates in industries that find themselves in competition with CE-based alternatives, are essential for successful CE implementation (Avdiushchenko 2018). These aspects are currently not reflected in MFA.

Chapter 5

Simple indicators for assessment of coupled resource systems

5.1 Indicator description

5.1.1 Circularity

Circularity (C), in physical terms, quantifies the amount of material reintroduced into the production chain after use, as opposed to material emitted to the environment or deposited in final sinks such as landfills. There are multiple definitions of circularity in MFA systems. For instance, it can be reported as the share of recovered end-of-life material in total material consumption (Cullen 2017; Hashimoto and Moriguchi 2004), in the sum of cyclical and virgin material input to the production process (Kovanda et al. 2012), in domestically processed material or in the sum of end-of-life waste and emissions (Mayer et al. 2018). For the purpose of the present study, a particularly comprehensive definition of C was chosen:

$$C = \frac{mass\ of\ recirculated\ substance/material}{mass\ of\ total\ substance/material\ input\ (incl.\ recirculation\ and\ stock\ mobilization)} \tag{5.1}$$

Thus, C is a percentage value, where 0% corresponds to a fully linear system and 100% to the theoretical optimum of a perfectly circular one, as illustrated in Figure 5.1. Although the flaws of C, such as incapability to account for material use time and efficiency of use, ambiguities regarding the treatment of by-products in the analysis and disregard of environmental impacts of recycling processes and of rebound effects are all well-known (Cullen 2017; Hashimoto and Moriguchi 2004; Korhonen et al. 2018), its simplicity still makes it a popular component of CE indicator sets (e.g. Government of Japan 2013; Mayer et al. 2018; Pauliuk 2018) or even as a single lead indicator (e.g. Wit et al. 2019).

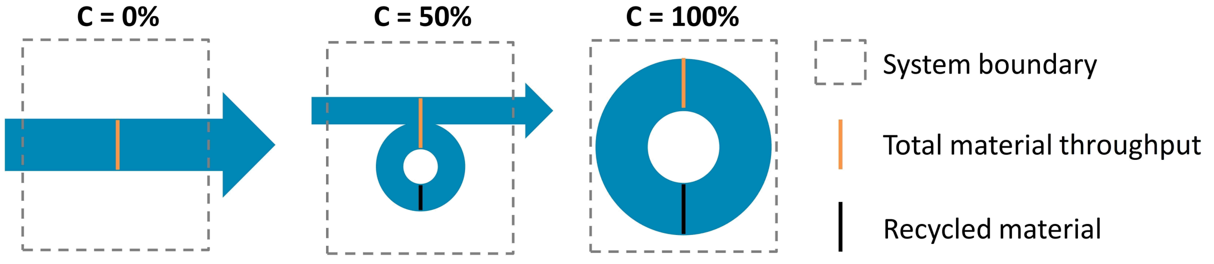

Fig. 5.1: Graphical illustration of the concept of C. The theoretical cases of a perfectly linear and a perfectly circular system are depicted to the left and right, respectively, while the picture in the middle shows a system with C=50%.

For the generic system introduced in Chapter 4.3, the flow "Recycled waste" corresponds to the nominator of Equation (5.1), while the sum of the flows "Import raw materials and goods", "Extracted raw material", "Environmental deposition/extraction", "Consumption deposition", and "Import waste" as well as stock extraction from the processes "Production", "Use", and "Waste management" forms the denominator. C is calculated with respect to net-values of import and export flows. The flow "Recycled waste" includes both recovered end-of-life material and by-products of production processes, provided that they are used as inputs in a production process other than the one they stem from.

5.1.2 Substance Concentration Efficiency

The Substance Concentration Efficiency (SCE) is based on statistical entropy (H), a concept developed by Shannon 1948 for information technology and adapted for resource management purposes by Rechberger and Brunner (2002). It measures the amount by which a substance is diluted or concentrated in a system, where a higher degree of dilution is assumed to go along with more adverse environmental effects and lower recyclability. The concept has, for instance, been applied for the evaluation of waste incineration technology (Rechberger and Brunner 2002), the fate of carbon in waste incineration (Kaufman et al. 2008), the Chinese copper cycle (Yue et al. 2009), sewage sludge treatment options (Lederer and Rechberger 2010), waste water treatment plants (Sobańtka and Rechberger 2013), regional N budgets (Sobańtka et al. 2014), lead smelting (Bai et al. 2015), P use (Laner et al. 2017) and lithium ion battery waste (Velázquez Martínez et al. 2018). Furthermore, the indicators Recyclability (Fang et al. 2018; Zeng and Li 2016) and Potassium Utilization Potential Factor (Vakalis et al. 2017) are based on H, and Luo et al. (2018) develop it further to assess the hazardous potential and resource potential of different MFA flows. H is defined as:

$$H = -\sum_{i=1}^{k} \frac{X_{ij}}{\sum_{i=1}^{k} X_{ij}} * log_2(c_{ij}), \tag{5.2}$$

where $\mathrm{X_{ij}}$ is the total mass of substance j in the mass flow i and $\mathrm{c_{ij}}$ is the concentration of substance j in mass flow i. Thus, for a flow of pure substance H=0 and increases with decreasing concentrations. The formula and its derivation are described in detail in Rechberger and Brunner (2002). Application of H to real, complex resource systems requires several additional considerations, as described below.

5.1.2.1 Emission flows

Emissions are diluted in the receiving media, which increases their entropy. To account for this, a virtual mass flow is introduced, corresponding to the mass necessary to dilute the emitted substance to a concentration that lies 1% above the natural background concentration. In an approximation, $\mathrm{c_{ij}}$ for aqueous and gaseous emissions is therefore defined as

$$c_{ij} = \frac{c_{geog,jl}}{100}, \tag{5.3}$$

where $\mathrm{c_{geog,jl}}$ specifies the environmental background concentration of substance j in media l (Rechberger and Brunner 2002). To better reflect the environmental impacts of emissions in the present study, where available, imission limits for substance j in the environmental media l are used for $\mathrm{c_{geog,jl}}$ instead of natural background concentrations. Appendix B.2 gives an overview of the values used for the emissions in the present study as well as their origin.

5.1.2.2 Compound specification

In its original conception, statistical entropy in resource systems was formulated for chemical elements. Non-metallic elements in particular, however, occur in numerous specifications and compounds with very different environmental impacts. This has led to the development of extended statistical entropy, where each flow i is further specified into different compounds m of the substance j (Sobańtka et al. 2012). Equation (5.2) thus becomes:

$$H = -\sum_{i=1}^{k}\sum_{m=1}^{n} \frac{X_{ij}}{\sum_{i=1}^{k} X_{ij}} * log_2(c_{ijm}) \tag{5.4}$$

Consequently, environmental background concentrations have to be formulated with respect to substance compounds as well:

$$c_{ijm} = \frac{c_{geog,jml}}{100} \tag{5.5}$$

Kaufman et al. (2008) suggest an alternative approach in which a forcing factor accounts for the environmental impact of different emissions. However, as this causes disruptions in the mass balance and requires the introduction of imaginary mass flows at the system's input side, the methodology of Sobańtka et al. (2012) is more suitable for the given purpose.

Compound specification is only undertaken where considered relevant from an environmental impact perspective and closely follows compound specification in current emission and imission limits. Hence, Equations (5.4) and (5.5) are only applied to liquid and gaseous flows of N. A distinction between N_2 (inert), NH_3 (particulate matter formation, acidification, terrestrial eutrophication), NO_x (particulate matter and tropospheric ozone formation, reduction in human respiratory functions, acidification) and N_2O (greenhouse gas) in air (Sutton et al. 2011) is made. Likewise, due to different eutrophication potentials,NH_4, NO_3 and organic forms of N in water (Zeng et al. 2016) are distinguished.

5.1.2.3 Computation of SCE

H can be analyzed along a system's production chain. For that purpose, the MFA structure has to be transformed into a stage diagram (Rechberger and Graedel 2002). Figure 5.2 shows the stage diagram corresponding to the MFA structure of the generic system (Figure 4.3). H of each stage quantifies the distribution of the substance among the flows crossing it. Stock growth and depletion are interpreted as import and export flows to the process concerned, respectively. Recycling flows are considered both as import and export flows in order to keep a closed mass balance.

Because H is based on substance distribution within a set of material flows, aggregation of flows with different substance concentrations will lead to a loss of information and an increase in H (Laner et al. 2017). Therefore, instead of the flows shown in Figure 5.2, the disaggregated sub-system flows they are made of enter into the calculation. As for C, import and export flows are considered as net-values.

Typically, assessments focus on the change in entropy a substance undergoes within a system. This can, for instance, be quantified with the Substance Concentration Efficiency (SCE):

$$SCE = \frac{H_1 - H_4}{H_1} \tag{5.6}$$

H_1 and H_4 are the statistical entropies in the first and the fourth (final) stage, respectively. The maximum value of SCE=100% is reached when the entirety of the substance leaves the

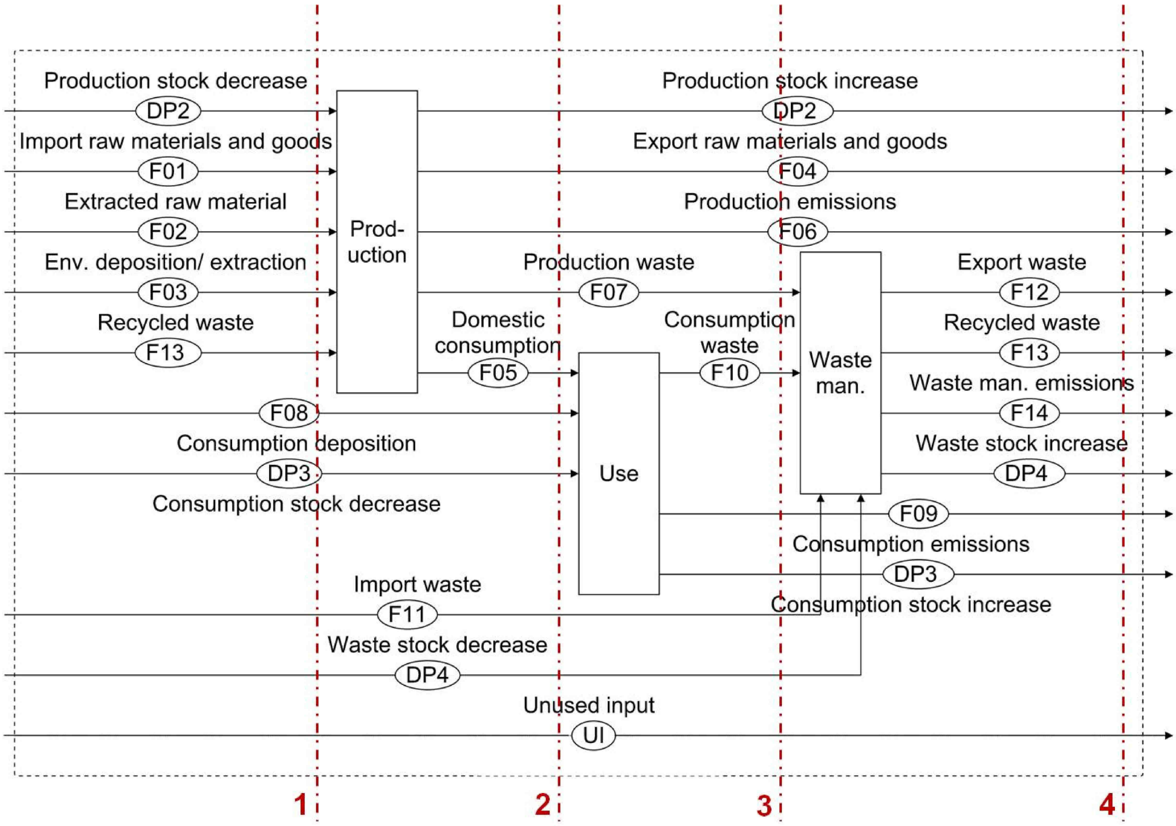

Fig. 5.2: Stage diagram of the Austrian P-N system. Red lines indicate the stages at which H is calculated.

system in a pure mass flow (H=0). SCE=0% means that the substance does not undergo any changes within the system in terms of entropy, and negative values of SCE correspond to the most common case of substance dilution e.g. due to emissions to the environment.

5.1.2.4 System turnover and downsizing

H in Equation (5.2) is normalized to total turnover and consequently does not reflect changes in absolute material throughput through the system, although downsizing the anthropogenic metabolism is regarded as a key factor to preserve valuable resources and ensure overall system sustainability (Akenji et al. 2016). This can be overcome by introducing an imaginary ("virtual") flow (Laner et al. 2017). A constant amount of each substance A_j is assumed to enter the system as input, which corresponds to the maximum system throughput encountered among all scenarios analyzed. A_j can be regarded as the functional unit of the scenario comparison. Each scenario x "utilizes" a part of this pool B_{jx} according to its system throughput. The rest remains in the imaginary flow, which is why it is referred to as unused input (UI):

$$UI_{jx} = A_j - B_{jx} \tag{5.7}$$

UI is directed through all stages of the system without passing any processes, as shown in Figure 5.2. Contrary to Laner et al. (2017), who assign it the same concentration as mineral fertilizer, it is considered to be a flow of pure substance and thus does not contribute to entropy.

5.1.3 Assessment benchmarks

For a meaningful comparison of C and SCE as indicators it is necessary to refer them to a common benchmark. The boundaries of C are absolute; values below 0% or above 100% are physically impossible. The minimum value for H is zero, corresponding to the entropy of a flow of pure substance. However, as flow concentrations can become indefinitely small, there is no upper boundary for H. Consequently, SCE has an absolute maximum at 100%, but is open in the negative realm. It is therefore not clear whether a higher percentage value of SCE would mean that the system performs better with respect to SCE than to C or vice versa.

Rechberger and Brunner (2002) solved this problem by introducing a maximum statistical entropy, H_{max}, defined as the H resulting if the entirety of substance in the system was directed to the environmental compartment with the lowest background concentration. If H at the system input and output is related to H_{max}, the resulting values for SCE are between -100% and 100%. Contrary to C, these boundaries are not absolute though, but depend to a large extent on the definition of the system. For instance, in the case study of the Austrian P-N system emissions to groundwater and emissions to surface water are distinguished, whereas the P system used to generate the time series only accounts for emissions to surface water. Because imission limits for P in surface water are about twice as high as in groundwater (see Appendix B.2), H_{max} in the first case would be considerably higher than in the latter. Likewise, a broader or refined specification of N would cause changes in H_{max}. This hampers comparability to the immutable boundaries of C. Besides, the likelihood of a situation occurring in which all substance in the system is directed to the environmental compartment with the lowest background concentration is highly improbable if not impossible in reality. Reference to H_{max} may thus conceal the differences in substance concentration or dilution within the system or between scenarios for more realistic situations with a considerably lower H.

For this reason, in addition to the 18 measure scenarios, a best-case and worst-case scenario for the Austrian P-N system is defined against which both C and SCE are assessed. These scenarios assume the same preconditions regarding climate and population as the status quo, but otherwise aim at making the system respectively as linear and diluting and as circular and concentrating as possible. For instance, the worst-case scenario assumes no treatment of wastewater, fertilizer losses between 50-90% of applied nutrient and unseparated deposition of all waste in landfills, whereas in the best-case scenario fertilizer losses are eliminated and all wastes are recycled, except for inorganic parts of residual waste, which are incinerated. A detailed overview of all flows in the best-case and worst-case scenario as well as the boundary values for C and SCE can be found in the enclosed electronic supplementary material. All values for C and SCE are normalized to the best and worst case as follows:

$$\overline{C_x} = \frac{C_x - C_{WC}}{C_{BC} - C_{WC}} \tag{5.8}$$

$$\overline{SCE_x} = \frac{SCE_x - SCE_{WC}}{SCE_{BC} - SCE_{WC}} \tag{5.9}$$

The indices x, WC and BC signify measure scenario x, worst-case scenario and best-case scenario, respectively.

Thus, normalized values of C and SCE are percentage values between 0% and 100%, where 0% corresponds to the worst case and 100% to the best case. The common reference basis also enables the computation of overall system values as averages of the individual values for each substance.

5.2 Application of C and SCE to the Austrian P-N system

Both C and SCE were mainly developed for assessments of metallic and mineral resources or other conservative substances that hardly undergo changes in physical form and species within the system. As described in Chapter 4 the application of such indicators to systems largely dominated by biomass flows, where consumption stocks play a negligible role and boundaries between industry and the environment are fluid, may not be straightforward. This makes P and N particularly interesting resources as a case study as it reveals information on the universality of the chosen indicators. Moreover, as N takes up multiple physical states in the system and occurs in various species with fundamentally different characteristics, whereas P is a more conservative substance, but considered as a critical raw material (Bureau de Recherches Géologiques et Minières et al. 2017), the indicators can be simultaneously evaluated with respect to their capability of quantifying efforts towards resource protection and towards environmental protection.

In addition to the information on the current state and effects of measure scenarios described in Chapter 3, the availability of a time series of stock and flow data for P for the years 1990-2013 (Zoboli et al. 2015) made it possible to analyze C and SCE for this substance with respect to their development over time.

5.2.1 C and SCE in the status quo

Tab. 5.1: Results for Circularity (C) and Substance concentration efficiency (SCE) in the Austrian P-N system for the status quo 2015. Overlines indicate normalized values; system results are computed as averages of P and N for the respective indicator.

	C	$\overline{C}$	SCE	$\overline{SCE}$
P	45%	50%	-47%	87%
N	15%	35%	-54%	63%
System		42%		75%

Table 5.1 gives an overview of the absolute and normalized results for C and SCE achieved in the Austrian P-N system in 2015. Full computation and analysis, also for the evaluation of the timeline and measure scenarios, are provided in the enclosed electronic supplementary material. Both P and N perform better if evaluated with respect to $\overline{SCE}$ than with respect to $\overline{C}$. The analysis of development of $\overline{C}$ and $\overline{SCE}$ since 1990 indicates that this may be attributed to the longer history of efforts for environmental protection and emission reduction compared to natural resource conservation and recycling (Dovers 2013). For instance, the connection rate to wastewater treatment plants increased from 71% in 1990 to 95% in 2015, while removal efficiency rose from around 50% to 82% for N and 90% for P (Überreiter et al. 2016). Consequently, annual per capita emissions of P in waste water treatment plant effluents could be decreased from 1.02 kg/cap in 1990 to 0.12 kg/cap in 2015 (see Figure 5.3a). Nutrient surplus in agriculture could also be reduced significantly over the past 30 years (Eurostat 2019), which is reflected in the decreasing amount of P added to agricultural soil stocks (Figure 5.3a). Through these achievements, emission flows with particularly low concentrations could be reduced. Hence, $\overline{SCE}$, as depicted in Figure 5.3b, increased. For $\overline{C}$ this effect is less pronounced in the first 10 years of the analysis. However, a steep increase can be noticed in the early 2000s (Figure 5.3b) due to the development of the bioenergy sector, which introduced additional recyclable by-products such as biogas digestate into the system.

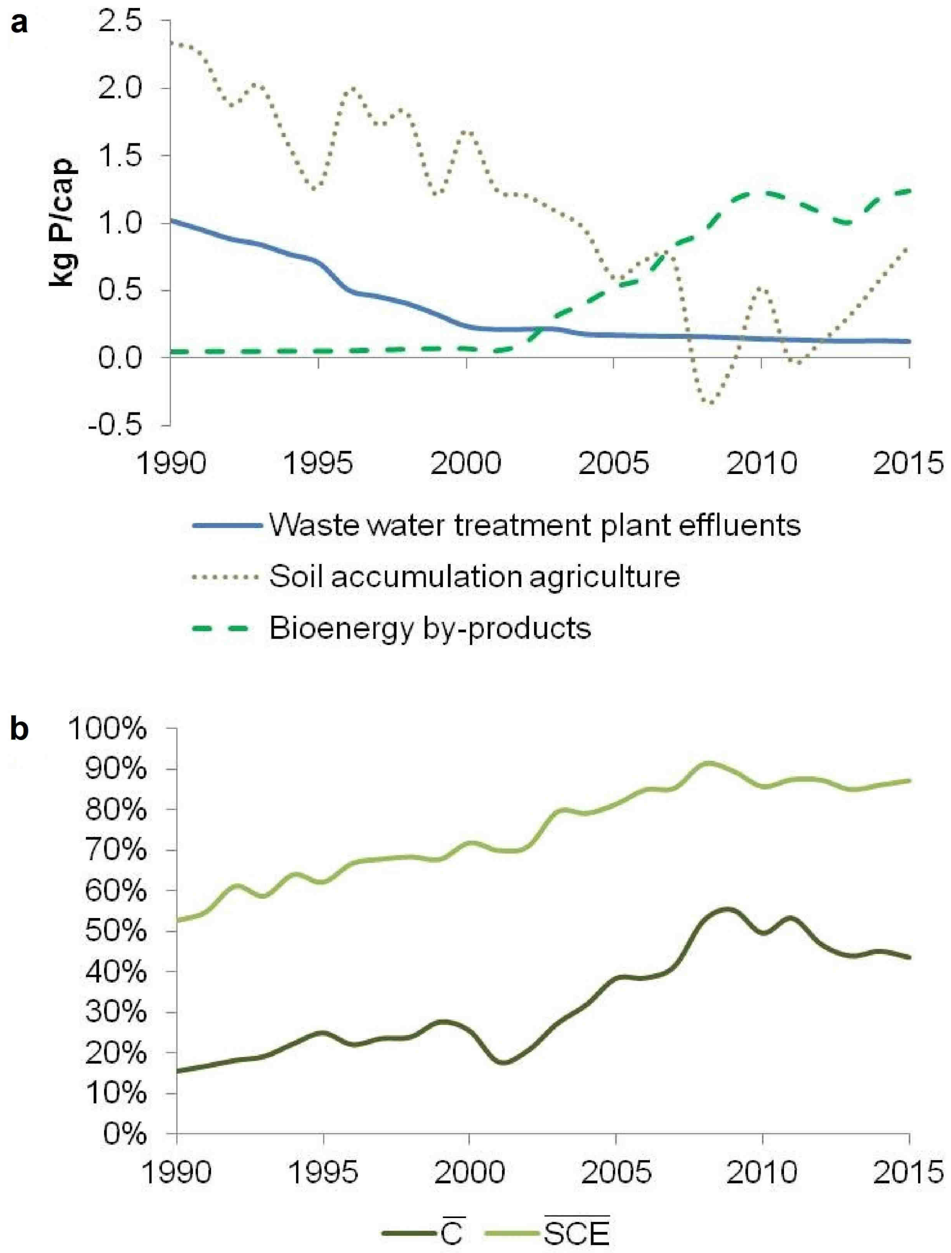

Fig. 5.3: a) Development of selected parameters in the Austrian P system 1990-2015. Data for the years 1990-2013 from Zoboli et al. 2015. b) Development of $\overline{C}$ and $\overline{SCE}$ in the Austrian P system 1990-2015.

The different characteristics of P and N are reflected in the evolution of their statistical entropies in the system, as shown in Figure 5.4. P mainly enters the system as import of fertilizer and food, whereas for N atmospheric deposition also plays a significant role. The latter occurs in a highly diluted form so that H in Stage 1 is already considerably higher for N (5.9) than for P (0.8; see enclosed electronic supplementary material). Furthermore, N plateaus after Stage 2. P, on the other hand, has a distinct peak in Stage 3 followed by a reduction in H during waste management. This is a result of the limited recycling potential of N due to the volatilization of large parts of N during common waste management practices such as incineration and composting. Although the different characteristics of the two substances are accounted for in the normalized values, results for $\overline{C}$ and $\overline{SCE}$ in Table 5.1 indicate that currently P is diluted less in the system and circulated more. Agricultural NH_3-emissions stemming from manure storage and fertilization, in particular, as well as nitrate leaching to groundwater present continuous problems as they each account for 30% of output entropy (see enclosed electronic supplementary material). The virtually unlimited supply of N in the atmosphere and the low cost of mineral N-fertilizer production (Sutton et al. 2011) combined with effective flue gas purification during waste incineration, where hazardous N-species are converted into N_2 (Böhmer et al. 2007; Stoiber 2017), may be the reason why N-recycling is currently less in focus than P-recycling.

