# Towards a Sustainable Urban Metabolism:

# Algae-to-Energy Systems as

# Clean Cycles

# in the Urban Water Chain

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**Saubere Wiederverwertungszyklen für einen nachhaltigen urbanen Metabolismus: Integration von Algensystemen zur Energiegewinnung in die urbane Wasserkette**

*Englischer Titel: Towards a sustainable urban Metabolism: Algae-to-Energy Systems as Clean Cycles in the urban Water Chain*

Dissertation vorgelegt von Dipl.-Geoökologin Eve Menger-Krug

Auf dem Weg hin zu einer nachhaltigen Zukunft müssen die Städte viele Herausforderungen meistern. Der urbane Metabolismus muss reorganisiert werden, von der heutigen linearen Form zu einer kreislauforientierten Form mit höherer metabolischer Effizienz und „sauberen Wiederverwertungszyklen". Infrastrukturen managen einen Großteil der Material- und Energieflüsse in urbanen Gebieten. Daraus ergibt sich die Frage, wie Infrastruktursysteme reorganisiert werden müssen um zu sauberen Wiederverwertungszyklen beizutragen. Der Fokus der vorliegenden Arbeit liegt auf der urbanen Wasserkette: von der Wasserversorgung bis zur Abwasser- und Klärschlammentsorgung; und dem potentiellen Beitrag zu sauberen Wiederverwertungszyklen durch Integration von Algensystemen zur Energiegewinnung.

Die Integration von Algensystemen zur Energiegewinnung in die urbane Wasserkette ist ein vielversprechendes Konzept. Kohlenstoff (C), Stickstoff (N) und Phosphor(P) sind wichtige Bestandteile des Abwassers und die Hauptnährstoffe für das Algenwachstum. Abwasser und Abwasserteilströme können als Nährmedium verwendet werden und die produzierte Biomasse kann zur Bioenergie Gewinnung eingesetzt werden, beispielsweise als Co-Substrat bei der Klärschlammvergärung. Algensysteme sind geschlossene Systeme. Dies ist ein wichtiger Aspekt, da Abwasser neben den Ressourcen CNP auch viele anthropogene Mikroschadstoffe (AMS) enthält. Dies beinhaltet Haushaltschemikalien, Pharmazeutika und Schwermetalle. Bei umweltoffenen Anwendungen gelangen diese AMS in die Umwelt.

Neben der Energiegewinnung können Algensysteme auch die Fracht von AMS im Ablauf reduzieren. Algen erreichen beachtliche Eliminationsraten für viele AMS unter Laborbedingungen, zudem ist bekannt, dass sie in der Umwelt AMS aufnehmen. Die wichtigsten Prozesse sind Sorption und Bioakkumulation, die AMS von der Wasserphase in die Biomasse transferieren. Dadurch ist die Biomasse ungeeignet für einige Anwendungen, wie Tierfutter oder

Düngemittel. Allerdings ist sie geeignet für die energetische Wiederverwertung, die in dieser Arbeit untersucht wird, wie beispielsweise Co-Vergärung und Co-Verbrennung mit Klärschlamm.

Das Ziel dieser Studie ist die Analyse der Integration von Algensystemen in die urbane Wasserkette und der Auswirkungen auf die Energie- und Emissionsbilanz im Vergleich zu dem Status quo. Zusätzlich wird die Relevanz der urbanen Wasserkette - mit Algensystemen und ohne Algensysteme - im Kontext des urbanen Metabolismus untersucht.

Dementsprechend ist die Studie in drei Teile aufgeteilt. Sie beginnt mit der Analyse des Status quo der urbanen Wasserkette in Deutschland. Vor diesem Hintergrund wird die Integration von Algensystemen auf Kläranlagen untersucht, als Konzept zur Erhöhung der metabolischen Effizienz. Im dritten Teil wird der Fokus der Studie erweitert um die Relevanz der urbanen Wasserkette im Kontext des urbanen Metabolismus zu untersuchen. Dabei werden drei wichtige Aspekte berücksichtigt: die urbane Energiebilanz, sowie die urbanen Flüsse von Nährstoffen und anthropogenen Mikroschadstoffen (AMS).

Methodisch findet eine Systemanalyse Anwendung. Sie kombiniert eine konventionelle Energiebilanz mit einer Substanzflussanalyse (SFA) sowie der Quantifizierung des energetischen Wertes der Ressourcen CNP. Das zugrundeliegende Modell besteht aus drei Ebenen. Es werden Fälle definiert, die verschiedene Prozessabläufe für die Wasserversorgung, für die Abwasser- und Schlammentsorgung, sowie für die Algensysteme abbilden. Die Fälle bilden die Grundlage für das Modell (1. Ebene). Für jeden der Fälle werden die Energieverbräuche in Form von Strom, Wärme und Treibstoffen für die verschiedenen Stationen der urbanen Wasserkette zusammengestellt. Zusammen mit der Energieerzeugung durch Biogas (auch Klärgas genannt) aus der anaeroben Schlammstabilisierung und durch die Klärschlammverbrennung, ergibt dies die externen Energieflüsse (konventionelle Energiebilanz, 2. Ebene).

Zur Erweiterung der konventionellen Energiebilanz, erfasst das Model außerdem den energetischen Wert der Ressourcen in den Stoffströmen. Das theoretische Energiepotenzial (TEP) von CNP wird quantifiziert. Es entspricht der maximalen Energiemenge, die aus den Ressourcen gewonnen werden kann und bezieht sich auf die chemische Energie von C und die "graue" Energie von N und P. Die graue Energie entspricht der Energie für die Herstellung einer äquivalenten Menge Düngemittel. Durch Anwendung der TEP-Faktoren (Energie pro Masse) auf die SFA kann das Modell die "internen" Energieflüsse entlang der urbanen Wasserkette abbilden. Die erweiterte Energiebilanz gibt somit ein holistisches Bild der Energieflüsse und erlaubt die Ableitung der metabolischen Effizienz: des Grads der Wiederverwertung bezogen auf das volle Energie-Potenzial der Ressourcen in den Stoffströmen CNP.

Das Konzept der sauberen Widerverwertungszyklen verlangt, dass die energetische Wiederverwertung von CNP nicht nur unter quantitativen, sondern auch unter qualitativen Gesichtspunkten analysiert wird. Deshalb wird die erweiterte Energiebilanz ergänzt durch eine Emissionsbilanz. Die Emissionsbilanz beinhaltet eine holistische Perspektive auf alle Umweltkompartimente. So erfasst die Analyse die Doppelrolle von CNP. CNP sind energetische Ressourcen, aber auch Schadstoffe wenn sie in die Umwelt emittiert werden. Weiterhin wird eine Modellsubstanz in die Analyse mit einbezogen (Perfluoroctansulfonat PFOS) um die Problematik der anthropogenen Mikroschadstoffe (AMS) zu diskutieren.

Obwohl das heutige System der urbanen Wasserkette zuverlässig arbeitet und die Hauptfunktionen bezüglich Trinkwasserversorgung, Siedlungshygiene und Schutz der Gewässer vor Eutrophierung erfüllt, ist die metabolische Effizienz niedrig. Für CNP zusammengenommen, liegt die metabolische Effizienz bei nur 13%. Bei der Wiederverwertung von N und P aus Klärschlamm in umweltoffenen Anwendungen in der Landwirtschaft gelangen außerdem AMS in die Umwelt. Sogar mit optimierten Prozessen für Biogasverwertung und Schlammverbrennung, bleibt die metabolische Effizienz für C unter 40%. Es ist bemerkenswert, dass die nicht wiederverwerteten Energiepotenziale im Betrag grösser sind als der gesamte externe Energiebedarf der urbanen Wasserkette. Während die vollständige energetische Wiederverwertung gegebenenfalls weder technisch noch ökonomisch machbar ist, unterstreichen die Ergebnisse die Relevanz von Konzepten zur Erhöhung der metabolischen Effizienz.