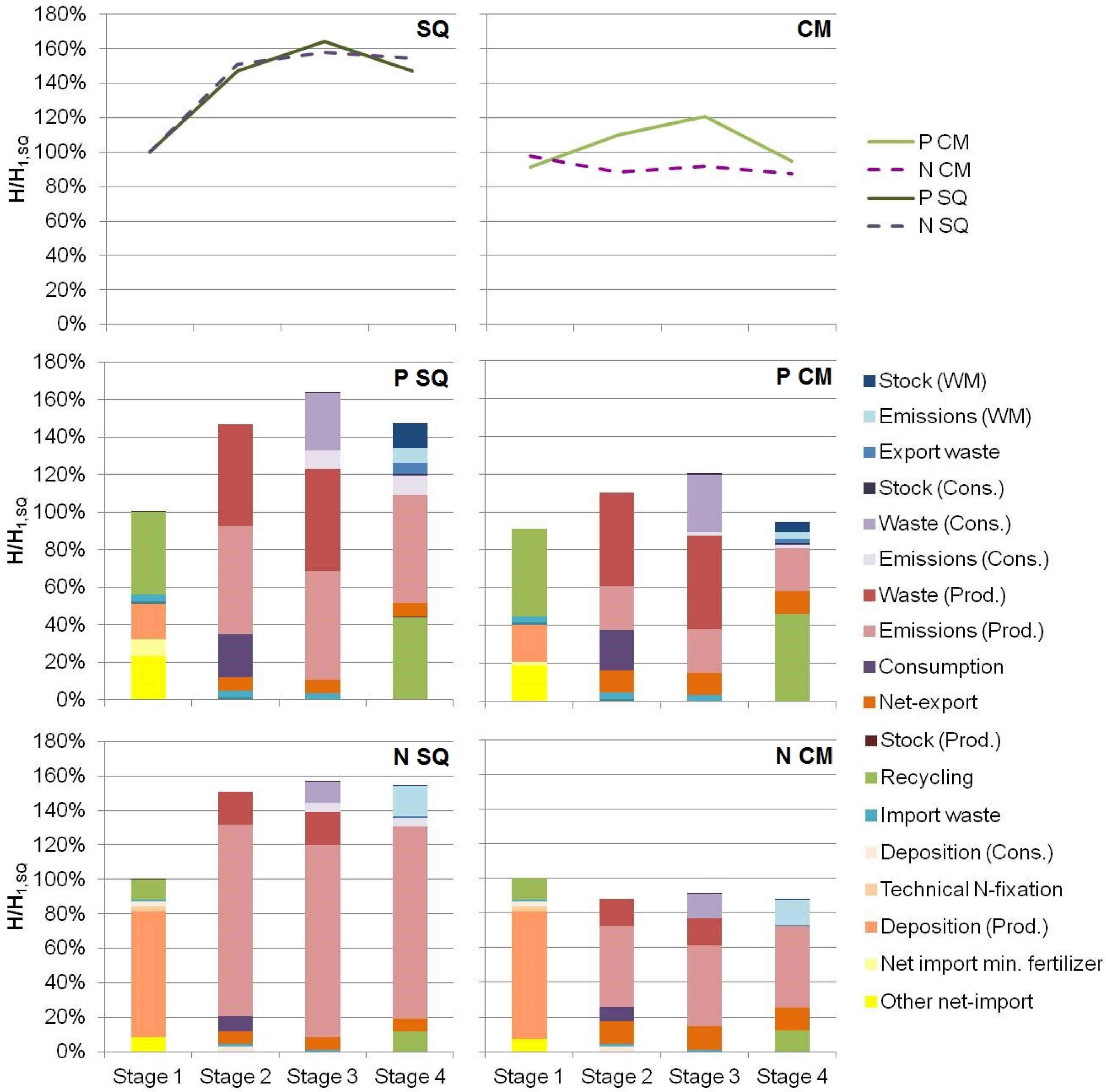

Fig. 5.4: Evolution of statistical entropy (H) over the four stages of the system (top) and breakdown to different components for P (middle) and N (bottom) for the status quo (SQ, left) and the combined measures scenario (CM, right). Values are depicted in relation to H in Stage 1 of the status quo.

5.2.2 C and SCE in the measures scenarios

The differences between P and N in the evolution of H through the stages of the system described in Chapter 5.2.1 are even more pronounced when looking at the combined measures scenario (implementation of all measures described in Chapter 3.1.2). As can be seen in Figure 5.4, reduction potentials for H are highest in Stage 2 for both substances, which can mainly be attributed to a decrease in production emissions. Reduction potentials for N in other stages are negligible. Nevertheless, the decrease in Stage 2 is with 60% so high that a shift in the system from a diluting to a concentrating one occurs ($H_1 > H_4$). For P, H is reduced in all stages and, in the final stage, H_4 almost matches input entropy H_1 again.

Overall, $\overline{SCE}$ of the system can be increased from 75% to 90% (see enclosed electronic supplementary material). $\overline{C}$ rises by 15% as well and reaches 57% in the combined measures scenario. Clear differences between the two indicators can be observed, however, when looking at the key factors causing these improvements:

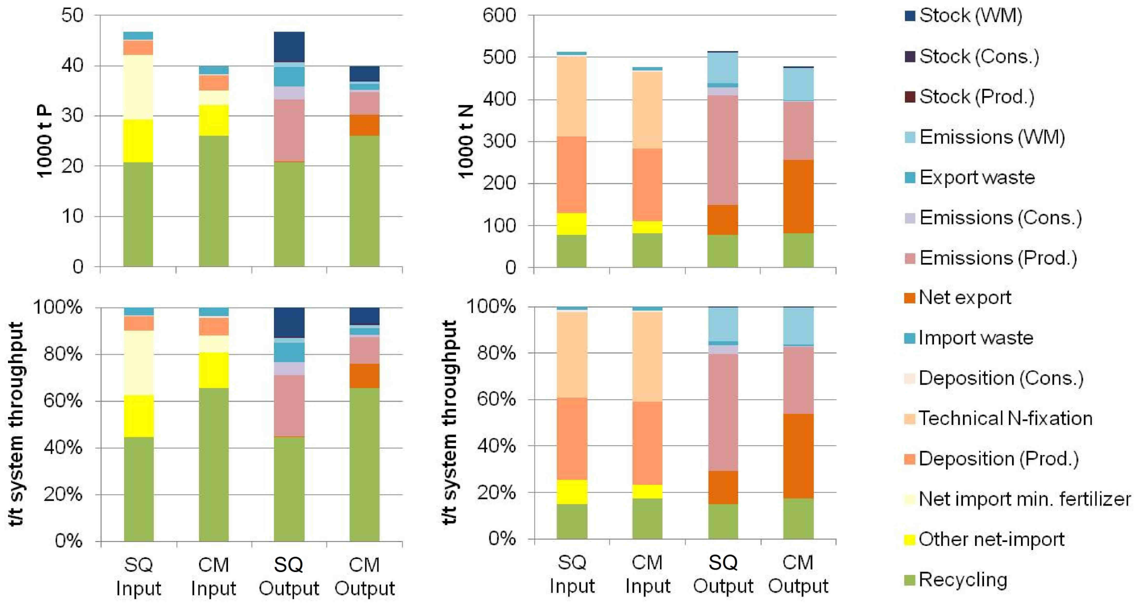

Fig. 5.5: Top: Contribution of different flows to mass throughput at the system's input (Stage 1) and output (Stage 4) side in the status quo (SQ) and combined measures scenario (CM) for P (left) and N (right) in absolute terms. The length of the bar signifies total system throughput. Due to the principle of mass conservation, system throughput is equal at the input and output side for each scenario. However, a reduction in system throughput occurs from SQ to CM. Different flows contribute to different extents to these reductions. Bottom: Relative contributions of different flows with respect to system throughput. Percentages for recycling in the lower diagrams are equivalent to C (share of recycling on total system throughput).

Comparing the absolute and relative contributions of different components to the system input in the status quo and the combined measures scenario (Figure 5.5) reveals that changes in C for P mainly stem from reductions in net-import of mineral fertilizer (-78%) and, to a lesser extent, increased recycling. Because mineral fertilizers have high nutrient concentrations, the remarkable reductions in mass have a low effect on H in Stage 1 compared to the changes between the status quo and the combined measures scenario occurring in the subsequent stages, as can been seen in Figure 5.4. Replacement of mineral fertilizer with lower concentrated recycled materials such as compost or meat and bone meal (MBM) may even lead to an increase of H in Stage 1, as is, for instance, the case in the scenarios "Increased nutrient recovery from MBM" and "Improved collection of organic household waste (see material). A decrease of 37% in domestic mineral fertilizer consumption can also be observed for N (see enclosed electronic supplementary material). However, due to the possibility of domestic production, Austria is a net-exporter of mineral N-fertilizer and thus these reductions manifest in an increase of net-export rather than on the input side of the system (see also Chapter 5.3). Like for P, changes in net-export are more pronounced in terms of mass than when looking at H. Improvements in $\overline{SCE}$ therefore mainly stem from reductions in emission flows with low environmental background concentrations.

The different characteristics of both the substances and the indicators also manifest themselves when looking at the performance of individual measures (Figure 5.6). For P, higher achievements in $\overline{C}$ than in $\overline{SCE}$ can be reached, whereas the opposite is true for N. This can be attributed to the fact that improvements in $\overline{C}$ are predominately produced by measures aimed at increasing recycling, while recycling potentials for N are limited. In the N system, demand- and emission reduction measures increase $\overline{SCE}$, but do not change or even decrease $\overline{C}$. An exception to this is the measure "Exploitation of maximum fertilizer efficiency". Classified as a demand-reduction measure, it is clearly the most effective individual measure, irrespective of the substance and indicator it is evaluated against. It should be noted, however, that some measures address several aims simultaneously so that the selection of the main aim is to some extent arbitrary. "Exploitation of maximum fertilizer efficiency" for instance both decreases mineral fertilizer demand and reduces agricultural emissions.

Figure 5.6 also shows that, in some cases, indicator choice determines whether a measure is regarded as an improvement or deterioration compared to the status quo. For instance, the measure "Full application of P- and N-optimized feed for cattle, pig and poultry" in the P system assumes higher nutrient use efficiency in livestock and thus less nutrient loss with manure. However, as manure is an important fertilizer in crop farming, the resulting deficit has to be filled by an increased application of mineral fertilizer, if not accompanied by increases in fertilizer efficiency. Consequently, $\overline{C}$ decreases by 1%. On the other hand, substitution of manure increases $\overline{SCE}$ by 1% due to the low entropy of mineral P-fertilizer. The opposite situation can be found for the increased recycling of MBM. MBM classified as category 3 material can be directly applied on fields (European Commission 2009) so that $\overline{C}$ under this measure increases by 3% for P and 2% for N. Yet, MBM has a lower fertilizer efficiency compared to mineral fertilizer, especially for P. Substitution of mineral fertilizer with MBM thus increases emission flows. The low background concentrations in water and in the atmosphere are the reasons why increases in H are more pronounced in Stage 4 than in Stage 1 and why $\overline{SCE}$ decreases. Another trade-off can be seen in the measure "Increased P-recovery from sewage sludge". Contrary to MBM, it is assumed that sewage sludge is incinerated prior to recovery of P fertilizer. While this is the most efficient individual measure to increase $\overline{C}$ for P (+7%), N is lost during incineration hence $\overline{C}$ decreases compared to the status quo (-1%), where part of the sewage sludge is directly applied on agricultural fields. In terms of $\overline{SCE}$, on the other hand, no trade-off between P and N can be observed. This is because the majority of incineration emissions occur in the form of N_2, which, due to the high presence of N_2 in the atmosphere, do not have a significant effect on H.

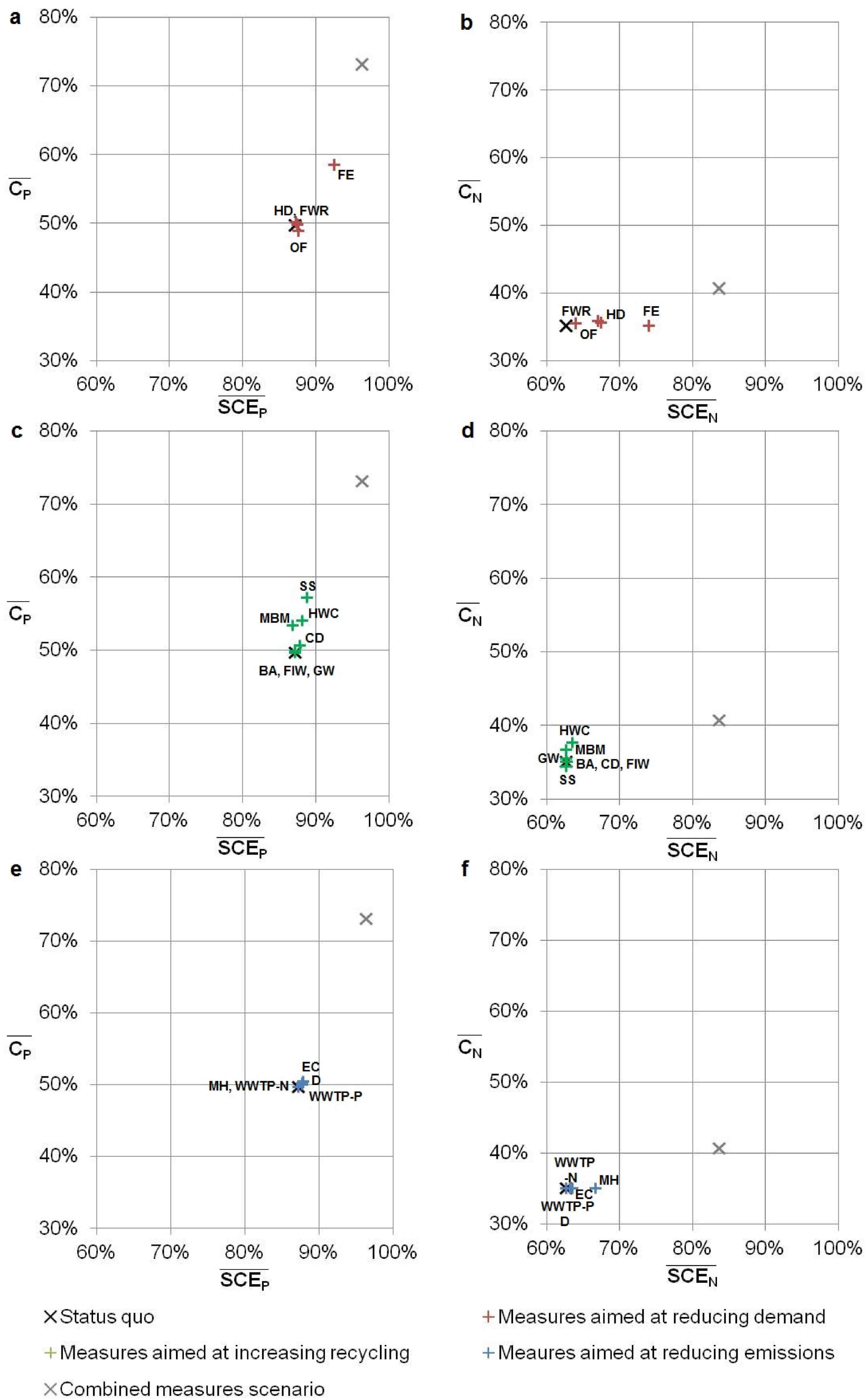

Fig. 5.6: Performance of $\overline{C}$ and $\overline{SCE}$ for P (left) and N (right) of individual measure scenarios grouped according to their main aim. Measures aimed at reducing nutrient demand (a,b), measures aimed at increasing Recycling (c,d) and measures aimed at reducing emissions (e,f). Abbreviations of measures as stated in Chapter 3.1.2.

5.3 Comparison of C and SCE as simple resource management indicators

Results in Chapter 5.2 show two clear advantages of $\overline{SCE}$ over $\overline{C}$: Due to the fact that environmental background concentrations or emission limits are usually several orders of magnitude lower than concentrations in solid flows, emission reductions are valued higher in $\overline{SCE}$ than for $\overline{C}$, which only looks at the input side of the system. This becomes evident when comparing the composition of mass throughput in the status quo and the combined measures scenario for N in Figure 5.5: Improved fertilizer efficiency and reduced demand for agricultural products following a change in consumption behaviour lead to a decrease in domestic fertilizer use as well as in agricultural emissions in the combined measures scenario. As Austria is a net exporter of mineral fertilizer, the reduction in gross fertilizer imports is reflected in an increase in net exports. Consequently, the ratio between net-export and production emissions changes from 22:78 in the status quo to 56:44 in the combined measures scenario. However, $\overline{C}$ does not differentiate between different parts of the system throughput other than with respect to recycled materials and therefore fails to account for this major shift in the production system. Similarly, increases in process efficiency that lead to higher production and lower emissions at same input levels are not reflected in $\overline{C}$. In the present case study constant end-consumption and trade levels are assumed for all measure scenarios so that emission reductions are indirectly reflected in $\overline{C}$ through a decrease in primary resource inputs in the denominator of Equation (5.1). However, even in this case $\overline{C}$ cannot differentiate between emissions with lower and higher environmental impacts as all flows are considered exclusively with regard to their mass. Evidently, impacts of a substance emitted to the environment do not only depend on concentration, but also on toxicity, location of emission, etc. Nevertheless, there is some indication that a correlation between H generation and severity of environmental impacts exists, especially if compound specification is undertaken (Rechberger and Brunner 2002; Sobańtka et al. 2014).

Another main advantage of $\overline{SCE}$ is the ability to account for dematerialization effects, as described in Chapter 5.1.2.4. Because unused input is regarded as pure substance and doesn't contribute to entropy, system downsizing will always lead to a decrease in H. Usually, output flows are more diluted than flows at the system input so that decreases due to dematerialization are more pronounced in the output state and subsequently $\overline{SCE}$ will increase. For $\overline{C}$, on the other hand, a trade-off exists as system downsizing will lead to a reduction in waste in absolute terms and thus a reduction in material potentially available for recycling. This can again be seen for N in Figure 5.5, where system throughput can be reduced by 7% between the status quo and the combined measures scenario, whereas the ratio between recycled materials and other components remains more or less unchanged.

However, looking at the group of measures primarily aimed at recycling, $\overline{C}$ might provide more meaningful results as $\overline{SCE}$ does not differentiate between flows stemming from recycling and primary inputs provided that they have the same concentrations. Like process efficiency gains in $\overline{C}$, recycling efforts will only indirectly affect $\overline{SCE}$, for instance through a reduction of unrecovered waste and emissions with lower concentrations than the recycling flow in the output stage. Potentials for resource protection of critical raw materials may thus be overlooked if only evaluated with respect to $\overline{SCE}$. In the case of fertilizers, which are the main output of recycling in the present case studies, primary products typically have considerably higher concentrations than recycled products. Therefore, substitution can even lead to a decrease in $\overline{SCE}$, as could be shown for P in the measure "Increased nutrient recovery from MBM" (see Chapter 5.2.2).

Regarding the consideration of N_2 emissions, the picture is less clear. On the one hand, N_2 emissions are irrelevant from an environmental perspective, which is reflected in the high concentration and thus low contribution to entropy assigned to N_2 flows. On the other hand, production of mineral fertilizer by conversion of N_2 to reactive forms via the Haber-Bosch process is highly energy intensive (Dawson and Hilton 2011). It is therefore preferable to keep N in the system for as long as possible. This is indirectly reflected in $\overline{C}$, where N_2 emissions are accounted for as linear flows.

In conclusion, both $\overline{C}$ and $\overline{SCE}$ capture the gains in sustainability when moving from the status quo system to the combined measures scenario in the Austrian P-N system. In terms of communicability, $\overline{SCE}$ might require more explanation than $\overline{C}$; nevertheless, the expression as a percentage value ensures straightforward understanding of indicator development over time or of juxtaposition of several scenarios. In this respect careful selection of a common reference basis is essential as it ensures comparability between different indicators and substances and enables the combination of individual results of substances in a coupled system into an overall system value.

Although results correspond well in their overall conclusion, $\overline{C}$ and $\overline{SCE}$ clearly prioritize different aspects of sustainability: While the increase in recycling activities is the main focus of $\overline{C}$, emission reductions, system downsizing and gains in process efficiency are better reflected in $\overline{SCE}$. Issues of resource criticality might be overlooked by $\overline{SCE}$ as primary resource flows often have low entropies and their substitution by recycled materials has small or even negative effects on $\overline{SCE}$. Nevertheless, considering that closing loops is not an aim in itself but rather a strategy to reach other sustainability goals (Bocken et al. 2014), $\overline{SCE}$ might be a more meaningful indicator for most assessments, especially if reactive, volatile substances are involved.

Finally, it is evident that neither $\overline{C}$ nor $\overline{SCE}$ can fully replace holistic in-depth evaluation methods such as LCA. In particular, issues related to energy demand of different recycling options and economic costs should also be taken into consideration as, for instance, delineated in Rechberger and Brunner 2002. However, simple straightforward indicators such as $\overline{SCE}$ offer a clear advantage by helping gain first insights into critical points in a system, pinpointing needs for more detailed investigations, providing preliminary estimations on a system's or substance's sustainability in comparison with other systems or substances or over different states in time. The benefits of such simple indicators are especially obvious when confronted with large and complex systems where sophisticated assessment methods often reach their limits in terms of data demand.

Chapter 6

Conclusion and Outlook

This thesis provides a thorough analysis of the potentials to integrate issues of resource interactions in MFA in the context of the Austrian P and N system. It shows, how coupling can be achieved by linking the substances in a flow or stock via its total mass, i.e. the goods layer. Although coupling significantly raises model complexity, material flows of more than one substance can be simultaneously analyzed in a rather complex system. The first aim of the study stated in 1 was therefore met and transition from mono-substance to multi-substance MFAs with the tools at hand (i.e. the MFA-software STAN) should thus be possible also for other resources.

Addressing the second aim of the study, a multi-level assessment framework is presented, where the topmost level constitutes a generic structure applicable to all resources and regions and on all scales, while study-specific flows and processes can be included in subsystems. This ensures greatest possible comparability without compromising level of detail.

Uncertainties, both in terms of input data and assumptions on model structure are also discussed. As MFA input data usually stem from diverse sources and information on data quality and uncertainty is often lacking, systematic characterization approaches are crucial for meaningful uncertainty assessments. Entering input data as directly as possible into the MFA system without the need for intermediate calculation steps supports such a systematic assessment. Moreover, it reduces the risk of suggesting false independencies during data reconciliation and thus an underestimation of uncertainties. However, as it often requires a splitting of flows, a trade-off with increasing model complexity exists. This is particularly relevant in coupled models, because already the integration of several substances into one model considerably raises complexity compared to a single substance analysis.

In terms of model structure real systems prove far too diverse to be transformable into a generic structure in an indisputable way. Ambiguities exist especially regarding the distinction between co-products and by-products. While the former constitute intermediate products in the production process, the latter are considered as waste, which is subsequently recycled. Biomass flows are often subject to ambiguities as well. Manure is a prominent example for the conflict between co-products and by-products and the questions whether emissions of biodegradable flows should be regarded as circular per se and where to set the boundary between industrial and natural processes in agricultural systems are open to discussion. An analysis of several case studies revealed considerable differences in assessment results of up to 300% depending on model structure assumptions. There is no "true" answer to any of these ambiguities; therefore, transparency, as provided by a multi-level framework seems to be the most effective way to ensure comparability between different studies.

However, in order to support the development and implementation of sustainable resource management policy, MFA results need to be integrated into further assessments or interpreted in a meaningful way. Apart from sophisticated assessments that require high amounts of data, computational capacity and scientific expertise, it is also important to provide decision makers with simple, ready-to-use evaluation indicators. The aim of such indicators is not to reflect the system's sustainability in a holistic way, but to enable first insights into where main problems and/or potentials are located and where further investigations are needed, as well as early trend detection and preliminary judgment of how well the system performs in comparison to other studies. Yet to date, a multitude of such indicators coexist alongside each other, which are often ill-defined and neglect essential processes and feedbacks in the system. The present thesis contributes to the efforts of alleviating these problems by testing two straightforward indicators on their applicability and informative value in the Austrian P-N system. The indicators selected are C, which quantifies the share of recycling on total system throughput and SCE, which measures the extent to which a substance is concentrated or diluted within a system.

As SCE is an indicator without physically defined boundaries, ensuring comparability between results for different resources or regions, which is crucial if the indicator should be applied for the analysis of coupled systems, constitutes again a major challenge. In the present thesis it was solved by introducing a case specific best-case and worst-case scenario, against which the current situation and different measures were assessed.

In response to the third aim of this thesis the analysis revealed three main advantages of SCE over C: First of all, by setting environmental background concentration with respect to imission limits first insights into the environmental impacts of emission flows can be gained. Especially when dealing with reactive and highly volatile substances such as N, the possibility of differentiating between different species within a flow is another great asset. Furthermore, contrary to C, SCE can capture effects of dematerialization and system downsizing, as well as changes in process efficiency. Nevertheless, due to the fact that primary materials often exhibit high flow concentrations and thus contribute less to the total entropy, issues of resource criticality may be overlooked under certain circumstances.

Regarding the forth aim on P and N management in Austria, previous findings of a largely linear use at current state, but high potentials for a more sustainable nutrient management can be confirmed. The study reveals numerous synergies between P and N, mainly due to the fact that both nutrients are contained in flows of organic waste, for which recycling efforts could be increased and because measures reducing fertilizer losses to water bodies often address both substances simultaneously. Trade-offs such as losses of N during incineration of sewage sludge or meat and bone meal for P-recovery from ashes also exist, but play a negligible role in comparison. Similarly, negative side effects of some measures such as increased mineral fertilizer demand following a decrease in manure generation or increased emissions due to lower fertilizer efficiency of recycled products can be mitigated by combining them with other measures, such as an increase in fertilizer efficiency. Therefore, a combination of all measures analyzed shows the highest potentials to decrease mineral fertilizer demand and reduce losses in the waste sector, as well as emissions to soil, water, and the atmosphere. The combination of measures also scored highest when evaluated with respect to C and SCE, which on the one hand confirms the reliability of the results and, on the other hand, indicates that both simple indicators are at least in the context of the present case study capable of reflecting interrelations in the system adequately.

This case study demonstrates that coupled MFA enables better understanding of the diverse interdependencies in resource systems and points out benefits and trade-offs that might have been overlooked in a single substance analysis. The present thesis thus provides the basis for a comprehensive assessment including also other resources related to the P and N system. For instance, relevant interrelations could exist to cadmium (Cd) and uranium (U), which are contained as hazardous by-elements in mineral P-fertilizer and rock phosphate (Kratz et al. 2016) or to heavy metals, antibiotics, and organic pollutants present in human and animal excrements that might enter the food production system with recycled fertilizer and manure (Bousek et al. 2018; Umlauf et al. 2011). In addition, the incorporation of energy flows would be informative in terms of thermodynamic limits of recycling (Korhonen et al. 2018). Furthermore, methods of dynamic MFA could be applied to deepen knowledge of temporal trends and management dynamics.

The high level of uncertainty encountered in the present model, even though Austrian P management was among others chosen as a case study for its high level of reporting and generally good quality of available data compared to other substances (Zoboli et al. 2015), stresses the need to continue the ongoing research on dealing with uncertainty in MFA.

Apart from refining some processes like the interaction between plants, soil and water in agriculture or improved documentation of the fate of certain flows of waste and by-products, future work should focus on integrating coupled MFA based on the multi-level reference framework into a holistic assessment that represents environmental impacts of different management options in more detail and also takes economic and social aspects of sustainability into account. Results of such an integrative assessment could be used for comparison with the straightforward indicators analyzed in this thesis. Consequently, further insights into the capabilities of these simple indicators to represent overall system sustainability could be gained, which may lead to their adaptation or refined recommendations on the context in which they should be applied.

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Appendix A

MFA system structures

A.1 Austrian P-N system 2015

A.1.1 Customized model

A.1.1.1 Status quo

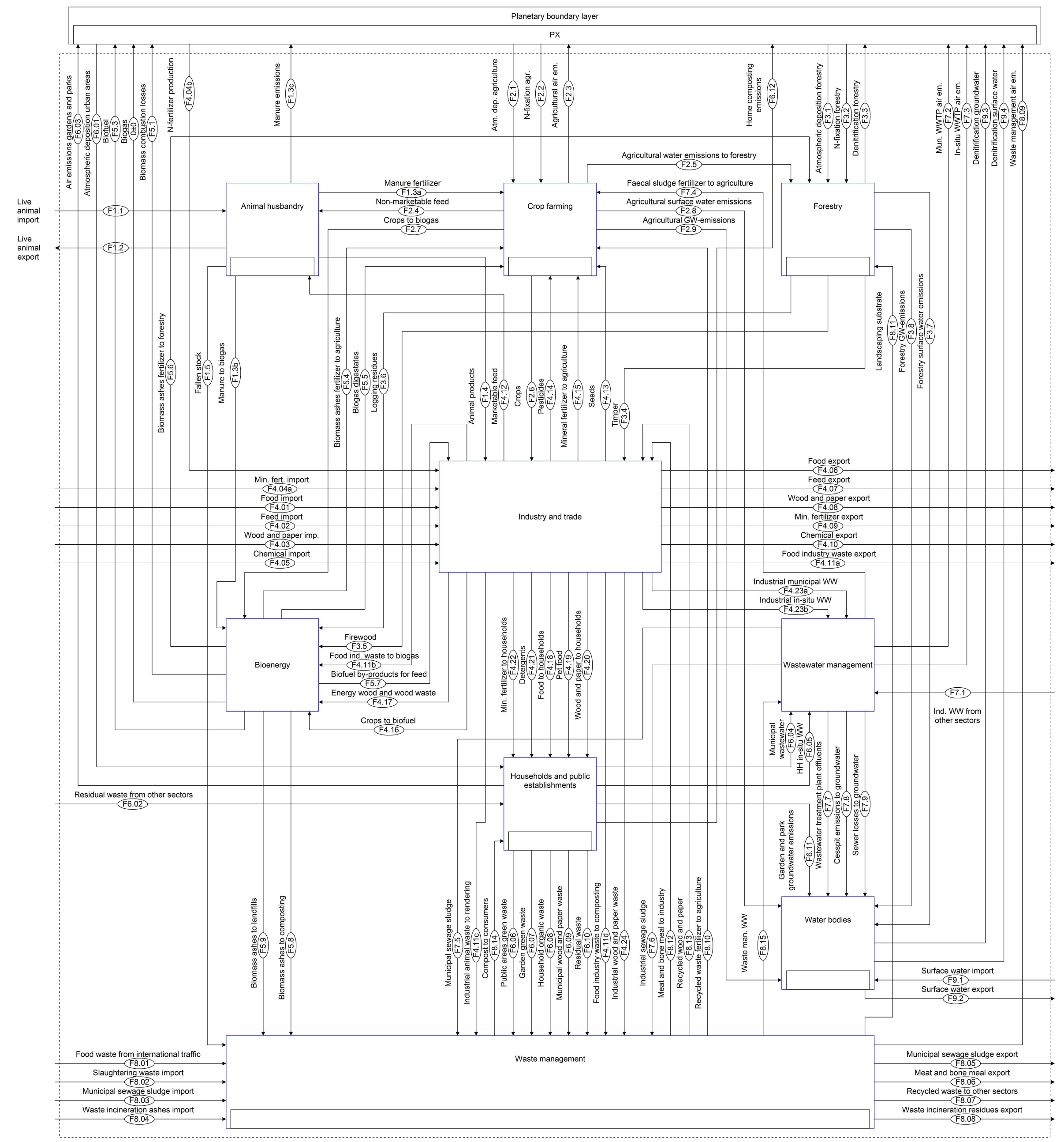

Fig. A.1: MFA structure of the Austrian P-N system 2015 - Top level.

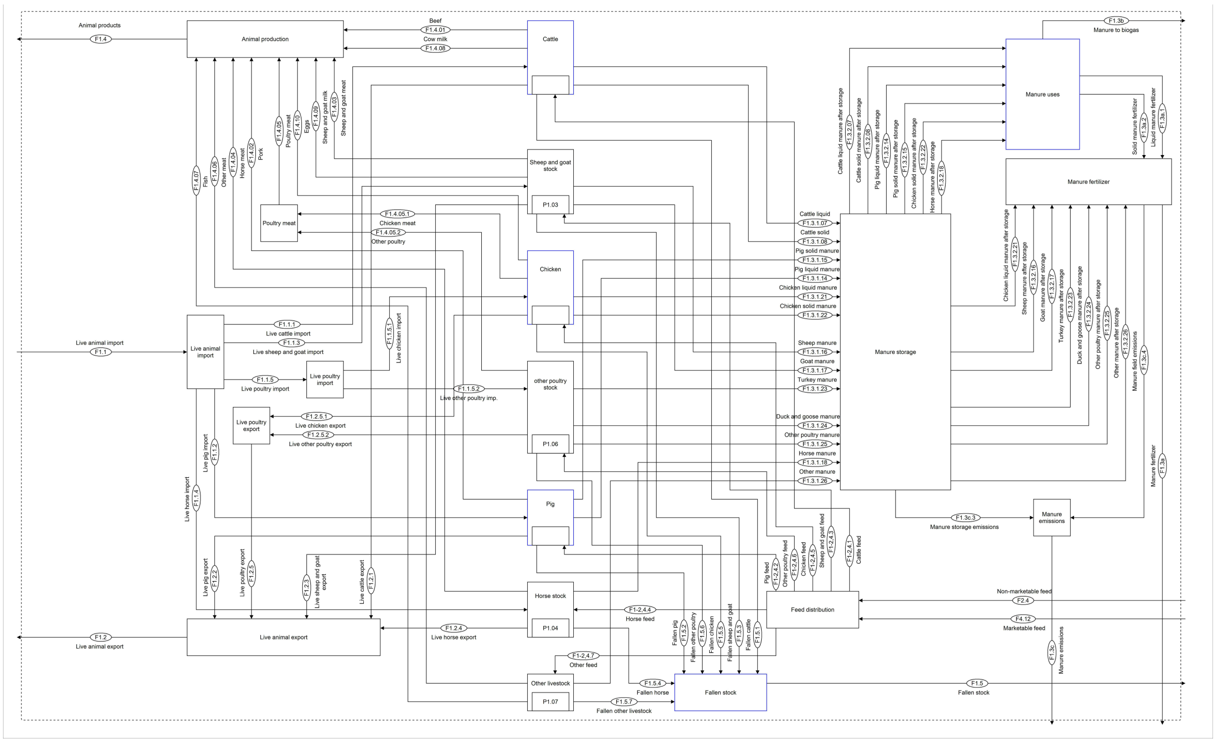

Fig. A.2: MFA structure of the Austrian P-N system 2015 - Subsystem *Animal husbandry*.

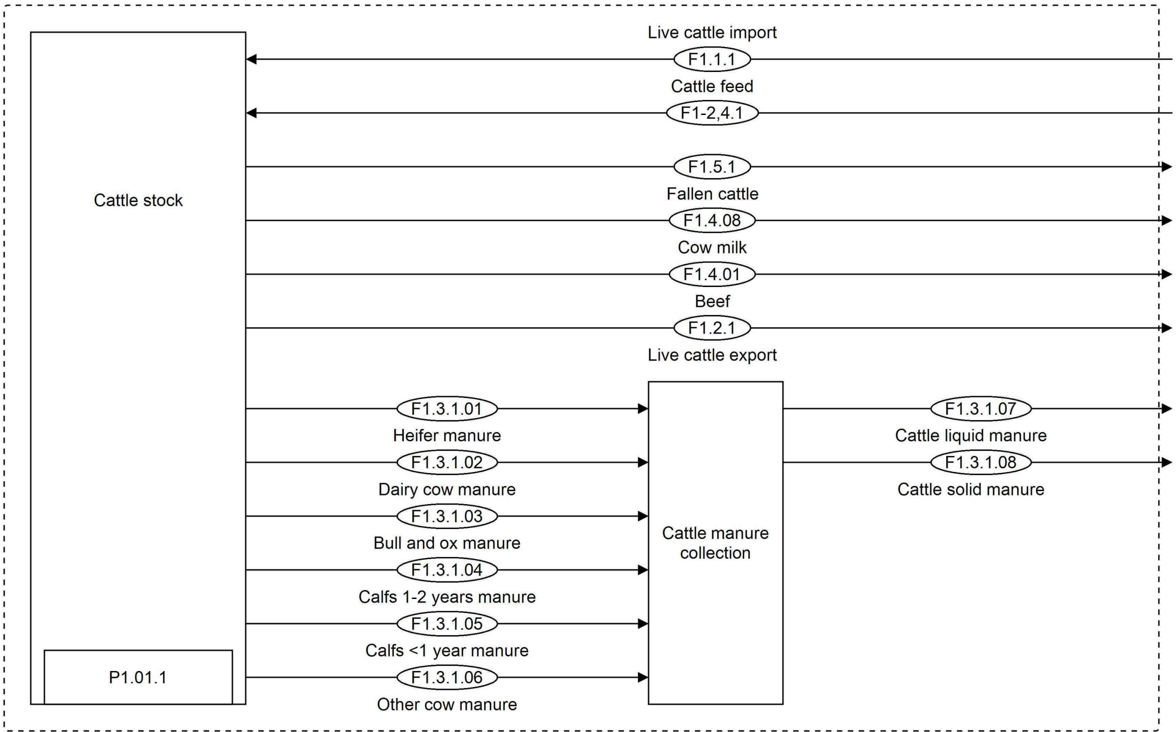

Fig. A.3: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Cattle*.

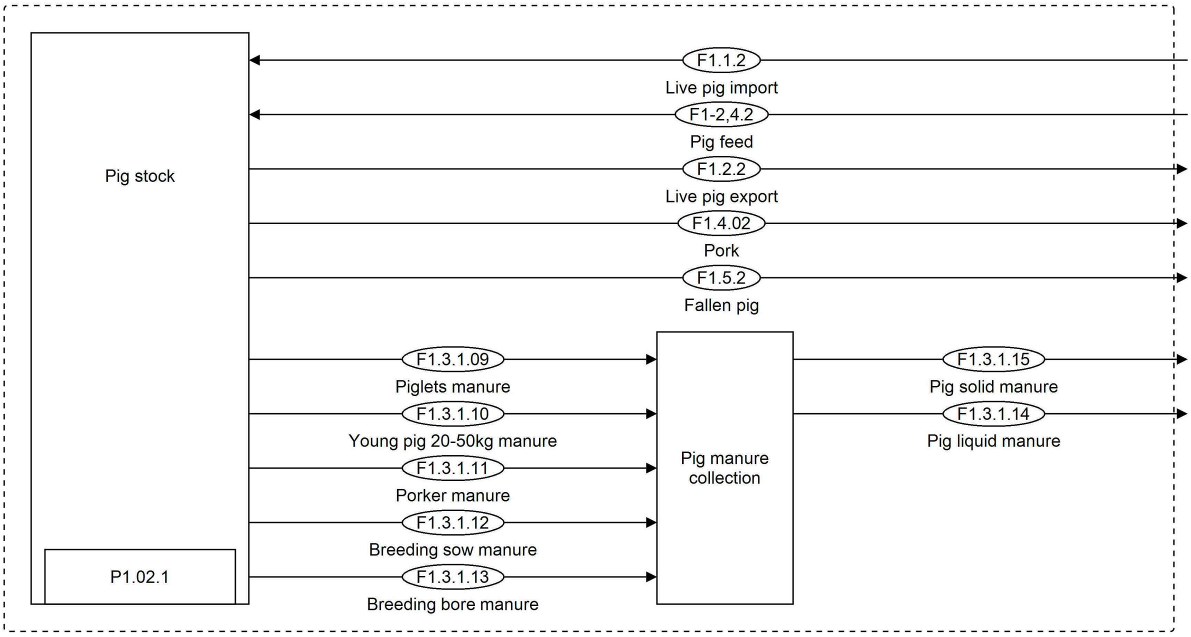

Fig. A.4: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Pig*.

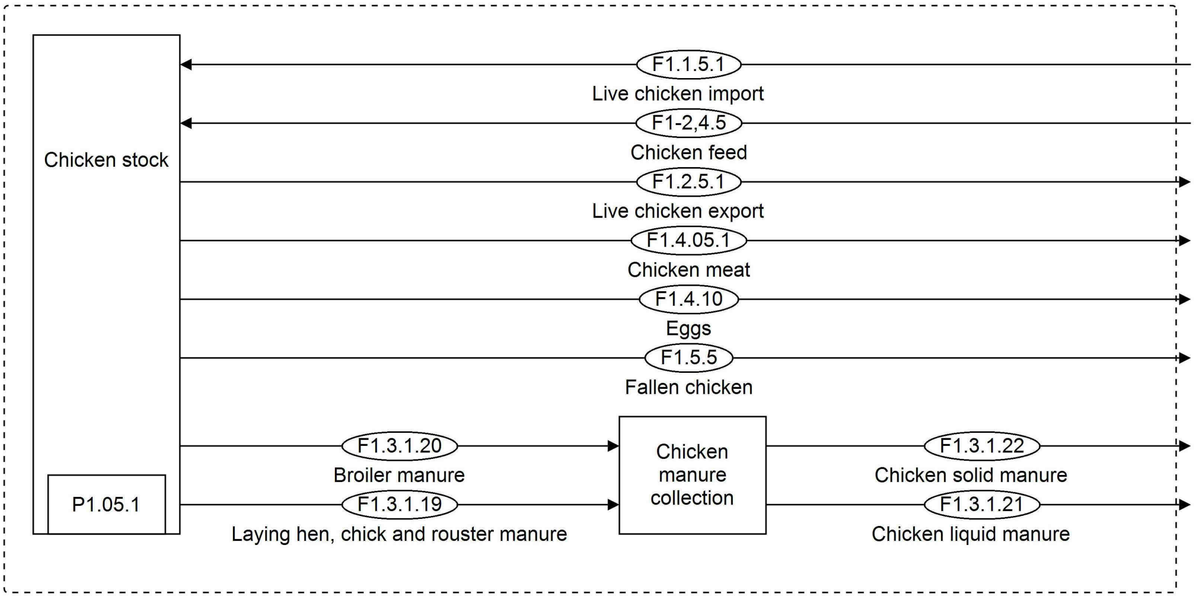

Fig. A.5: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Chicken*.

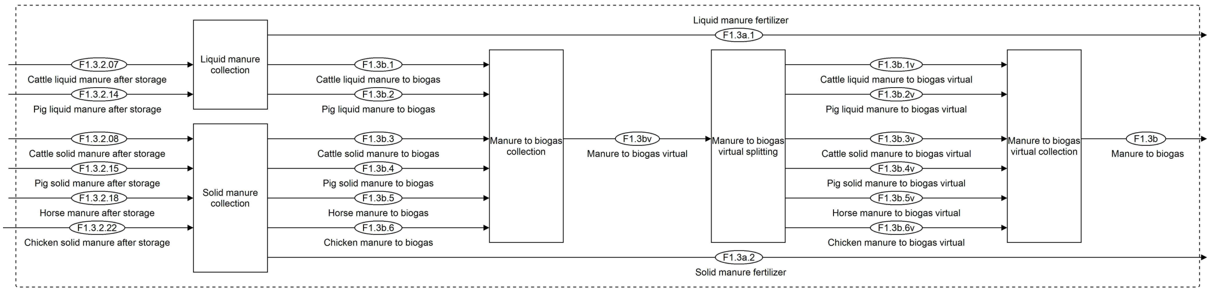

Fig. A.6: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Manure uses.*

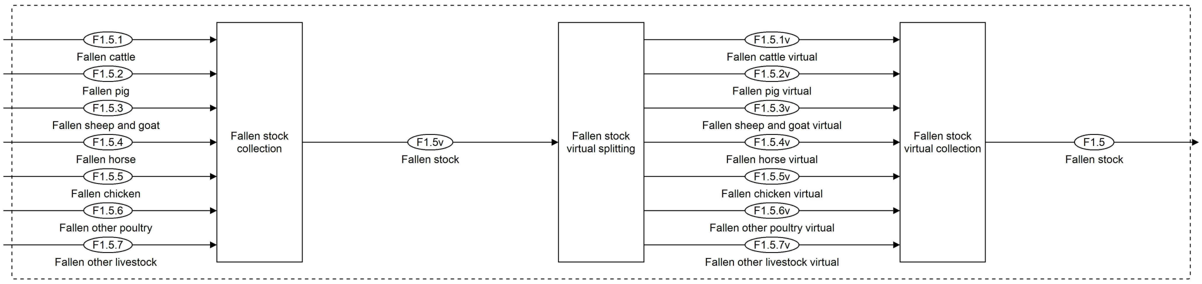

Fig. A.7: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Fallen stock.*

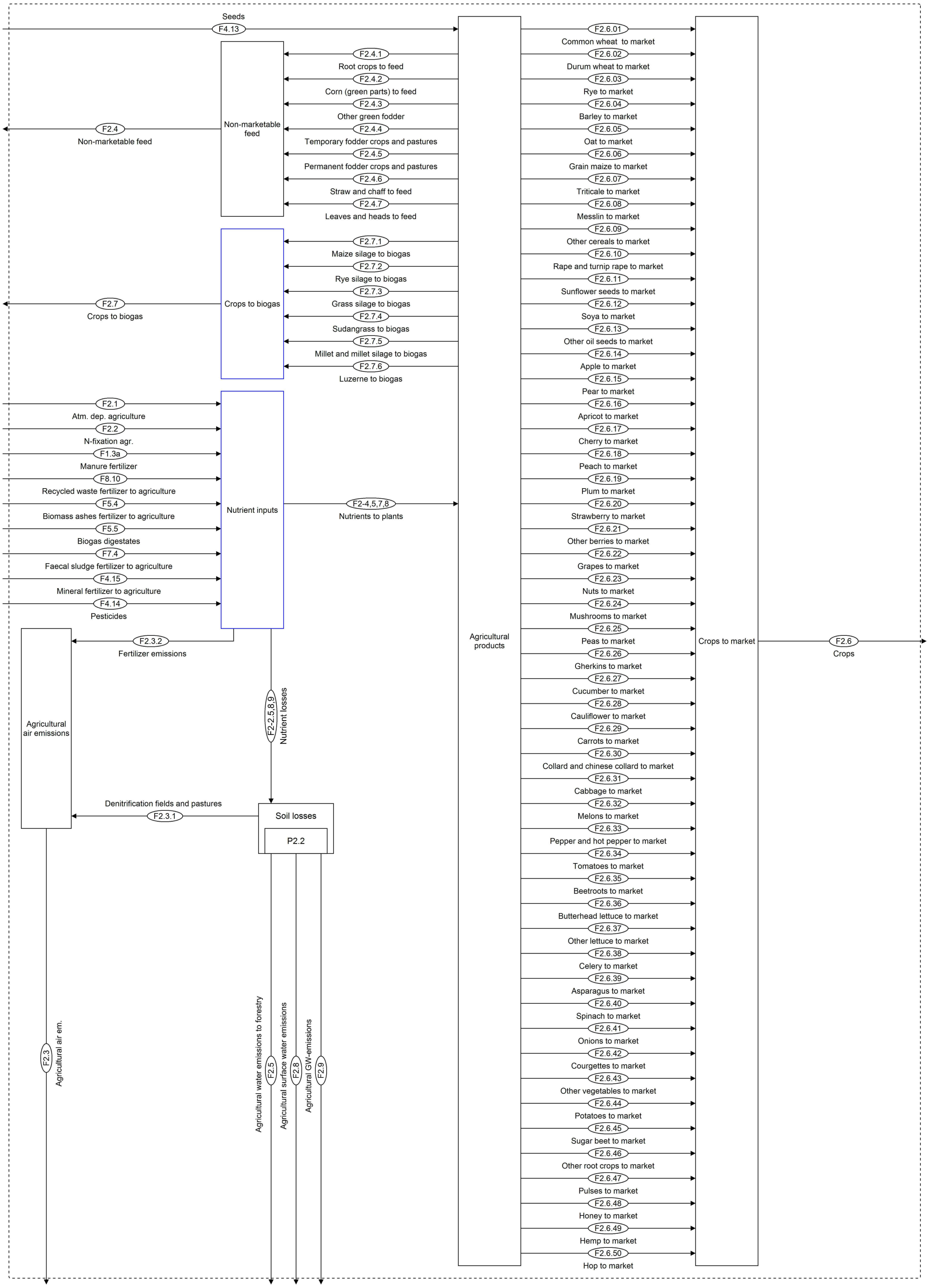

Fig. A.8: MFA structure of the Austrian P-N system 2015 - Subsystem *Crop farming*.

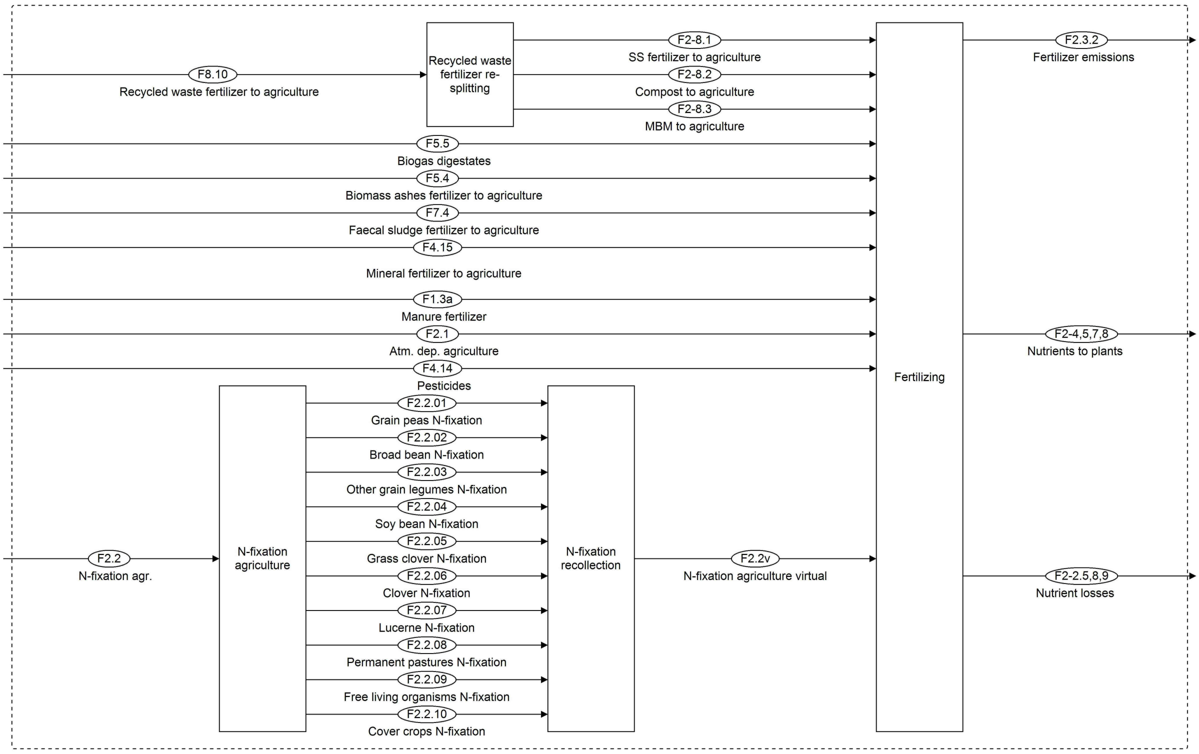

Fig. A.9: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Nutrient inputs*.