Die Integration von Algensystemen zur Energiegewinnung ist ein solches Konzept. In der vorliegenden Arbeit wird ein Prozessablauf für die Integration von Algensystemen auf Kläranlagen erarbeitet. Die gesamte Biomasseerzeugungs-und verwertungskette bis hin zur Stromerzeugung findet auf der Kläranlage statt und benötigt keine Ressourcen von außerhalb der Kläranlage. Während Algensysteme zur Energiegewinnung in der Literatur große Aufmerksamkeit zukommt, ist diese Arbeit die Erste, die den integrierten Prozess im Detail analysiert. Basierend auf einer SFA der Hauptnährstoffe CNP werden die Implikationen für Energie- und Emissionsbilanz erarbeitet. In einer Szenarioanalyse werden wichtige Einflussfaktoren identifiziert, darunter die Ernteeffizienz und die anaerobe Abbaubarkeit der Biomasse. Die Ergebnisse zeigen, dass aus der Stoffstromperspektive betrachtet, Algensysteme die Kläranlage in einen Energieproduzenten verwandeln können. Dazu wird eine Fläche von 6 m² pro angeschlossenem Einwohner benötigt, aber keinerlei externe Ressourcen wie Wasser, Düngemittel oder $CO_2$. Neben dem hohen Flächenbedarf, der allerdings pro Energiemenge geringer ist als für andere Energiepflanzen, haben Algensysteme einen weiteren Nachteil. Obwohl die Grenzwerte eingehalten werden, erhöhen Algensysteme die Fracht von CNP im Ablauf. Die höhere Fracht ist vor allem auf die nicht geerntete Biomasse zurückzuführen, die im Ablauf verbleibt. Durch Nachbehandlung des Ablaufs kann dieser Effekt minimiert werden, mit moderatem Einfluss auf die Energiebilanz.

Im Kontext des urbanen Metabolismus sind zur Bewertung der Algensysteme drei wichtige Aspekte hervorzuheben. Erstens, im Kontext der urbanen Energiebilanz kann die konsequente energetische Wiederverwertung der C Ressourcen aus Abwasser in Algensystemen erheblich zur Stromerzeugung aus erneuerbaren Quellen beitragen. Die Bioenergie aus Algensystemen kann ein wichtiger Pfeiler der zukünftigen nachhaltigen Energiesysteme sein, da Bioenergie im Gegensatz zu den meisten erneuerbaren Energien grundlastfähig und regelbar ist.

Zweitens ist die Wiederverwertung von N und P aus Abwasser sehr relevant im Kontext des urbanen Metabolismus. Die Stoffströme im Abwasser repräsentieren einen Großteil der urbanen Nährstoffflüsse von N und P. Aber die urbane Wasserkette ist auch ein wichtiger Emissionspfad für die anthropogenen Mikroschadstoffen (AMS), welches der dritte wichtige Aspekt für die Analyse ist. In Algensystemen können die Nährstoffe aus Abwasser ohne Emissionen von AMS während der Biomasseerzeugung wiederverwertet werden, anders als in umweltoffenen Anwendungen in der Landwirtschaft. Dies charakterisiert die Algensysteme zur Energiegewinnung auf Kläranlagen als „saubere Wiederverwertungszyklen". Desweiteren haben die Algensysteme das Potenzial zu „reinigenden Wiederverwertungszyklen". Durch Sorption und Bioakkumulation können Algensysteme auf Kläranlagen die Fracht verschiedenster AMS im Ablauf reduzieren. Mit dem vorgeschlagenen Prozessablauf zur Integration von Algensystemen werden die meisten AMS dann bei der Co-Verbrennung von Klärschlamm und Algenbiomasse mineralisiert. So wird die Emission von AMS von der urbanen Wasserkette reduziert, was im Kontext der urbanen Flüsse von AMS von hoher Relevanz ist. In diesem Sinne hat auch die Fläche, die den Algensystemen gewidmet ist eine doppelte Funktion: zur Bioenergiegewinnung und zur Elimination von AMS aus Abwasser während relativ langer hydraulischer Verweilzeiten.

Die untersuchten Aspekte: die Errichtung eines nachhaltigen urbanen Energiesystems, das kreislauforientierte Management der urbanen Nährstoff-Flüsse und die Minimierung der urbanen Emissionen von AMS sind heute von großer Wichtigkeit und werden in der Zukunft noch wichtiger werden. Während Algensysteme auf Kläranlagen für keine dieser Herausforderungen eine allumfassende Lösung bieten, können die Synergien, welche durch die Algensysteme ermöglicht werden, zur Lösung aller drei Herausforderungen beitragen.