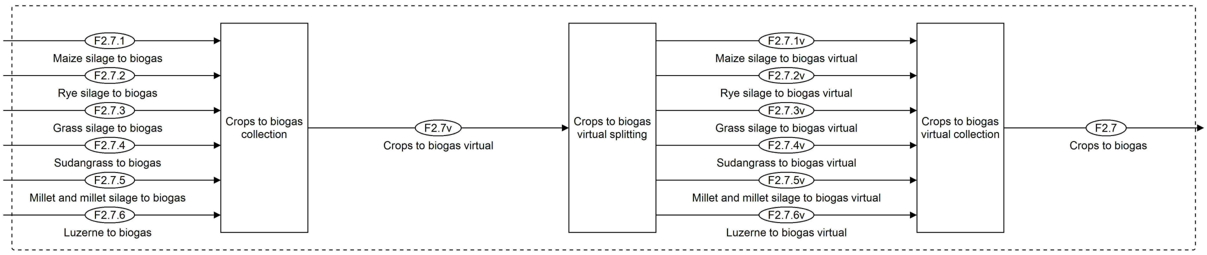

Fig. A.10: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Crops to biogas*.

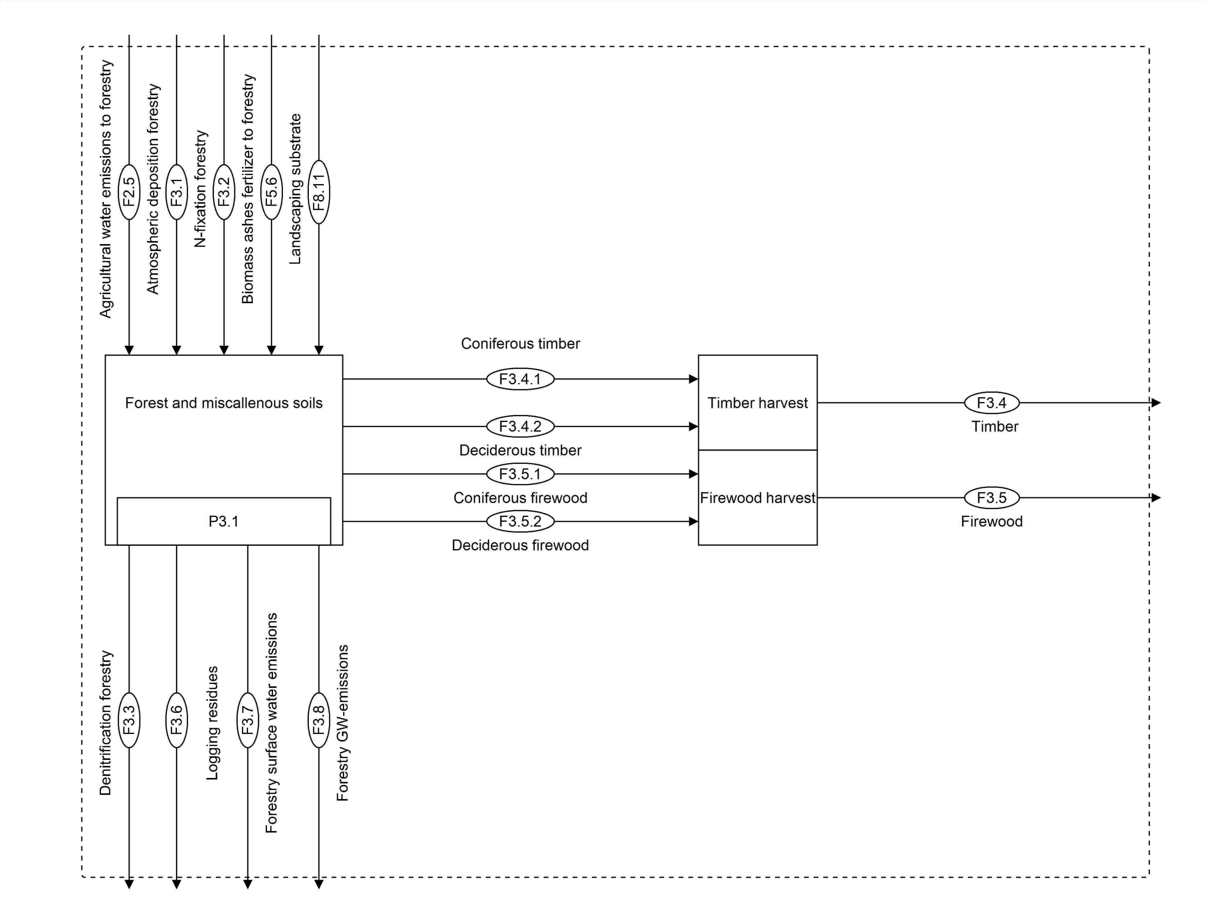

Fig. A.11: MFA structure of the Austrian P-N system 2015 - Subsystem *Forestry*.

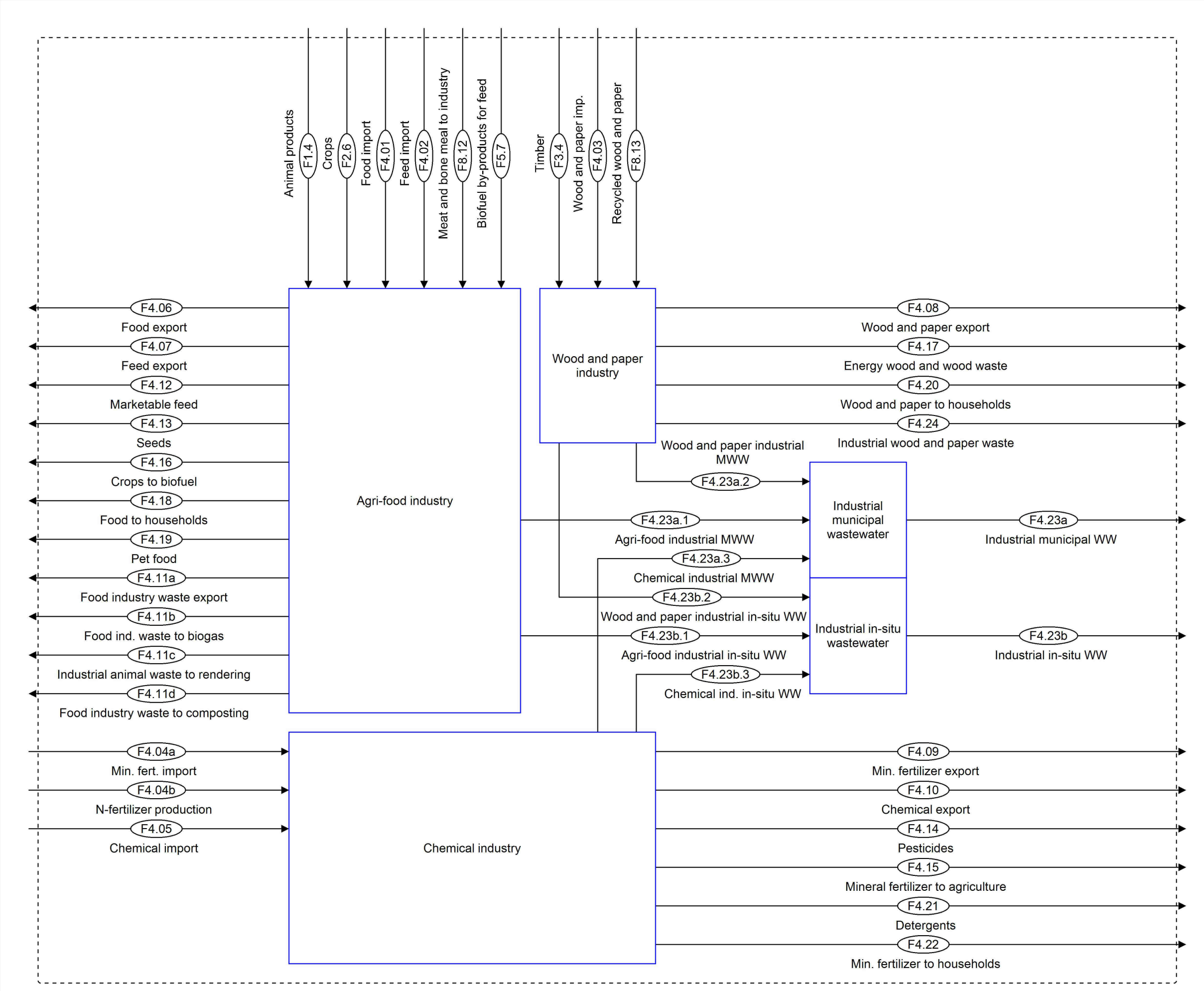

Fig. A.12: MFA structure of the Austrian P-N system 2015 - Subsystem *Industry and trade.*

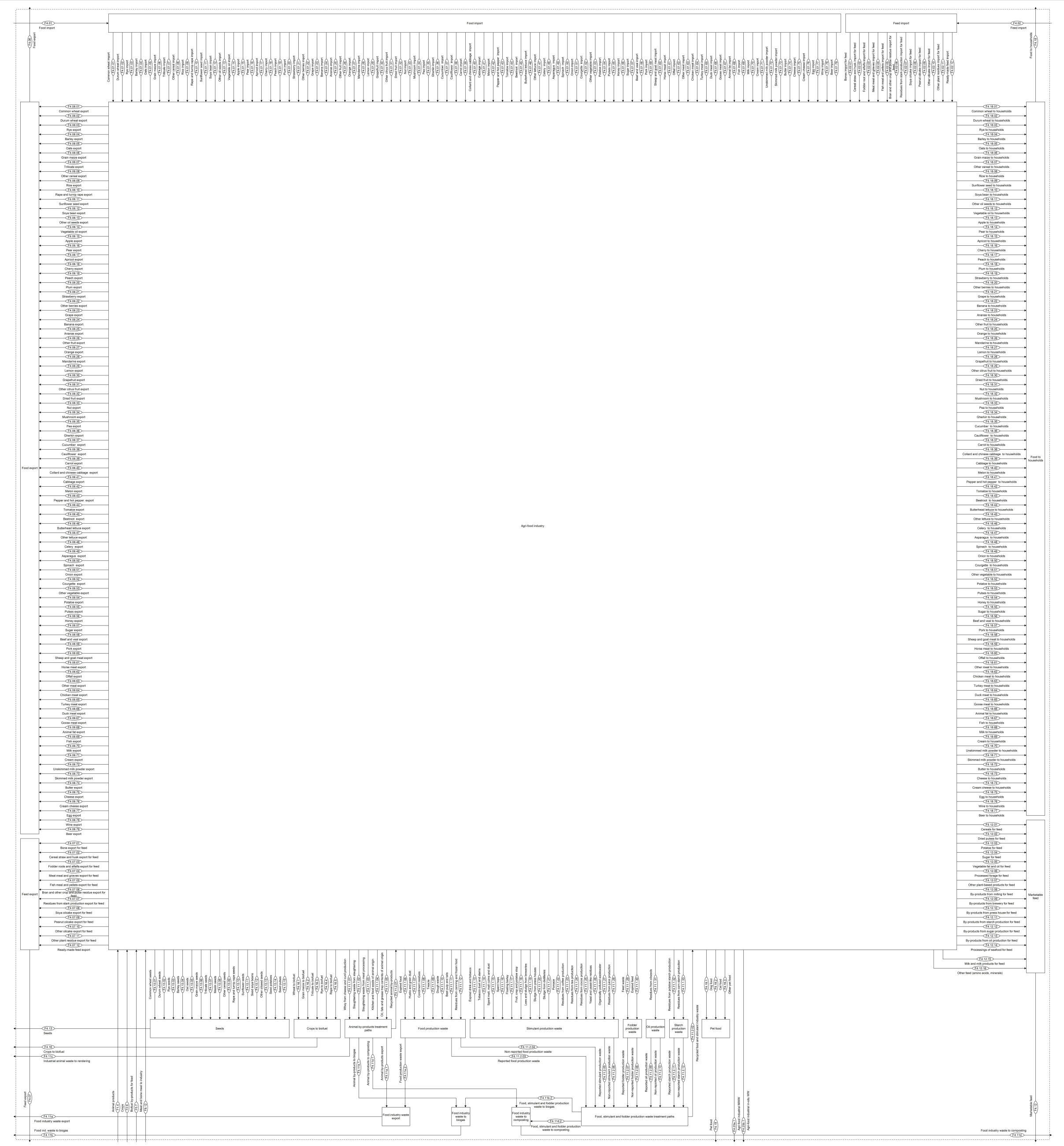

Fig. A.13: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Agri-food industry*.

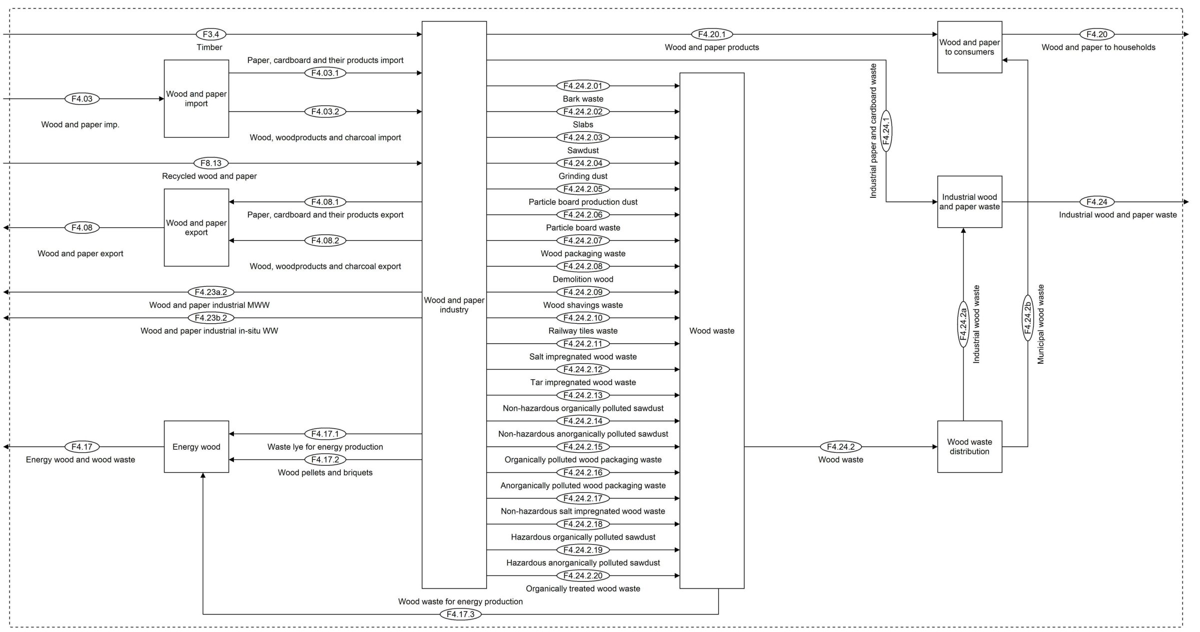

Fig. A.14: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Wood and paper industry.*

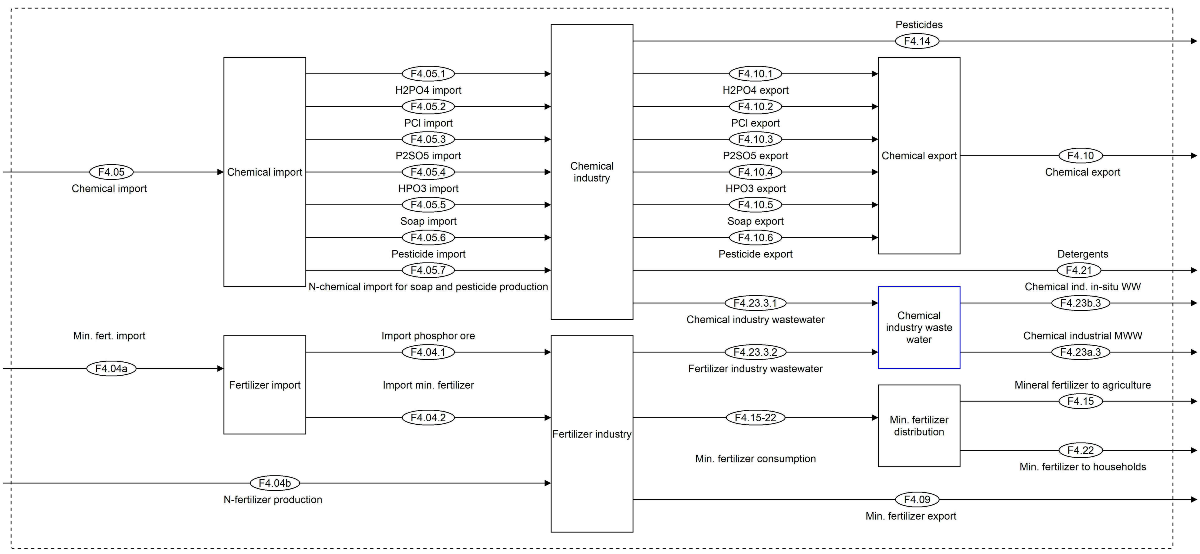

Fig. A.15: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Chemical industry.*

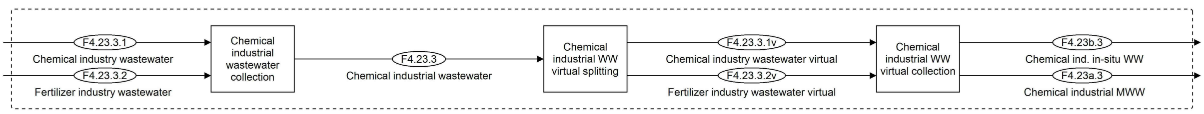

Fig. A.16: MFA structure of the Austrian P-N system 2015 - Subsubsubsystem *Chemical industry wastewater.*

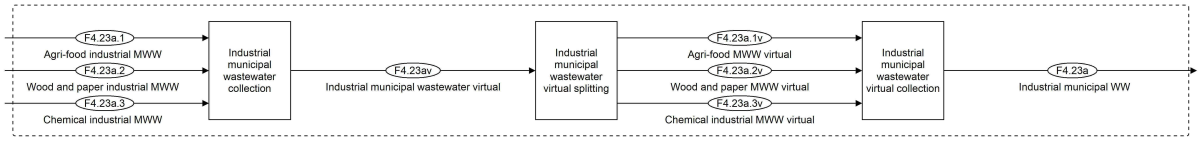

Fig. A.17: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Industrial municipal wastewater*.

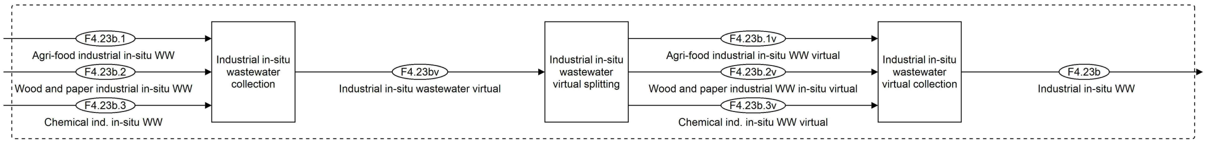

Fig. A.18: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Industrial in-situ wastewater*.

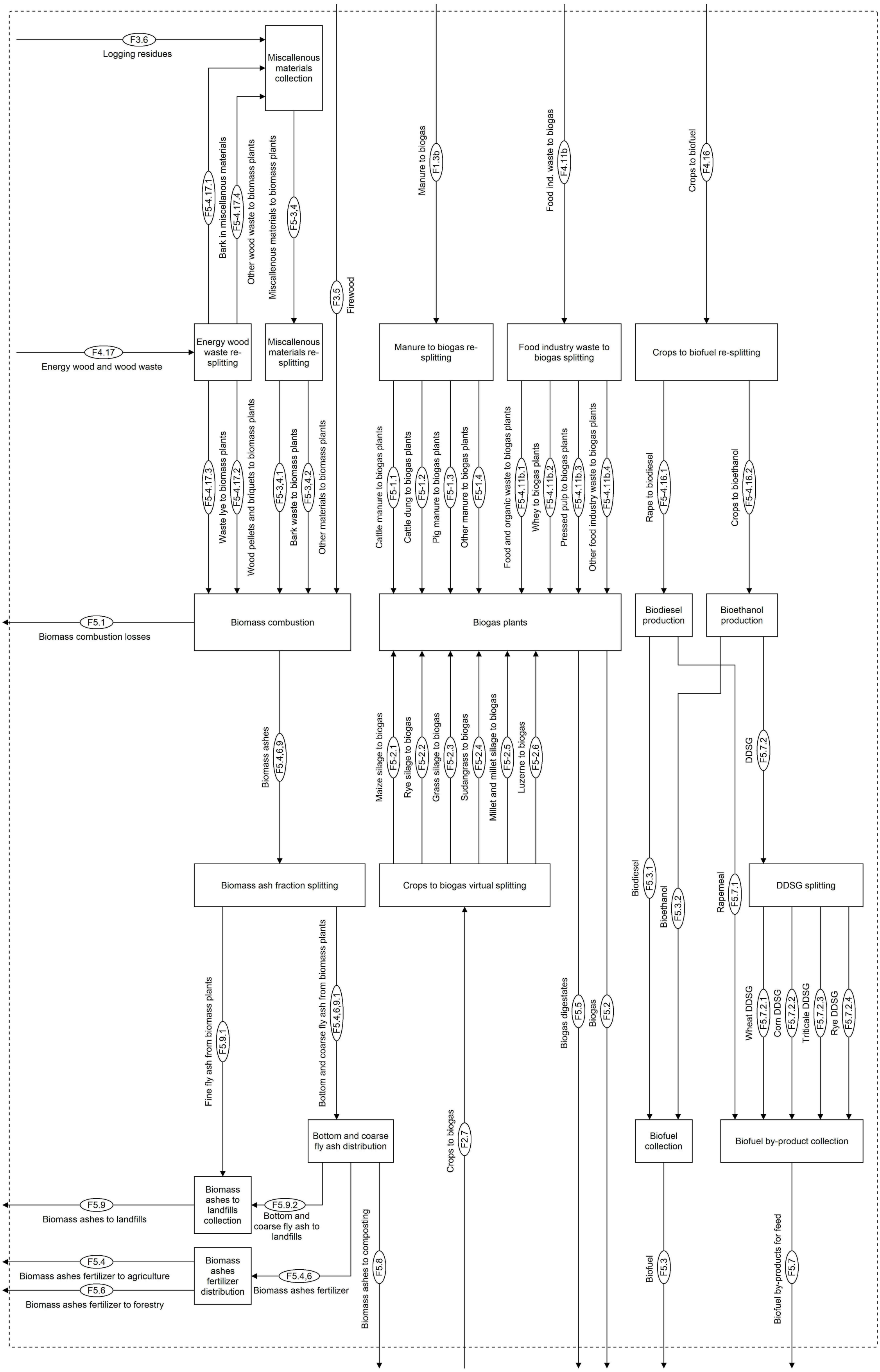

Fig. A.19: MFA structure of the Austrian P-N system 2015 - Subsystem *Bioenergy*.

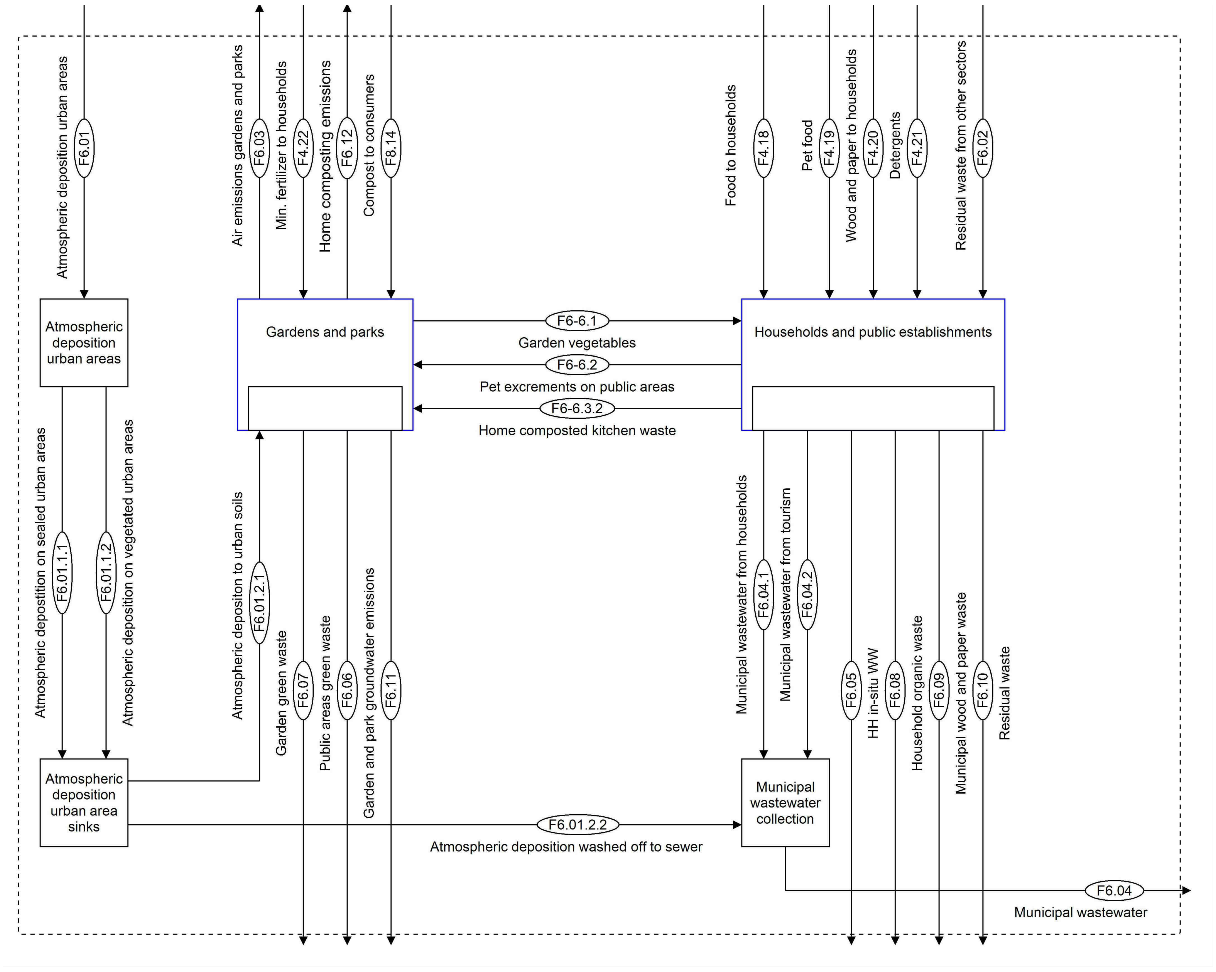

Fig. A.20: MFA structure of the Austrian P-N system 2015 - Subsystem *Households and public establishments.*

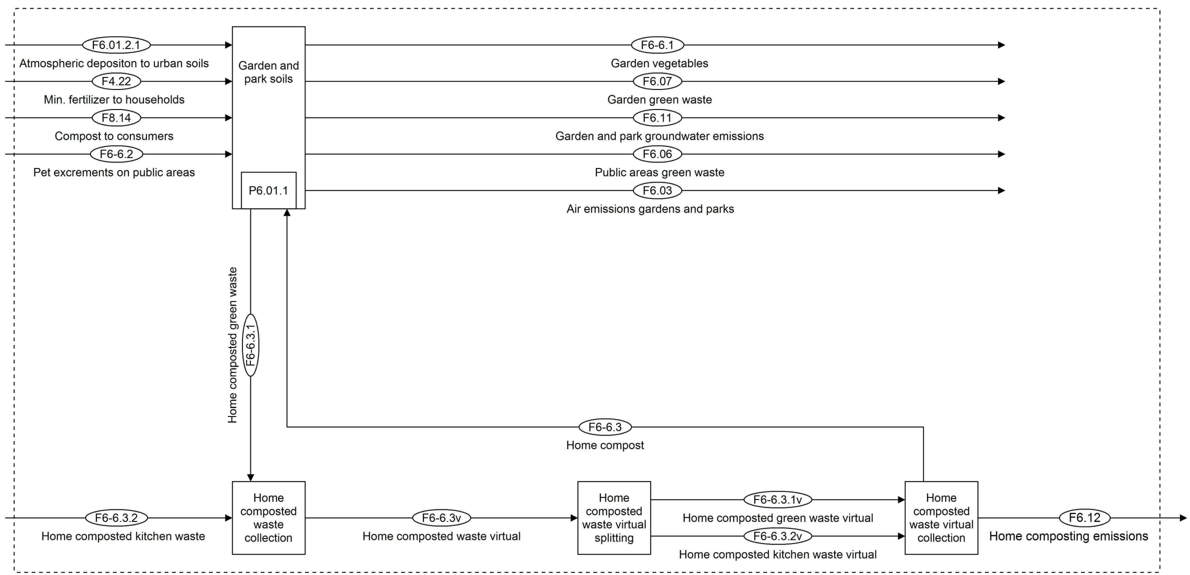

Fig. A.21: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Gardens and parks.*

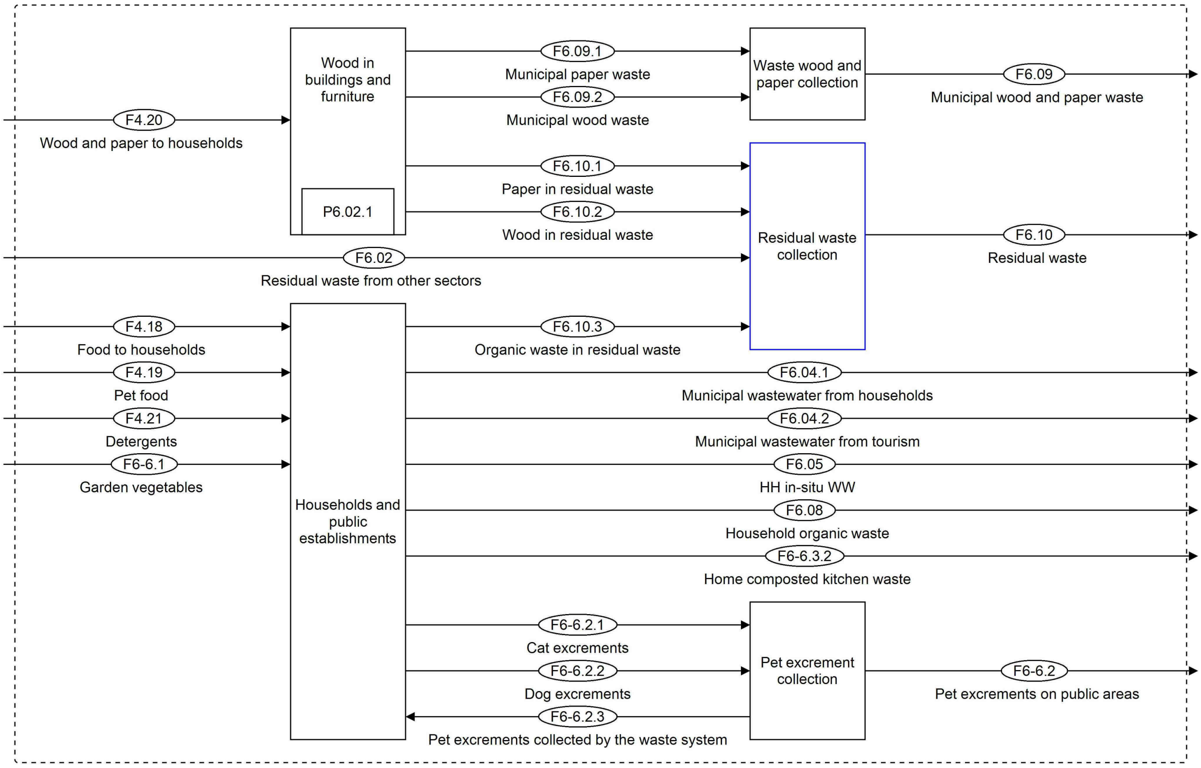

Fig. A.22: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Households and public establishments.*

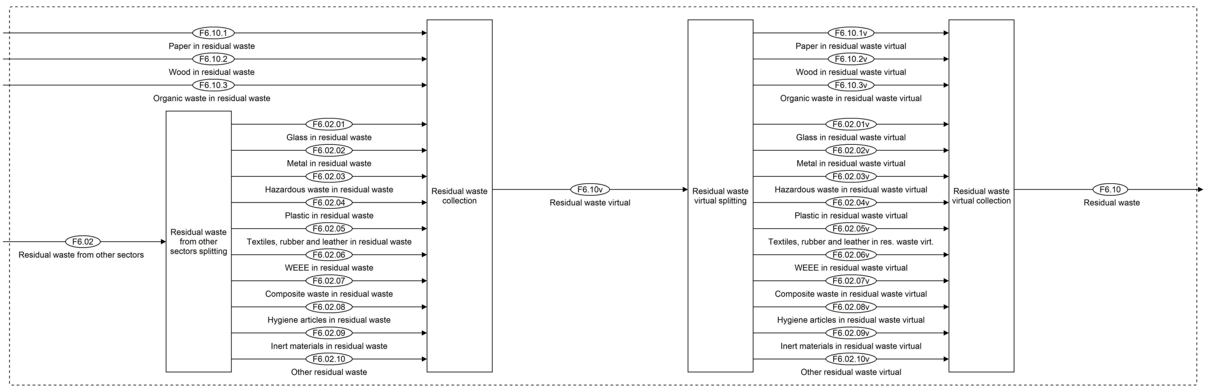

Fig. A.23: MFA structure of the Austrian P-N system 2015 - Subsubsubsystem *Residual waste collection.*

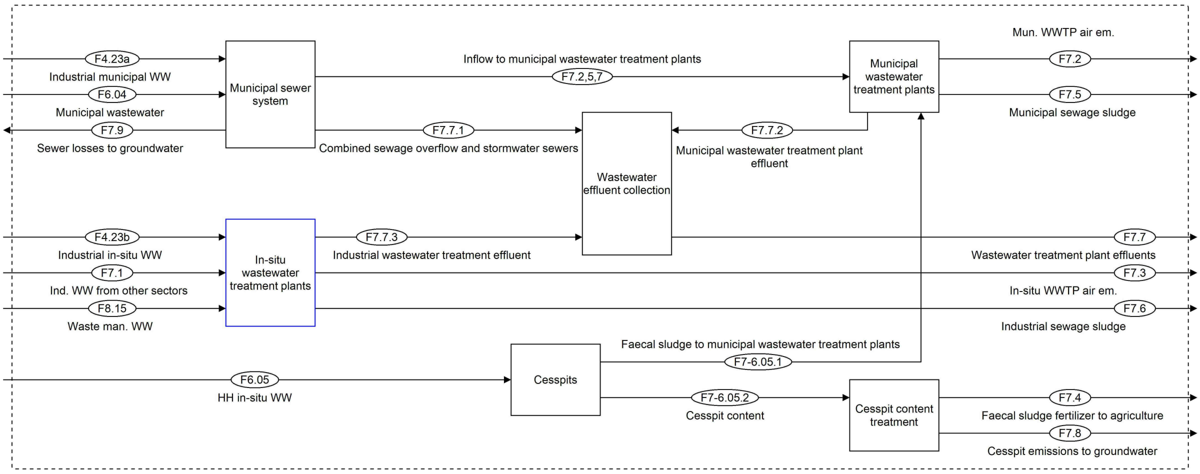

Fig. A.24: MFA structure of the Austrian P-N system 2015 - Subsystem *Wastewater treatment.*

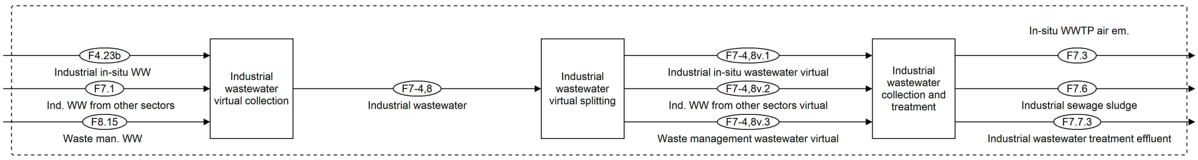

Fig. A.25: MFA structure of the Austrian P-N system 2015 - Subsubsystem *In-situ wastewater treatment plants.*

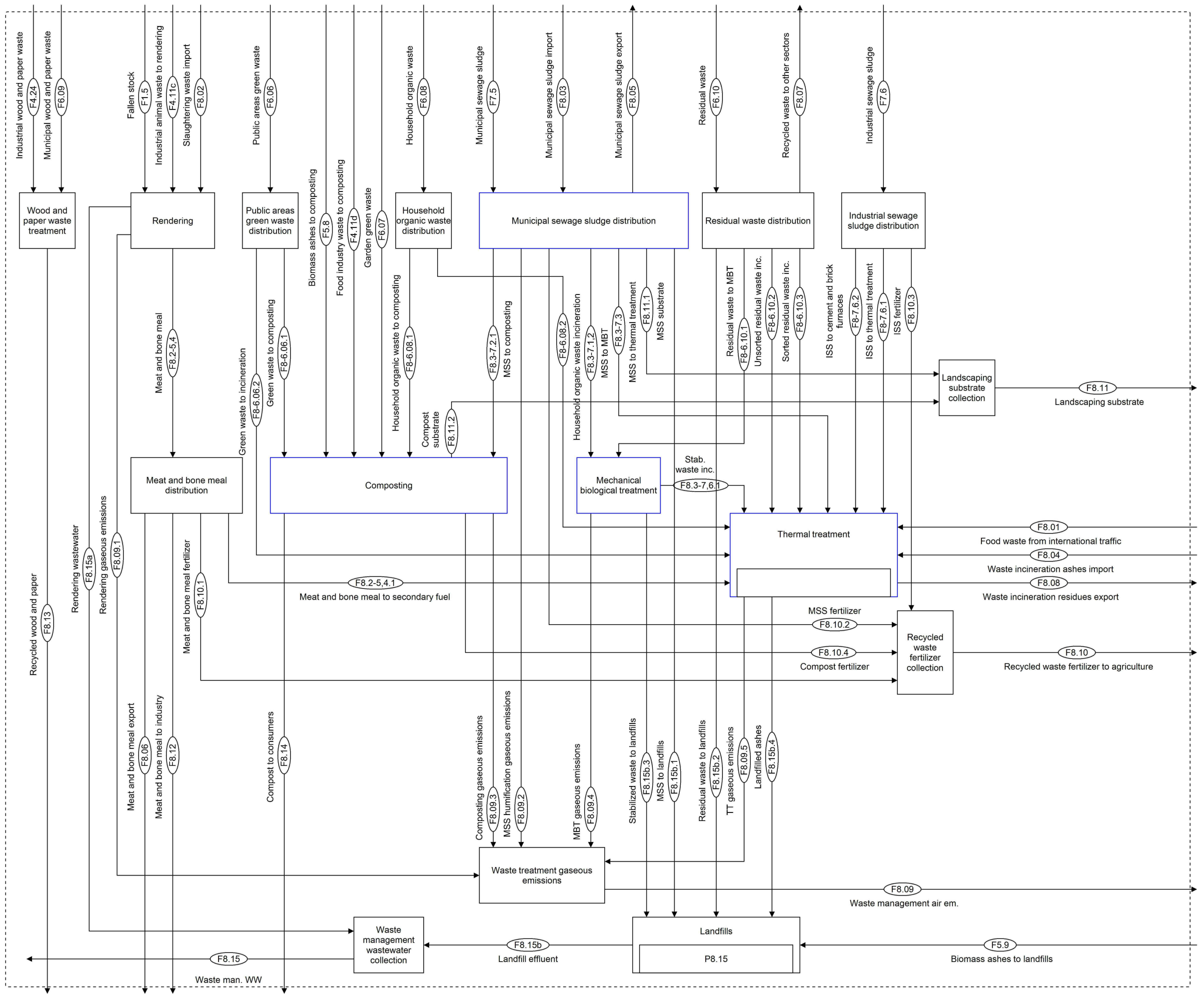

Fig. A.26: MFA structure of the Austrian P-N system 2015 - Subsystem *Waste management*.

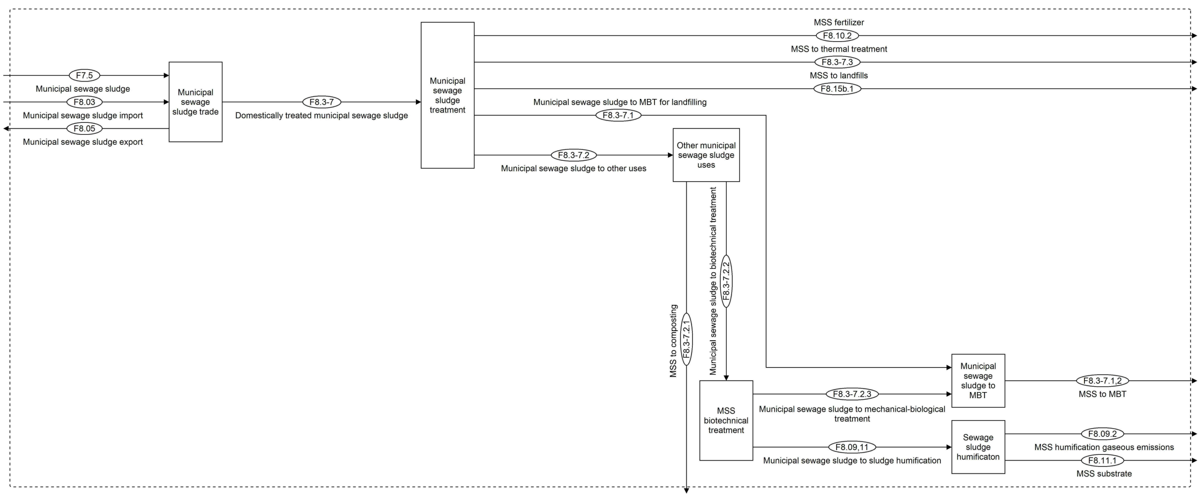

Fig. A.27: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Municipal sewage sludge distribution*.

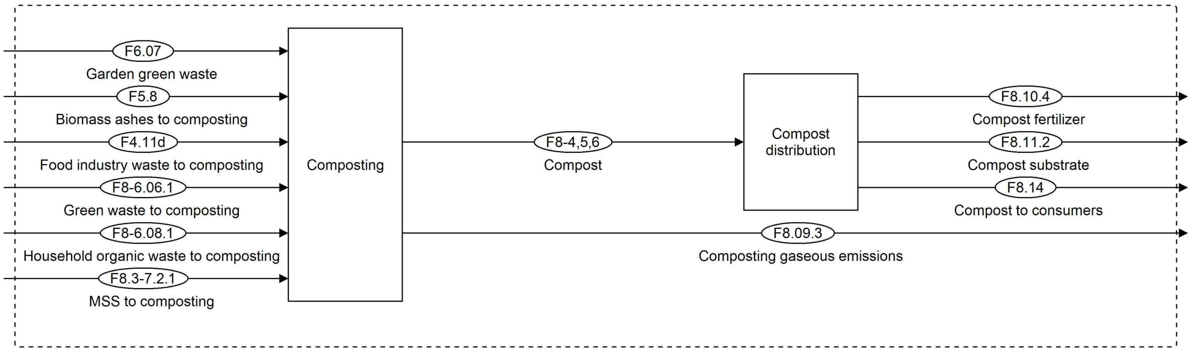

Fig. A.28: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Composting*.

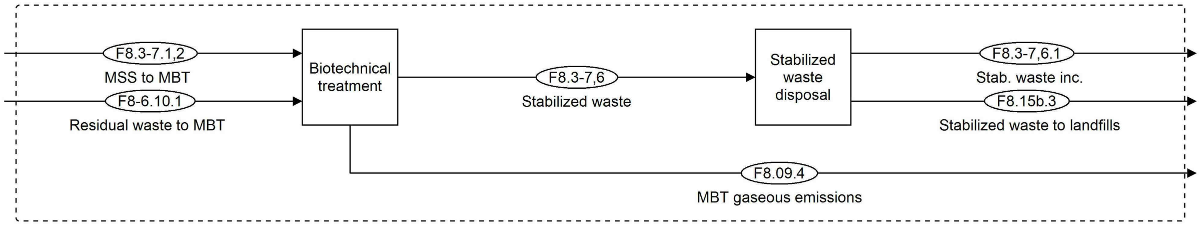

Fig. A.29: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Mechanical biological treatment*.

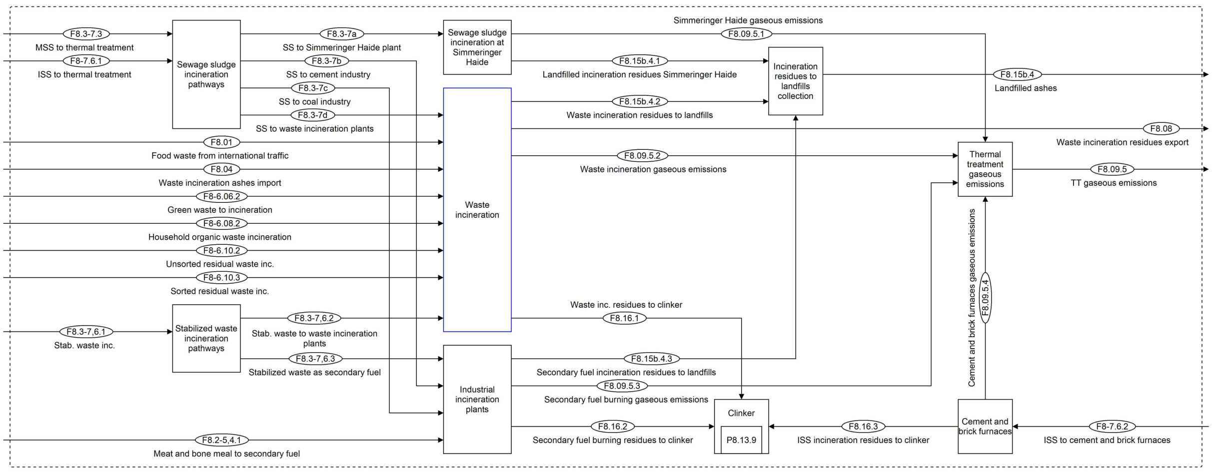

Fig. A.30: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Thermal treatment*.

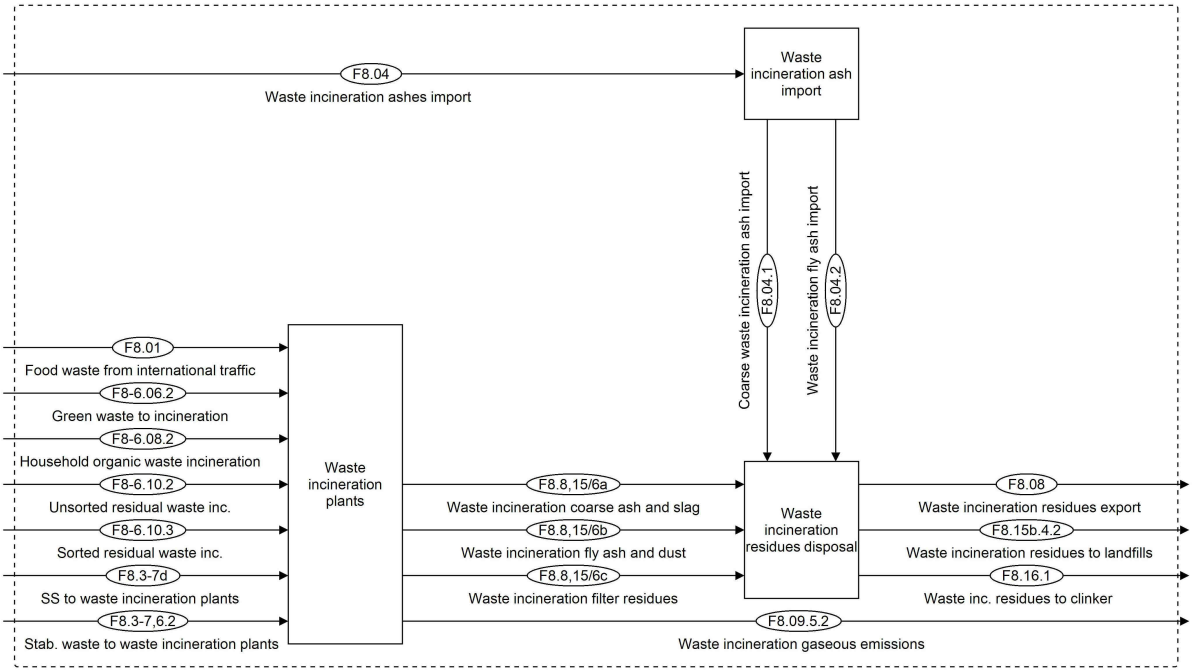

Fig. A.31: MFA structure of the Austrian P-N system 2015 - Subsubsubsystem *Waste incineration.*

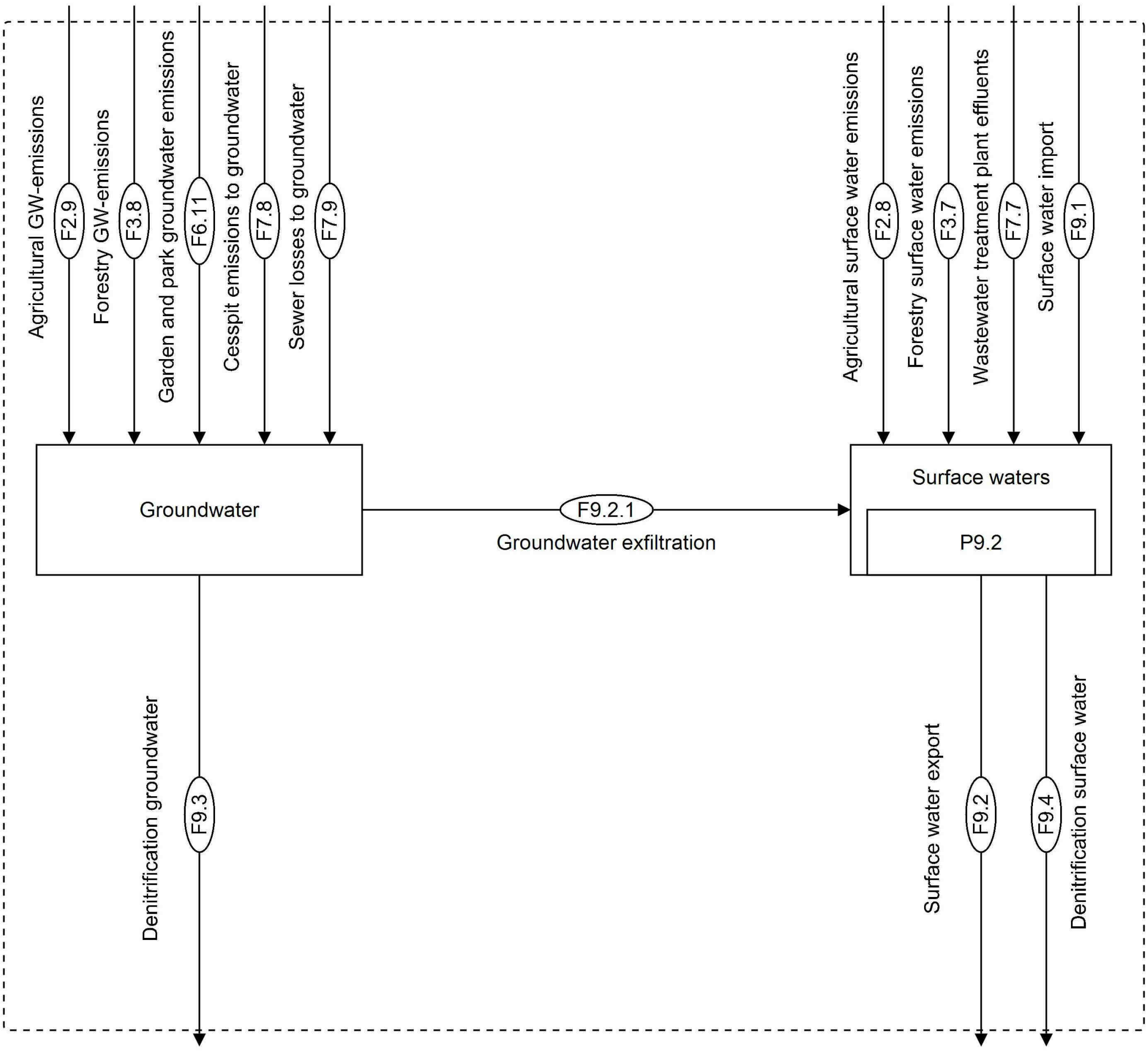

Fig. A.32: MFA structure of the Austrian P-N system 2015 - Subsystem *Water bodies.*