Die Ergebnisse der Analyse unterstreichen das Potenzial von Algensystemen zur Energiegewinnung auf Kläranlagen und zeigen die Machbarkeit von der Stoffstromperspektive aus betrachtet. Somit ist eine weitergehende Forschung anhand von Pilot-und Demonstrationsanlagen vielversprechend. Für Pilot-und Demonstrationsanlagen sind die Ergebnisse dieser Arbeit hilfreich, da das integrierte System mit seinen Wechselwirkungen im Modell dargestellt ist. Durch Anpassung des Modells an konkrete Anlagen, können Informationen über die zu erwartende Energieausbeute, über Änderungen der Auslastungen anderer Behandlungsstufen und Rückflüsse innerhalb des Systems, sowie über die zu erwartenden Ablaufwerte bereitgestellt werden. Komplementär können Daten von Pilot-und Demonstrationsanlagen genutzt werden, um das Modell zu verfeinern. In diesem Sinne kann das vorgestellte Modell als Werkzeug zur Systemoptimierung genutzt werden.

In dieser Arbeit werden Algensysteme zur Energiegewinnung auf Kläranlagen bezüglich Stoffströmen und Energieflüssen analysiert. Weiterhin gibt es noch viel Forschungsbedarf zu Fragen der Ökonomie, der nötigen politischen Rahmenbedingungen und der Akzeptanz. Die ökonomische Bewertung von Algensystemen ist abhängig von der zukünftigen Entwicklung der Energiepreise, welche wiederrum von den politischen Rahmenbedingungen gesteuert werden. Heute wird beispielsweise die Energie aus Abwasserressourcen in Form von Klärgas geringer vergütet als Bioenergie von Energiepflanzen.

In dieser Arbeit wird eine Methode zur Analyse der Stoff-und Energieflüsse von Wasser-und Abwasserinfrastrukturen erarbeitet: die erweiterte Energiebilanz. Sie erweist sich als nützliches Werkzeug zur Analyse von kreislauforientierten Konzepten, da sie zusätzlich zu den externen Energieflüssen auch die internen Energieflüsse und die metabolische Effizienz des Systems abbildet. Wie durch den Ansatz der sauberen Wiederverwertungszyklen vorgegeben, wird neben der Energiebilanz auch die Emissionsbilanz erfasst, inklusive der Emissionen von AMS anhand einer Stellvertretersubstanz. Diese Methode kann auch für die Analyse anderer kreislauforientierter Konzepte nützlich sein.

**Towards a Sustainable Urban Metabolism:**
**Algae-to-Energy Systems as Clean Cycles in the urban Water Chain**

On the way to a sustainable future, there is mounting pressure to reorganize the urban metabolism from its present linear form towards higher metabolic efficiency and clean cycles. This applies also to the urban water chain, which is an important part of the urban metabolism. The focus of this study is the integration of algae-to-energy systems in the urban water chain. This is a promising concept to recycle nutrients from wastewater. The elements carbon (C), nitrogen (N) and phosphorus (P) are major constituents of wastewater, and key nutrients for algae growth. Algae systems produce biomass that can be harvested for bioenergy generation. Algae cultivation represents a closed system for recycling. This is an advantage, because wastewater also carries considerable amounts of anthropogenic micropollutants (AMPs) from urban areas, such as household chemicals, pharmaceuticals or heavy metals, mirroring the common use of chemicals in modern society.

The scope of this study includes the status quo of the urban water chain in Germany, including the water supply as well as the wastewater and sludge management (part 1). In the second part, a detailed analysis of the integration of algae systems on the level of wastewater treatment plants (WWTPs) is presented. Then, the scope of the study is extended to put the results in context of the urban metabolism (part 3). The relevance of the urban water chain - with and without algae - for the urban energy balance and the urban flows of nutrients and AMPs is assessed.