A.1.1.2 Adaptations in model structure under measure scenarios

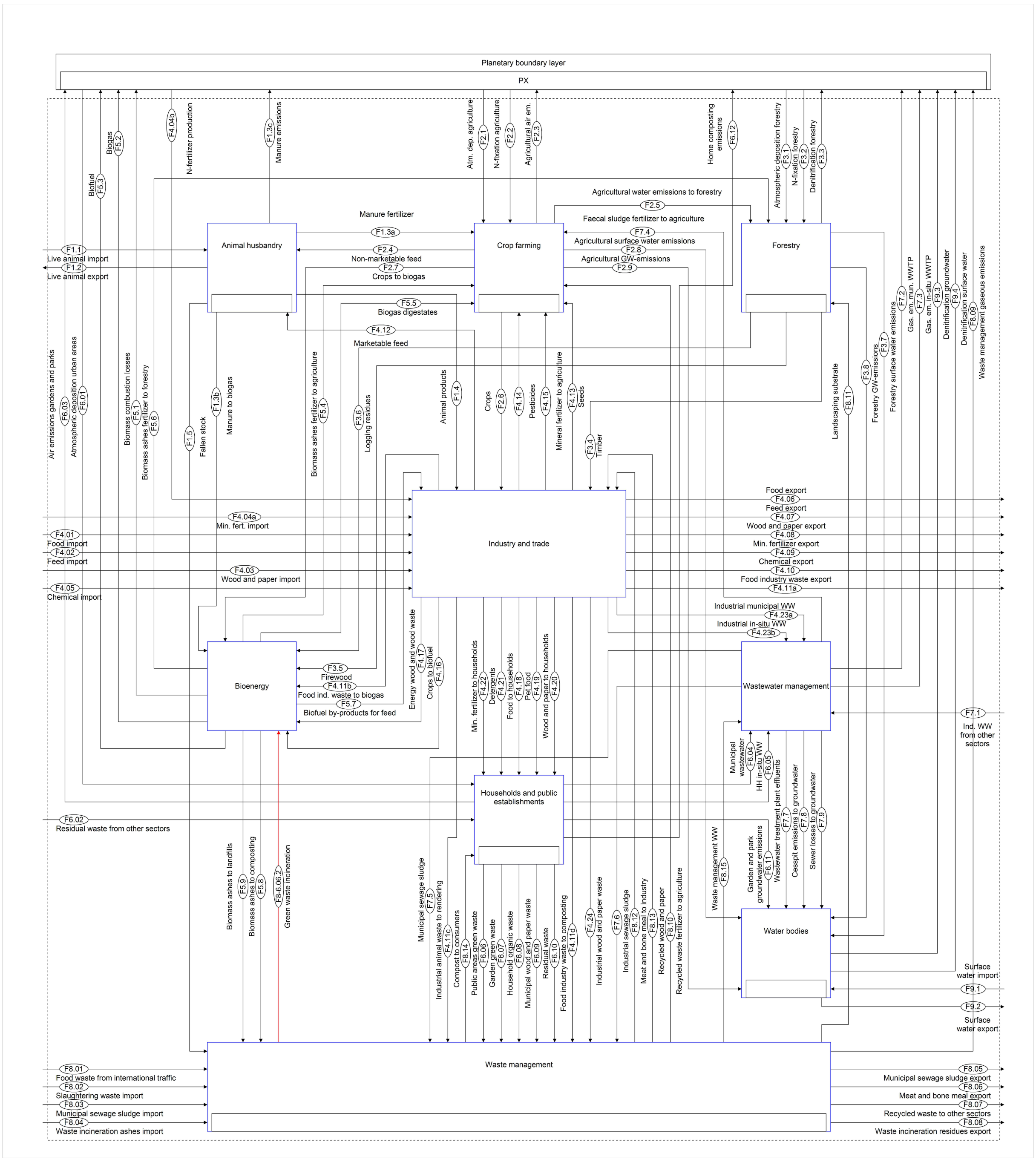

Fig. A.33: MFA structure of the Austrian P-N system 2015 - Top level. Changes to the original model structure under the measure *Increased recycling of green waste0*. Changes to the original model structure are marked in red.

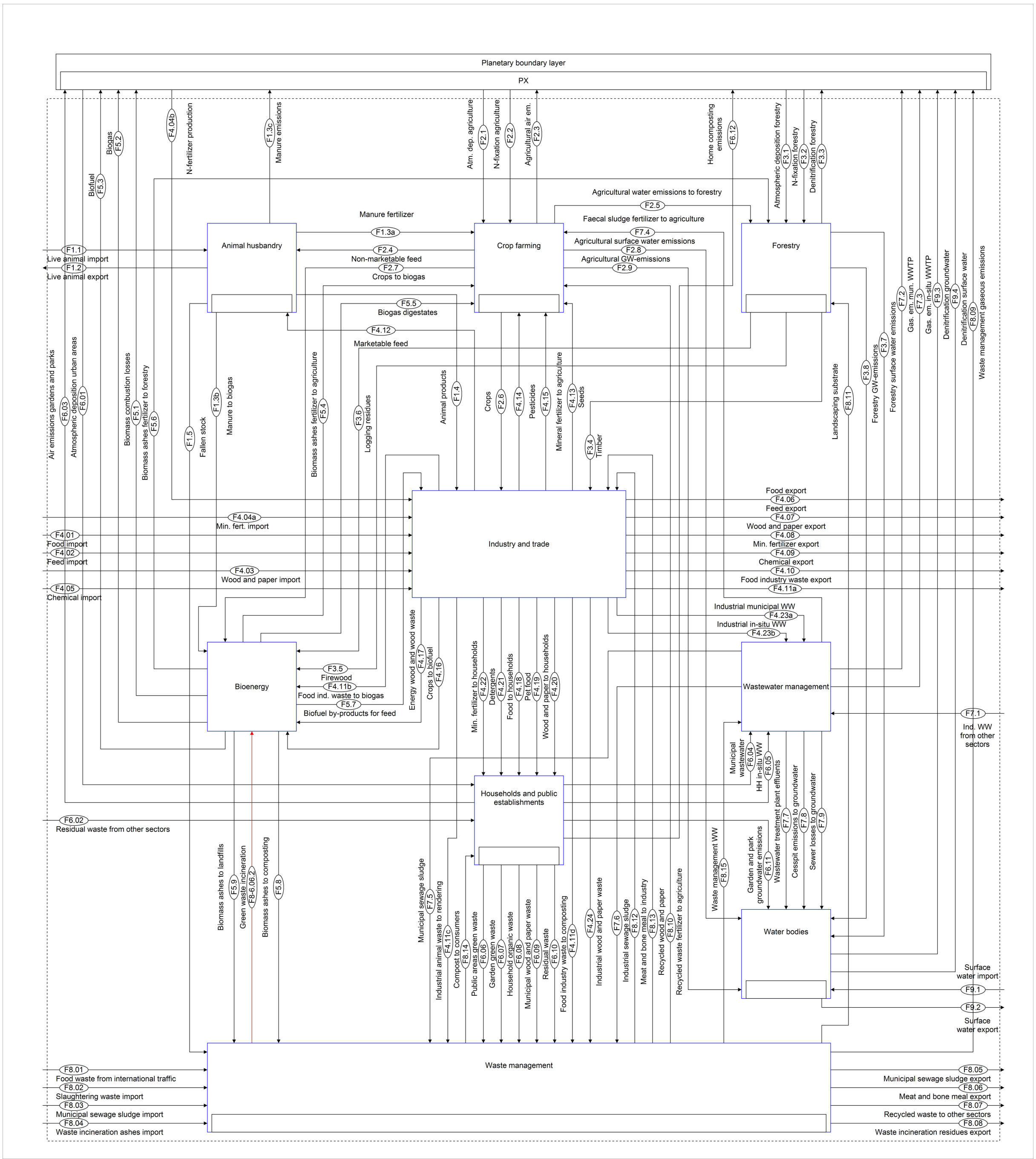

Fig. A.34: MFA structure of the Austrian P-N system 2015 - Top level. Changes to the original model structure under the Combined measure scenario. Changes to the original model structure are marked in red.

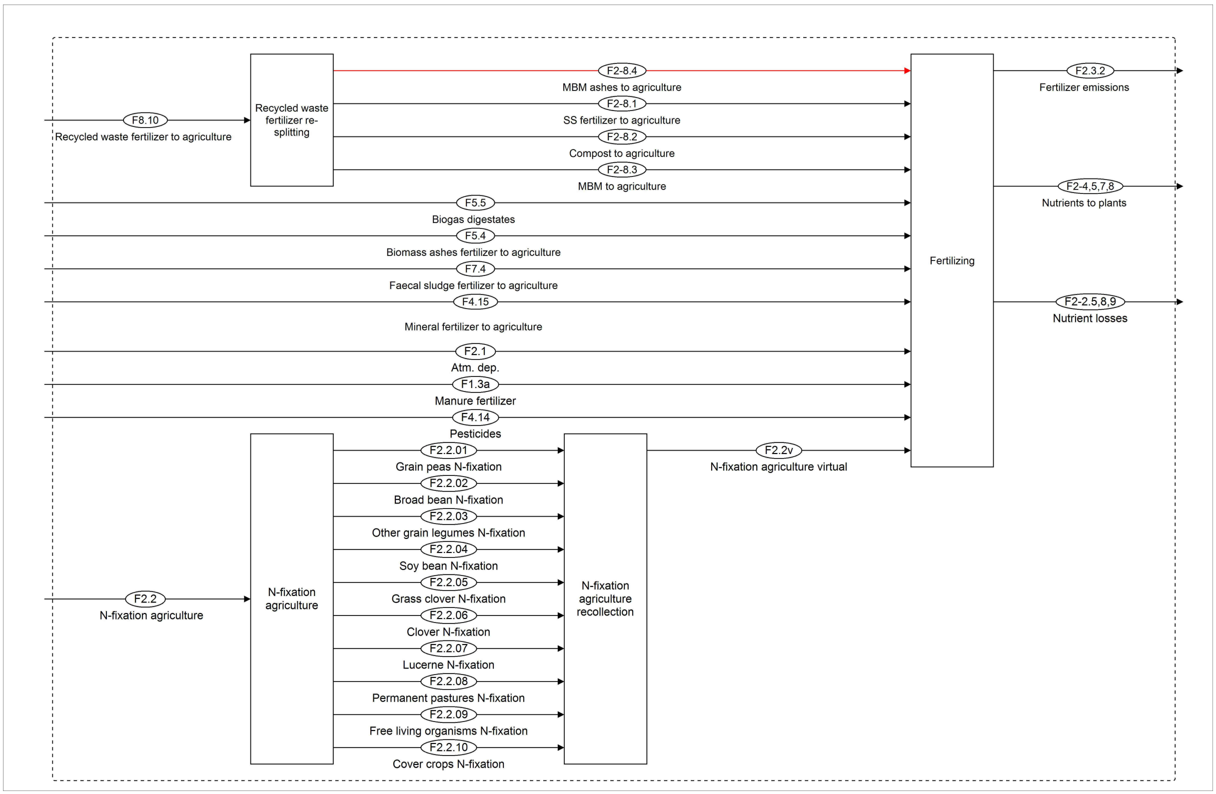

Fig. A.35: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Nutrient inputs*. Changes to the original model structure under the measure *Increased nutrient recovery from meat and bone meal*. Changes to the original model structure are marked in red.

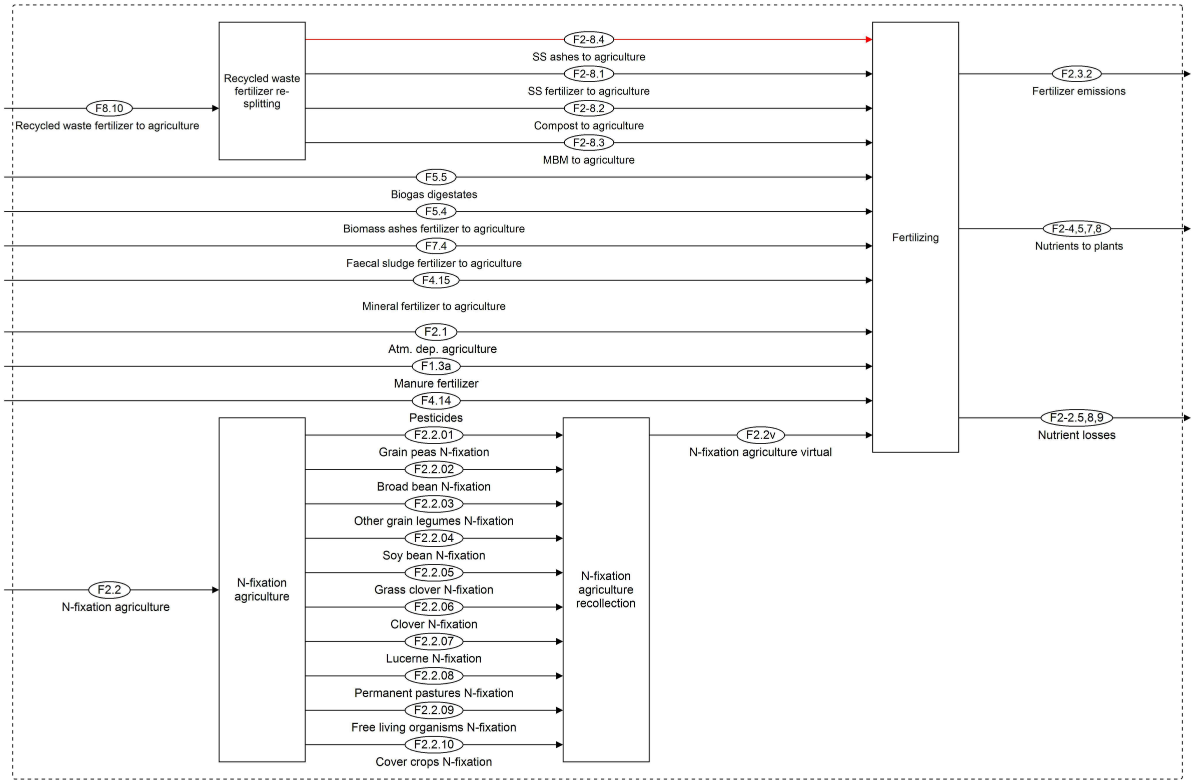

Fig. A.36: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Nutrient inputs*. Changes to the original model structure under the measure *Increased P-recovery from sewage sludge*. Changes to the original model structure are marked in red.

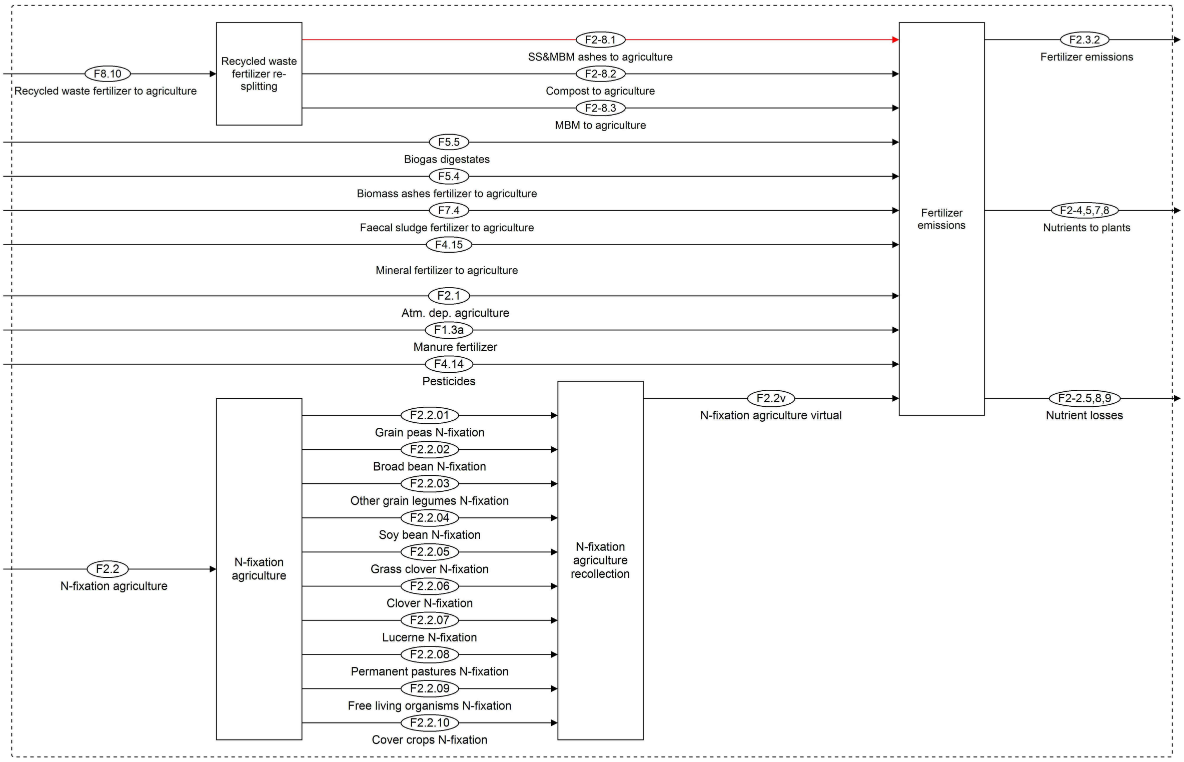

Fig. A.37: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Nutrient inputs*. Changes to the original model structure under the Combined measures scenario. Changes to the original model structure are marked in red.

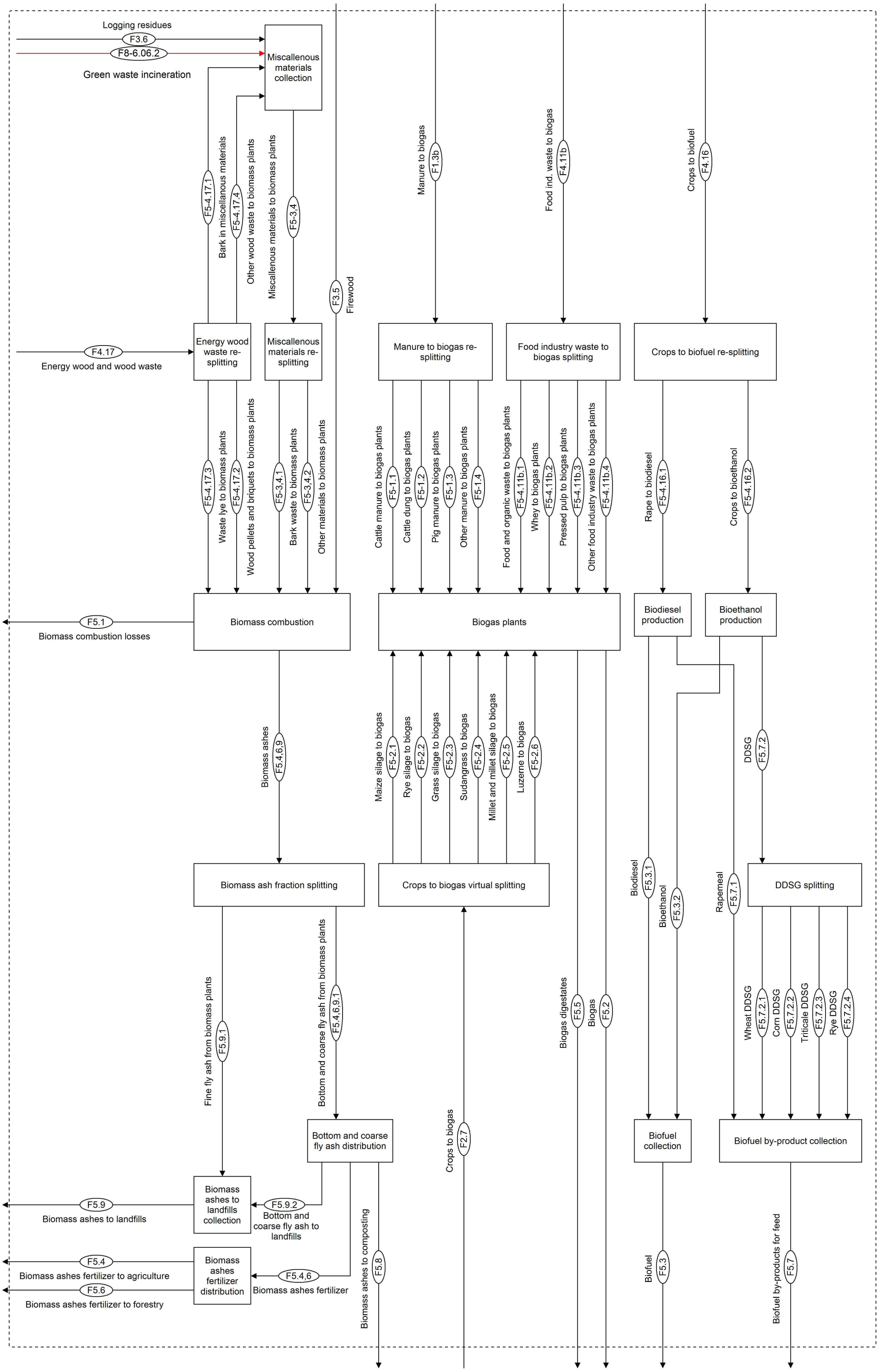

Fig. A.38: MFA structure of the Austrian P-N system 2015 - Subsystem *Bioenergy*. Changes to the original model structure under the measure *Increased recycling of green waste*. Changes to the original model structure are marked in red.

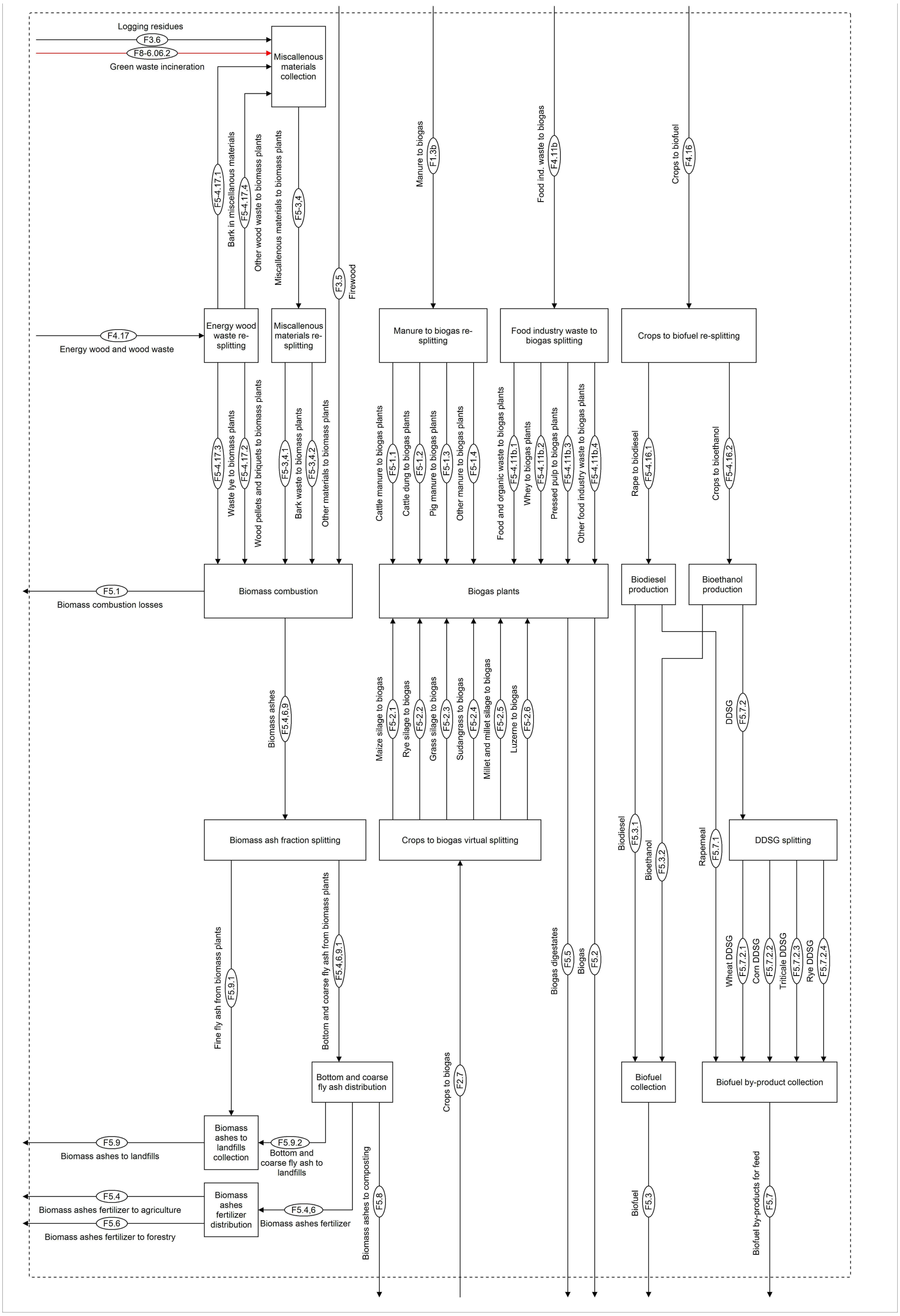

Fig. A.39: MFA structure of the Austrian P-N system 2015 - Subsystem *Bioenergy*. Changes to the original model structure under the Combined measures scenario. Changes to the original model structure are marked in red.

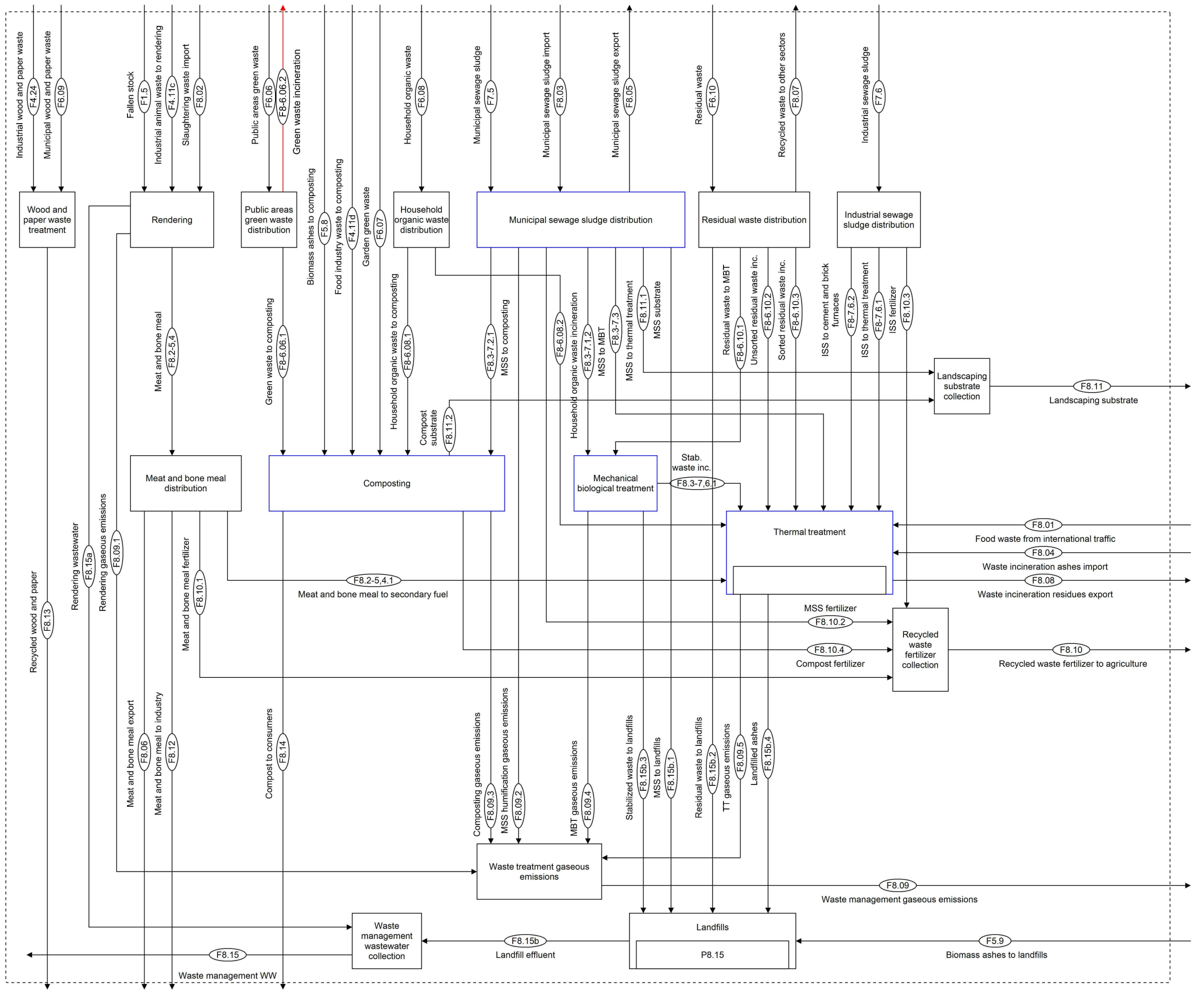

Fig. A.40: MFA structure of the Austrian P-N system 2015 - Subsystem *Waste management*. Changes to the original model structure under the measure *Increased recycling of green waste*. Changes to the original model structure are marked in red.

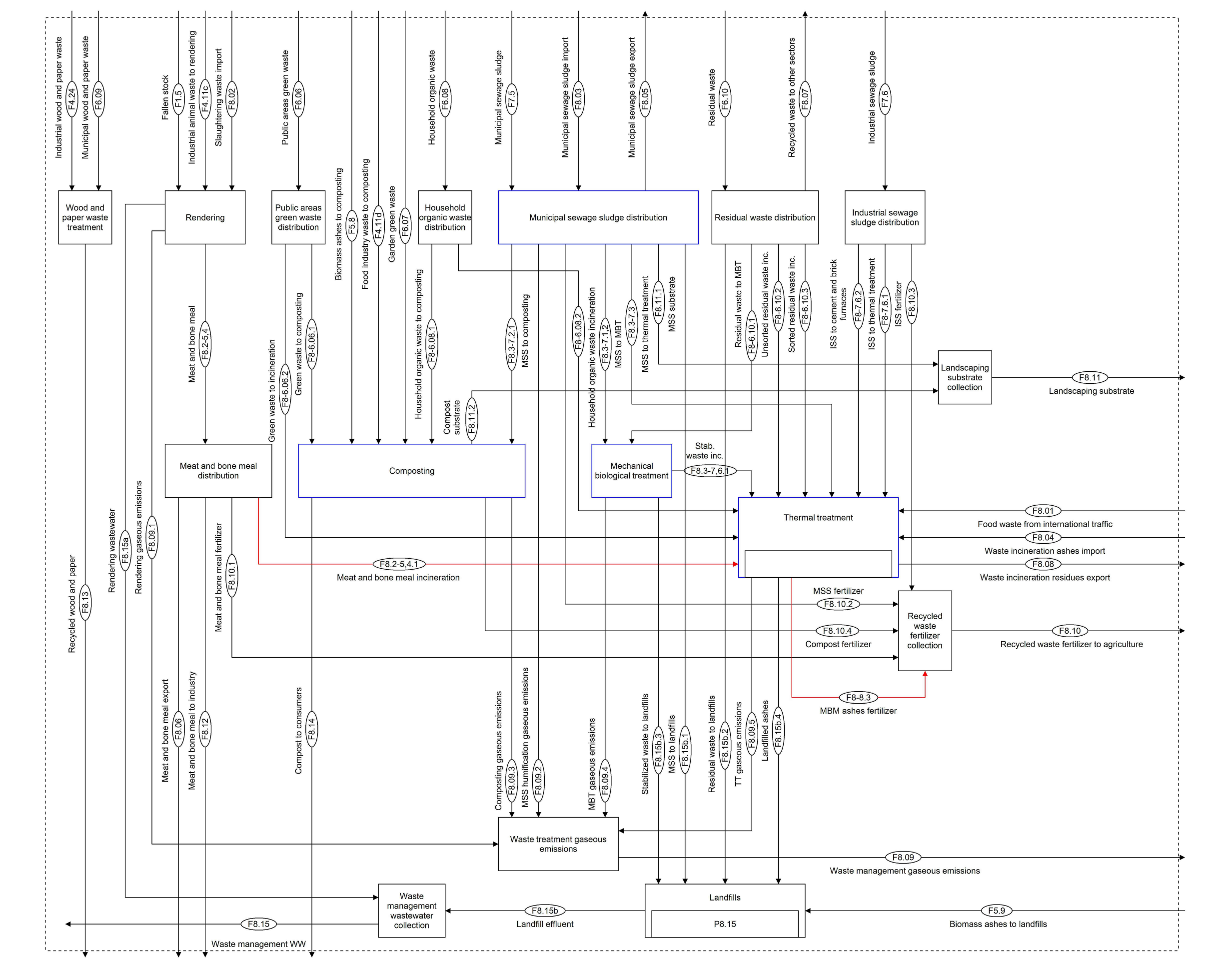

Fig. A.41: MFA structure of the Austrian P-N system 2015 - Subsystem *Waste management*. Changes to the original model structure under the measure *Increased nutrient recovery from meat and bone meal*. Changes to the original model structure are marked in red.

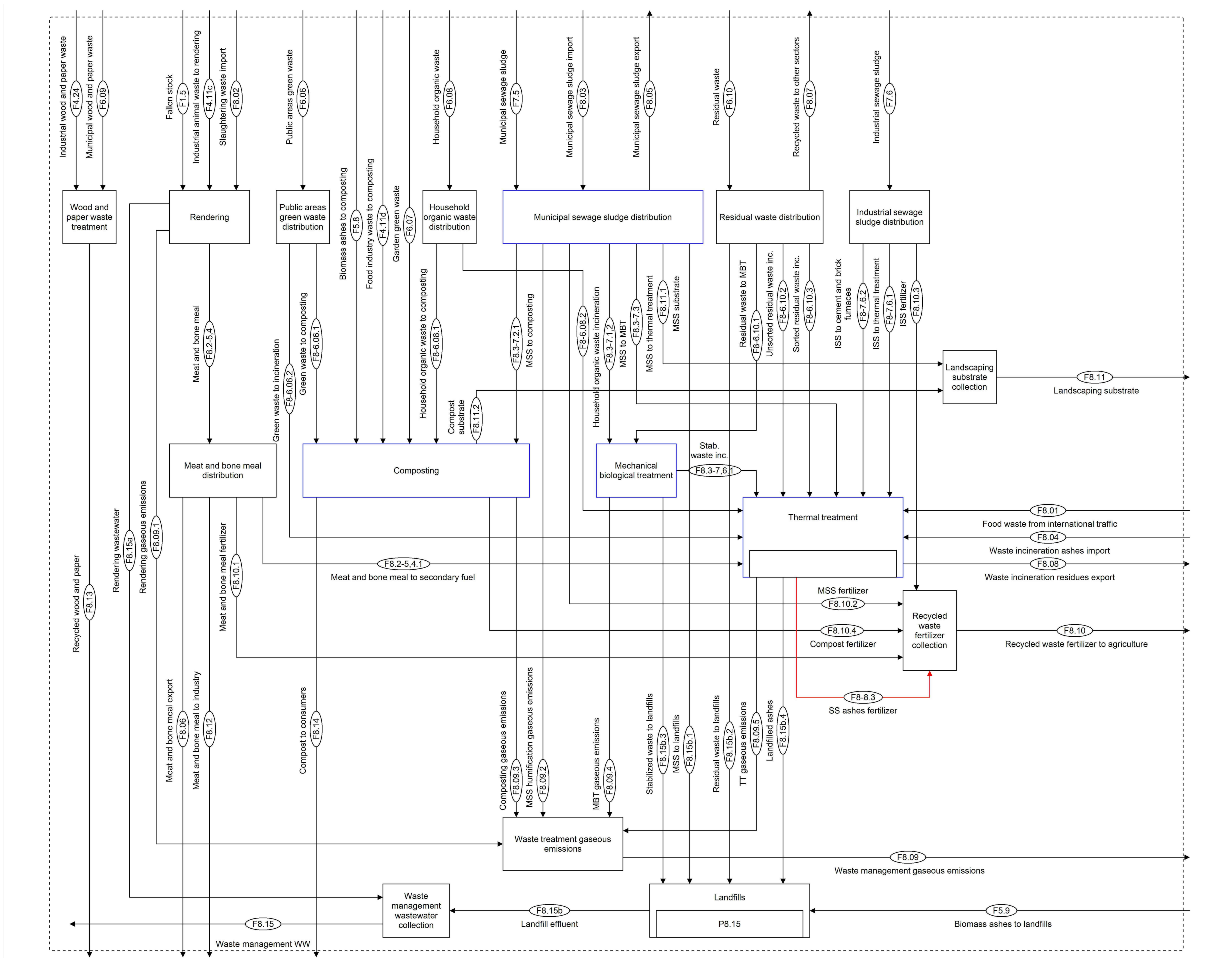

Fig. A.42: MFA structure of the Austrian P-N system 2015 - Subsystem *Waste management*. Changes to the original model structure under the measure *Increased P-recovery from sewage sludge*. Changes to the original model structure are marked in red.

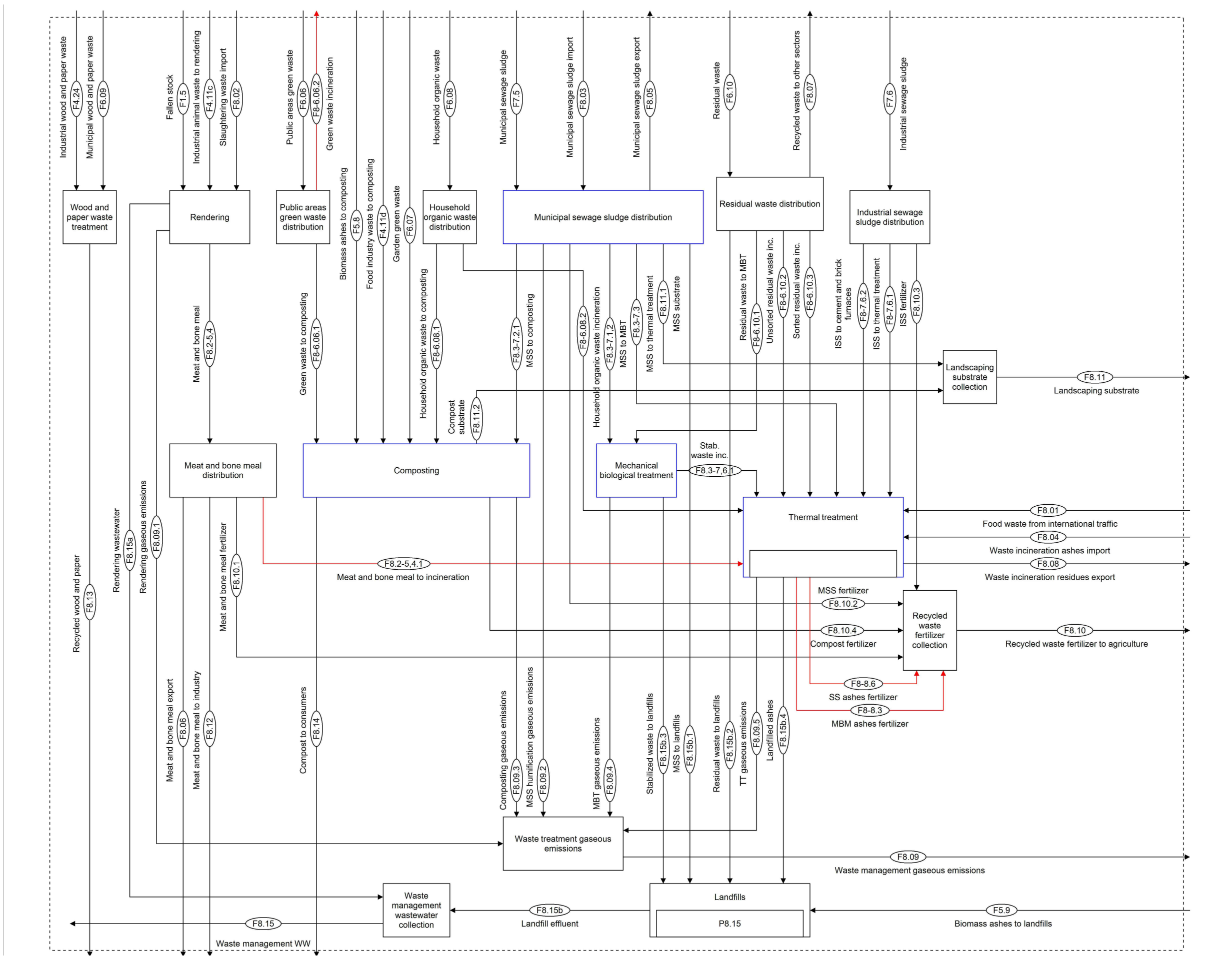

Fig. A.43: MFA structure of the Austrian P-N system 2015 - Subsystem *Waste management*. Changes to the original model structure under the Combined measures scenario. Changes to the original model structure are marked in red.

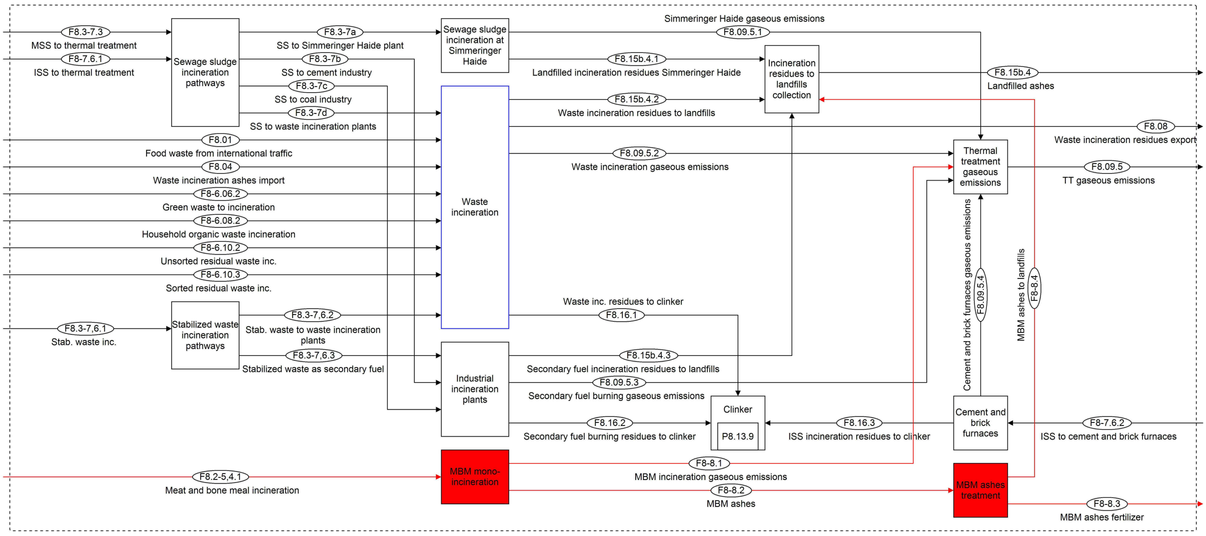

Fig. A.44: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Thermal treatment*. Changes to the original model structure under the measure *Increased nutrient recovery from meat and bone meal*. Changes to the original model structure are marked in red.

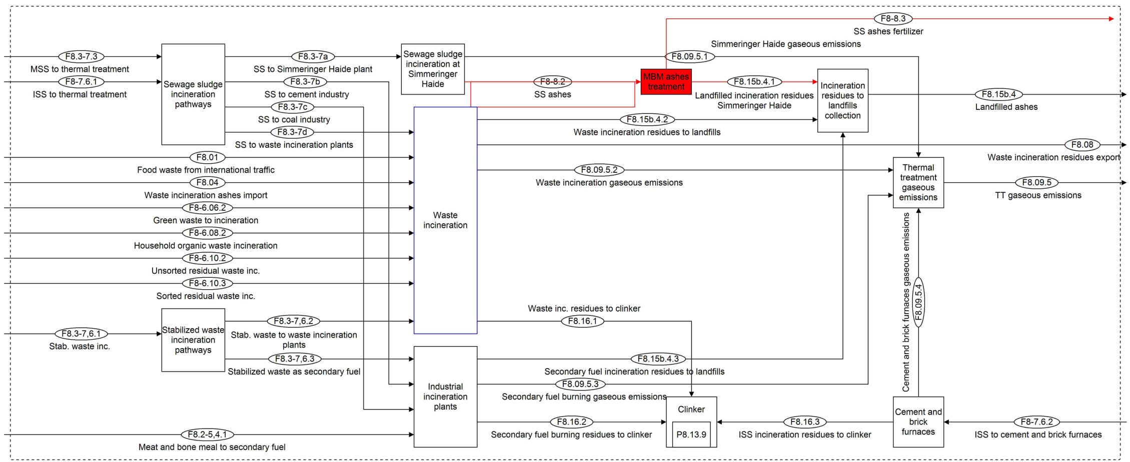

Fig. A.45: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Thermal treatment*. Changes to the original model structure under the measure *Increased P-recovery from sewage sludge*. Changes to the original model structure are marked in red.

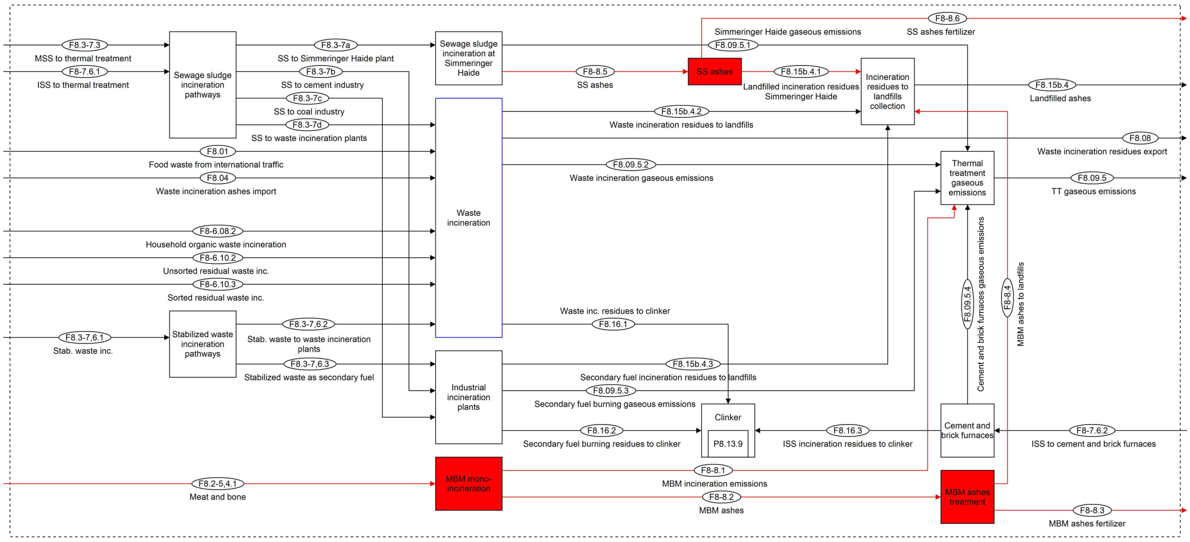

Fig. A.46: MFA structure of the Austrian P-N system 2015 - Subsubsystem *Thermal treatment*. Changes to the original model structure under the Combined measures scenario. Changes to the original model structure are marked in red.

A.1.2 Multi-level generic framework

Subsystems beyond the 2nd level are not shown, as they correspond to the depiction in Appendix A.1.1.