Methodologically, a systems analysis is employed combining a conventional energy balance with a substance flow analysis (SFA) and the assessment of the energetic value of the resources. Different technical setups (cases) for water supply, wastewater and sludge management – with and without algae systems; are investigated. The gross consumption for handling and treatment of flow streams is compiled: electricity, thermal energy and fuel consumption for each step of the urban water chain. Together with the own generation of energy from biogas use or sludge incineration, these energy flows represent the external energy flows of the system.

To extend the usual approach to energy balances, the analysis also assesses the internal energy flows. The theoretical energy potential (TEP) is assigned to the resources CNP. It reflects the energetic value of the resources in the flow streams, related to the chemical energy of C and the "grey" energy of the nutrients N and P, which gain indirect energy credits when substituting energy intensive fertilizers. By applying the respective TEP factors (energy per unit mass) to the SFA, the internal energy flows associated to CNP are traced along the urban water chain. This extended energy balance gives a holistic picture of the energy flows along the urban water chain and allows assessing the metabolic efficiency: the degree of reuse in relation to the full energetic value of the resources.

The clean cycle approach requires assessing not only the quantity, but also the quality of CNP recycling. Therefore, the extended energy balance is accompanied by the emission balance. The emission balance for CNP includes a systemic perspective on all compartments. The framework for analysis captures the double role of CNP: they are potential energetic resources or act as pollutants when misplaced to the environment. In addition, a model substance (PFOS) is included in this study to discuss the problem of anthropogenic micropollutants (AMPs).

While the current system works reliably and fulfills its main functions for public health and protection of water resources, the results show the low metabolic efficiency of the urban water chain today. For the resources CNP, the average metabolic efficiency is 13%. Even with optimized biogas and incineration processes, the metabolic efficiency for C stays below 40%. It is noteworthy that the non-recovered energy potentials are higher than the primary energy demand of the urban water chain. While a full energetic reuse may not be technically or economically feasible, the results underline the importance of concepts for increased metabolic efficiency of the urban water chain.

The integration of algae systems at wastewater treatment plants (WWTPs) is such a concept. In this study, a technical setup is proposed to integrate algae systems in the existing treatment steps of WWTPs. The whole algae process chain, from cultivation to production of bio-electricity, takes place at the WWTP, relying only on the resources available on site. While algae systems receive much attention, this is the first detailed study of the integrated system. Based on the SFA of the major nutrients CNP, the implications for energy and emission balance of the WWTP are elaborated. A scenario analysis highlights the harvesting efficiency and the anaerobic digestibility of algae biomass as key factors for the performance of algae systems. The results show that the bio-electricity from algae systems can turn the WWTP into an energy producer, requiring a cultivation area of 6 $m^2$ per person served by the WWTP but no external input of fertilizer, water or $CO_2$. On the downside, while meeting limit values, the algae systems increase the load of CNP in effluent in absence of post treatment. Algae systems also have a large area demand, albeit lower than other energy crops

The analysis of algae systems in context of the urban metabolism highlights three important aspects. Firstly, in context of the urban energy balance, the consequent energetic reuse of resources from wastewater in algae systems can considerably contribute to electricity production from renewable sources on an urban scale. Bio-electricity is an important pillar for sustainable energy systems. Secondly, in context of the urban nutrient flows, the recycling of the resources of N and P in wastewater is highly relevant for a sustainable urban metabolism. These flows represent a large share of the urban nutrient flows. But the urban water chain is also a major pathway of AMPs, which is the third important aspect to consider. Algae systems can recycle nutrients from the urban water chain, without emission of AMPs during biomass cultivation, in contrast to “open” applications in agriculture. Thus, they can be characterized as a clean cycles. Furthermore, algae systems even have the potential for “cleaning cycles”.

By sorption and bioaccumulation, they can reduce the load of many AMPs in effluent. Most AMPs are degraded during sludge and algae co incineration in the investigated technical set-up, reducing the AMP emission from the urban water chain. This is highly relevant for a sustainable urban metabolism. Thus, the area designated to algae systems has a double function: to produce bioenergy and to allow for a long hydraulic retention time for AMP elimination from wastewater.

All three topics discussed above: sustainable energy systems, the cycle oriented management of nutrients, and the chemical pollution problem, are important today and their importance will likely further increase in the future. While algae systems cannot provide the single solution for any of these challenges, the synergies they offer can contribute to solving all of them.