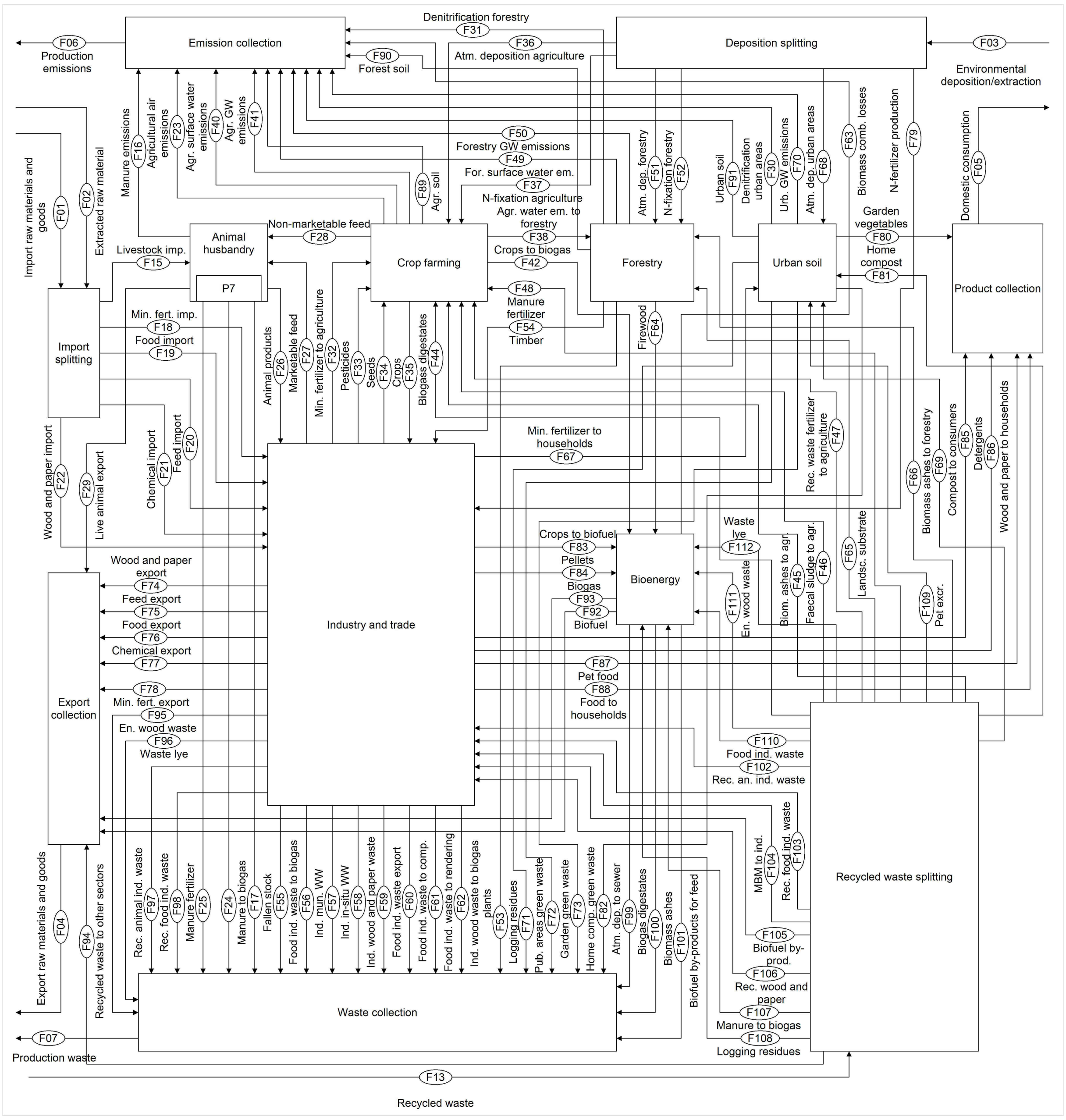

Fig. A.47: MFA structure of the Austrian P-N system 2015 in the multi-level generic framework - Subsystem *Production*.

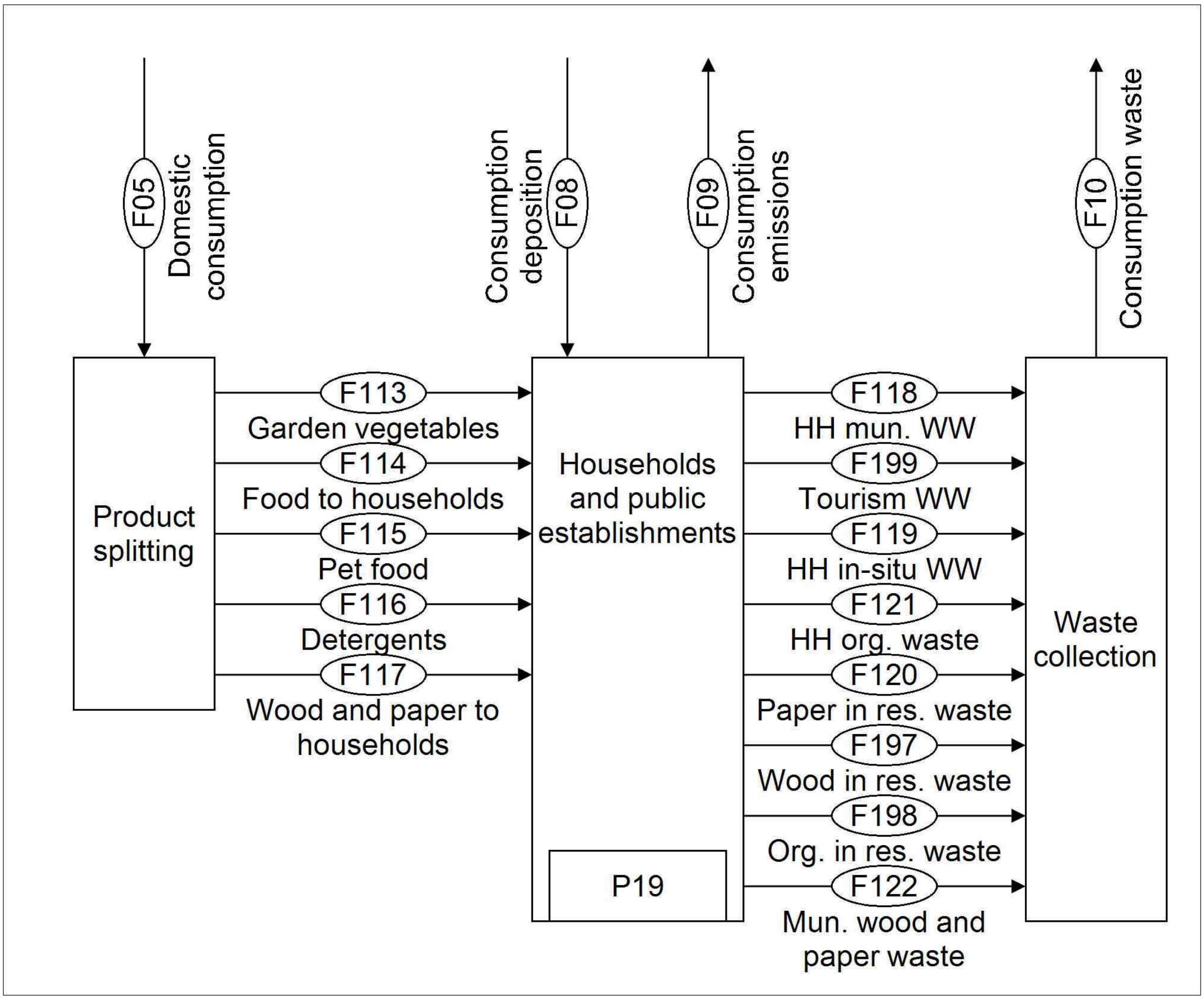

Fig. A.48: MFA structure of the Austrian P-N system 2015 in the multi-level generic framework - Subsystem *Use*.

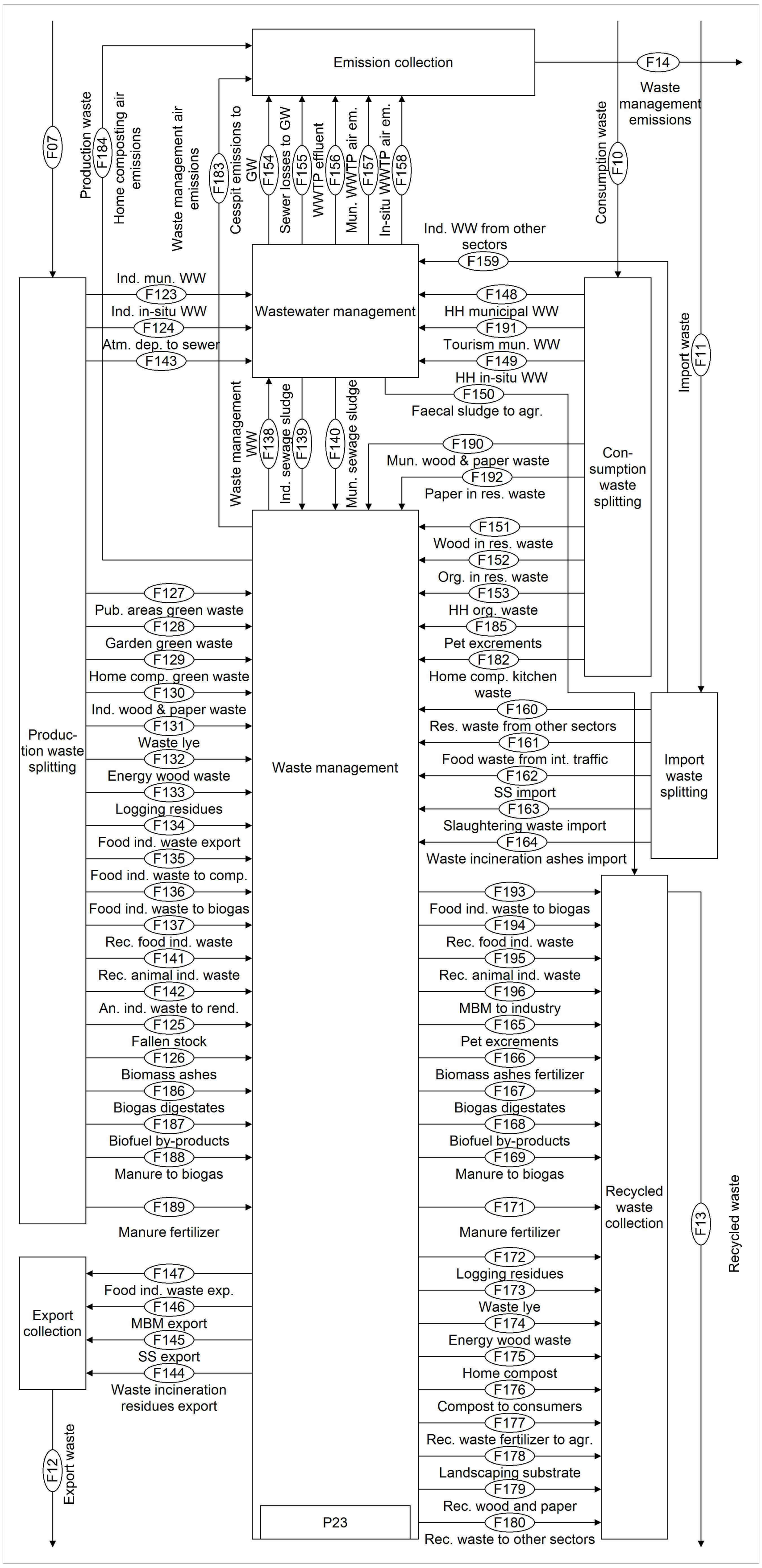

Fig. A.49: MFA structure of the Austrian P-N system 2015 in the multi-level generic framework - Subsystem *Waste management*.

A.2 EU 28 copper system 2014

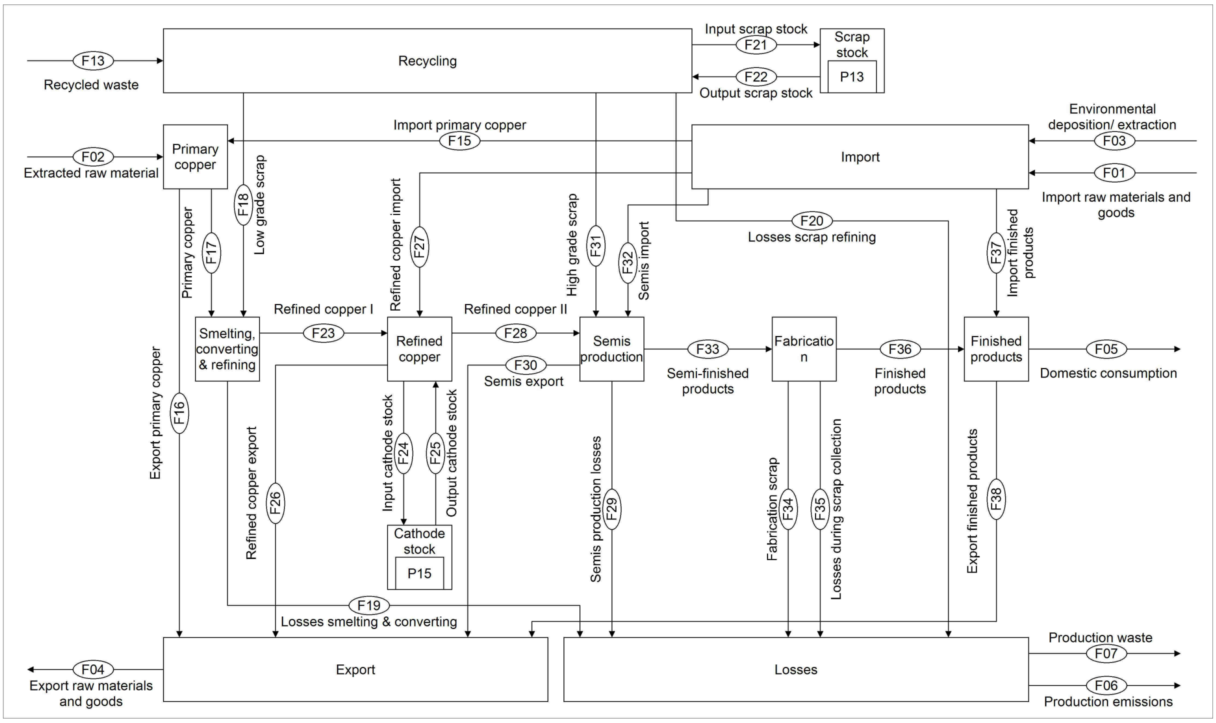

Fig. A.50: MFA structure of the EU 28 copper system 2014 in the multi-level generic framework - Subsystem *Production*.

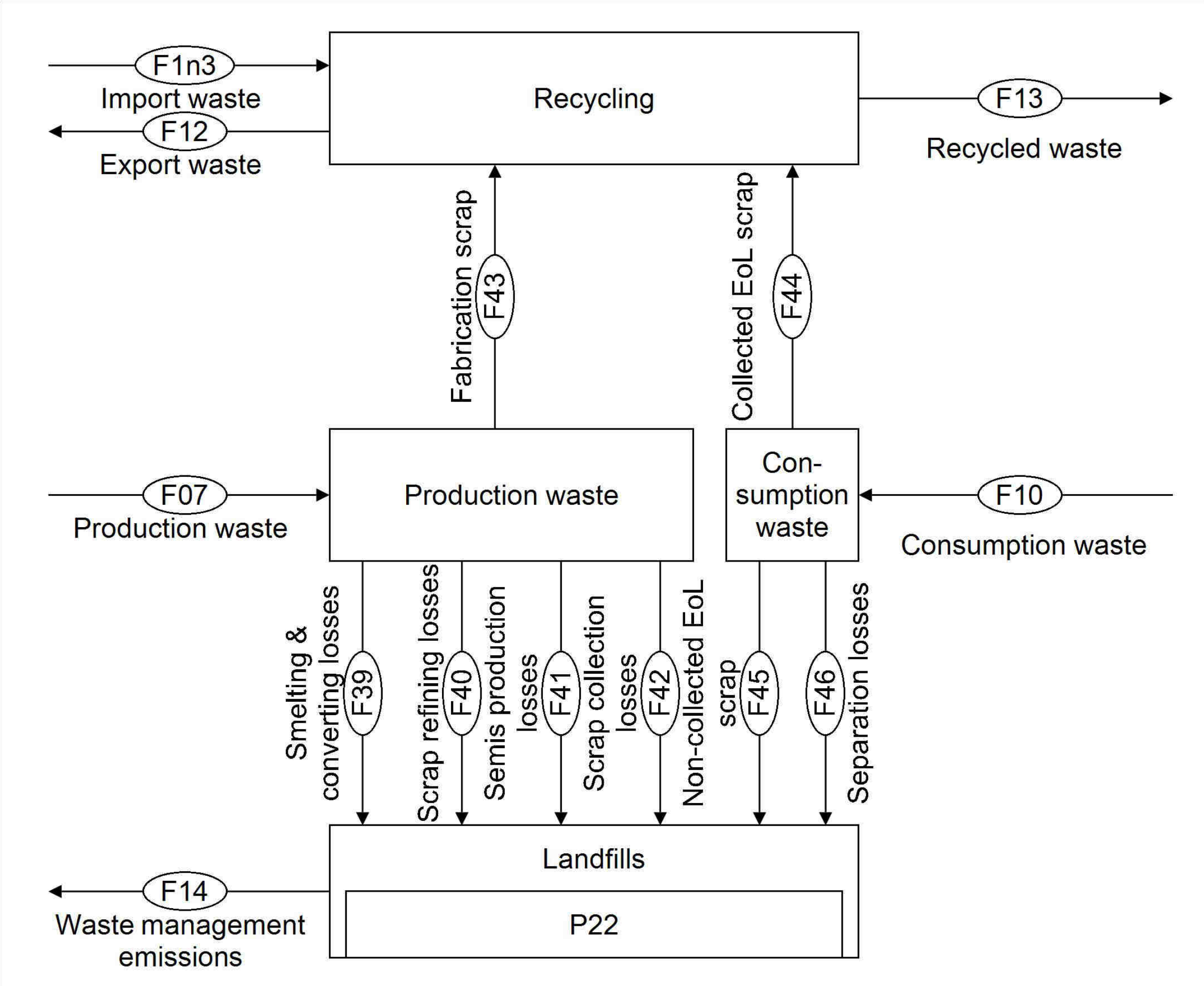

Fig. A.51: MFA structure of the EU 28 copper system 2014 in the multi-level generic framework - Subsystem *Waste management*.

A.3 Karlskoga combined heat and power plant 2016

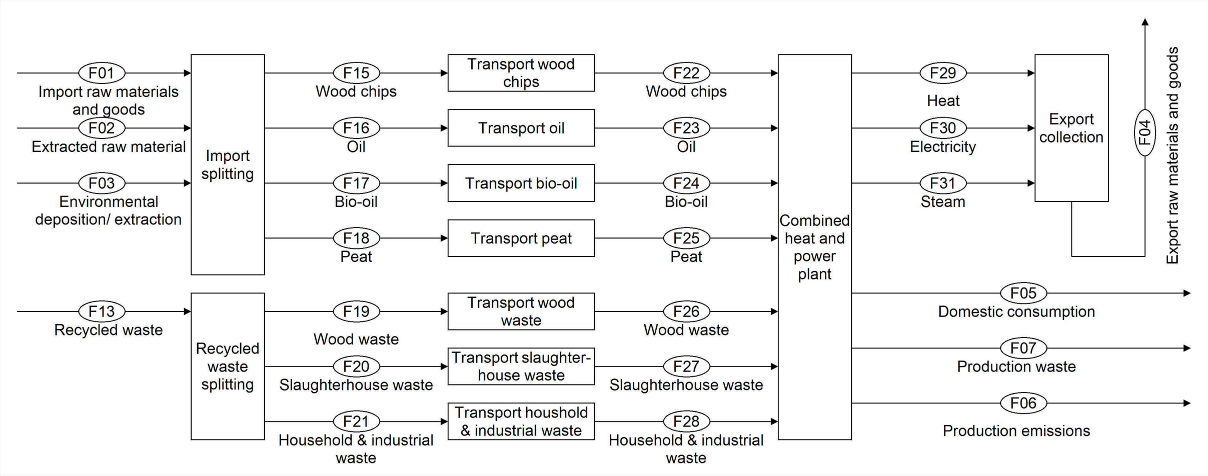

Fig. A.52: MFA structure of the Karlskoga combined heat and power plant 2016 in the multi-level generic framework - Subsystem *Production*.

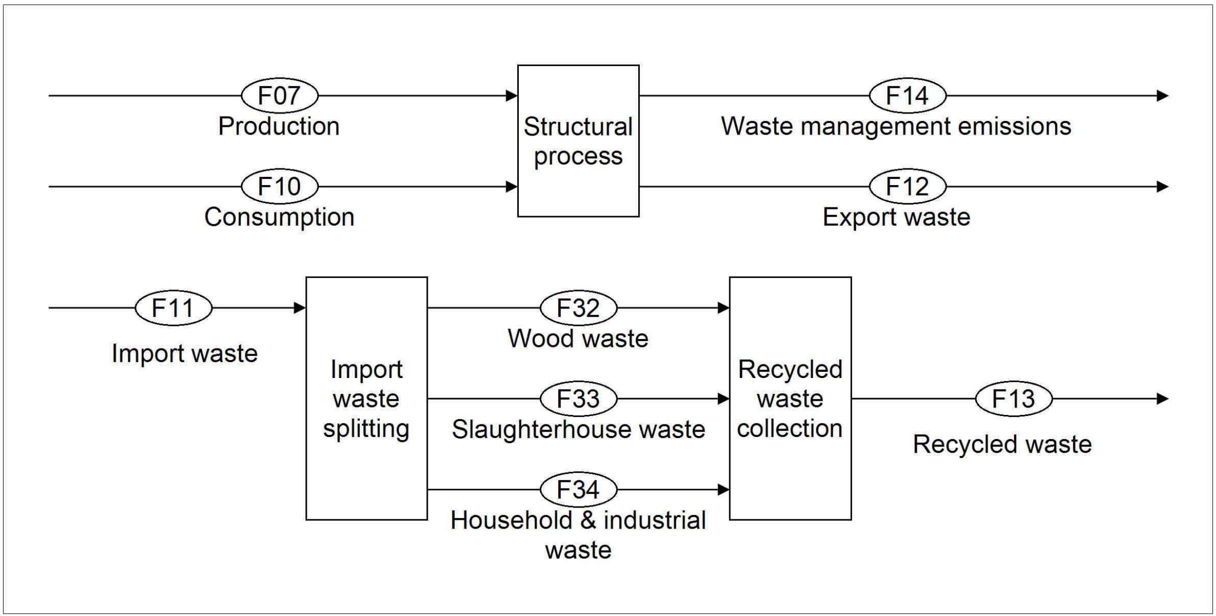

Fig. A.53: MFA structure of the Karlskoga combined heat and power plant 2016 in the multi-level generic framework - Subsystem *Waste management*.

Appendix B

Additional information on applied formulas

B.1 Calculation of EIPs

$$MFD_{P/N} = F4.04a + F4.04b - F4.09 \tag{B.1}$$

$$LWS_{P/N} = \Delta P8.13.9 + \Delta P8.15 + F4.11a + F8.05 + F8.06 + F8.08 \tag{B.2}$$

$$WE_{P/N} = F2.8 + F2.9 + F3.7 + F3.8 + F6.11 + F7.7 + F7.8 + F7.9 - F9.3 - F9.4 \tag{B.3}$$

$$SA_P = \Delta P2.2 + \Delta P3.1 + \Delta P6.01.1 \tag{B.4}$$

$$AE_N = F1.3c + F2.3 + F3.3 + F5.1 + F6.03 + F6.12 + F7.2 + F7.3 + F8.09 + F9.3 + F9.4 \tag{B.5}$$

where:

$\mathrm{MFD_{P/N}}$: Demand of mineral P/N fertilizer for domestic use [t/a]
$\mathrm{LWS_{P/N}}$: P/N losses in waste sector [t/a]
$\mathrm{WE_{P/N}}$: P/N emissions to ground- and surface water [t/a]
$\mathrm{SA_P}$: P accumulation in soil [t/a]
$\mathrm{AE_N}$: N emissions to atmosphere [t/a]

and Fx refers to the mass flow of P/N as listed in the enclosed electronic supplementary material.

B.2 Natural background concentrations and imission limits

Tab. B.1: Natural background concentrations and imission limits for gaseous and aqueous emission flows in the Austrian P-N system.

Media	Substance/ compound	c_{geog}	Origin	Reference
Atmosphere	N_2-N	0.637	Tropospheric N_2 concentration	Salby 2012
	NH_3-N	3.02*10^{-10}	Mean global NH_3 concentration	Roney et al. 2004
	NO_x-N	7.46*10^{-10}	Imission limit (annual mean) for vegetation protection	Spangl and Nagl 2016
	N_2O-N	1.66*10^{-7}	Tropospheric N_2 concentration	Salby 2012
Surface water	NH_4-N	3.83*10^{-8}	Annual mean (average of Austrian measurement sites)	Philippitsch et al. 2018
	NO_3-N	5.00*10^{-6}	Imission limit under good ecological status	BMLFUW 2010b
	N_{org}-N	9.58*10^{-9}	25% of c_{geog} NH_4	Sobańtka et al. 2012
	P (all forms)	2.15*10^{-7}	Imission limit under good ecological status	BMLFUW 2010b
Groundwater	NH_4-N	3.49*10^{-7}	Threshold value for NH_4	BMLFUW 2010a
	NO_3-N	1.02*10^{-5}	Threshold value for NO_3	BMLFUW 2010a
	N_{org}-N	8.74*10^{-8}	25% of c_{geog} NH_4	Sobańtka et al. 2012
	P (all forms)	9.78*10^{-8}	Threshold value for PO_4	BMLFUW 2010a
Agricultural soil	P (all forms)	4.70*10^{-5}	Fertility requirement for cropland and pastures	Baumgarten et al. 2006
Forest/natural soil	P (all forms)	1.30*10^{-5}	Fertility requirement for spruce and pine	Fürst 2014
Urban soil	P (all forms)	4.70*10^{-5}	Same fertility requirement as for arable land	Assumption
Landfills	N (all forms)	0.003	Concentration in combined household/industrial landfills	Östman et al. 2006
	P (all forms)	0.008	Concentration in landfills for coarse ashes	Lederer et al. 2014