# Towards a Sustainable Urban Metabolism: Algae-to-Energy Systems as Clean Cycles in the Urban Water Chain

## Summary


*Aim of study*

On the way to a sustainable future, there is a mounting pressure to reorganize the urban metabolism from its present linear form towards a higher metabolic efficiency and clean cycles. The focus of this study is the urban water chain, including water supply and wastewater and sludge management. It is an important part of the urban metabolism. Firstly, it has an indisposable essential function for hygiene and public health in dense human settlements. Secondly, the urban water chain is a hot spot for anthropogenic emissions. This refers to the nutrients carbon (C), nitrogen (N) and phosphorus (P), which act as pollutants if misplaced in the ecosphere (misplaced resources). It also refers to anthropogenic micropollutants (AMPs), such as household chemicals, flame retardants, impregnation agents, cleansers, pharmaceuticals or heavy metals, for which the urban water chain is an important pathway. The load of AMPs mirrors the wide application of chemicals in modern society. Thirdly, the infrastructures are large energy consumers. While the handling of flow streams consumes considerable amounts of energy, they are harboring resources for energetic reuse. The resources CNP have an energetic value. Energetic reuse of C resources provides bio-electricity, for example via anaerobic processes (biogas) or incineration of sludge. N and P resources gain *indirect* energy credits when substituting energy intensive fertilizers in agricultural applications (grey energy). But the presence of AMPs in the flow streams is a challenge for reuse.

The integration of algae-to-energy systems in the urban water chain is a promising concept to introduce clean cycles. CNP are major constituents of wastewater, and the key nutrients for algae growth. The biomass produced in algae systems can be harvested for bioenergy generation. The cultivation represents a *closed system* for recycling. This is advantageous, considering the presence of AMPs. In “open” agricultural applications, these AMPs are emitted to the environment.

Furthermore, algae systems can also reduce the load of AMPs in effluent. Algae are known to accumulate AMPs in the environment. Under laboratory conditions, they reach considerable elimination rates for many AMPs and. The main processes are sorption and bioaccumulation, which transfer AMPs from effluent to algae biomass. The biomass is unsuitable for application as animal feed or fertilizer, but suitable for energetic use, for example co digestion and co incineration with sludge. The aim of this study is to analyze the integration of algae systems in the urban water chain in terms of energy balance and emission balance, compared to status quo, and to assess the relevance of the UWC – with and without algae systems - in context of the urban metabolism.

*Structure*

This study has three parts. It starts with a detailed description of the status quo of the urban water chain in Germany, including the metabolic efficiency *i.e.* the degree of reuse of resources CNP. Before this background, a concept for increased metabolic efficiency is evaluated in the second part: the integration of algae systems at wastewater treatment plants (WWTPs). In the third part: Connecting the urban water chain to urban metabolism, the scope of the analysis is extended to assess the relevance of the urban water chain - with and without algae systems - in context of the urban metabolism. Three important aspects are considered: the role of the urban water chain in context of the urban energy flows, the urban nutrient flows and the urban flows of anthropogenic micropollutants (AMPs).

*Methods*

Methodologically, a systems analysis is employed combining a conventional energy balance with a substance flow analysis (SFA) and the assessment of the energetic value of the resources. The underlying model includes three layers. The technical setup builds the foundation of the model (layer 1). Different technical setups (cases) are defined to describe the status quo of the urban water chain, including the water supply, the wastewater and the sludge management; and the integration of algae systems. For the cases, the gross consumption for handling and treatment of flow streams is compiled: electricity, thermal energy and fuel consumption for each step of the urban water chain. Together with the own generation of energy from biogas use or sludge incineration, these energy flows represent the external energy flows of the system (layer 2).

To extend the usual approach to energy balances, the analysis also assesses the energetic value of the resources in the flow streams, termed internal energy flows. The theoretical energy potential (TEP) is assigned to the resources CNP. It relates to the chemical energy of C and the "grey" energy of the nutrients N and P. By applying the respective TEP factors [kWh/kg] to the SFA, the internal energy flows associated to CNP are traced along the urban water chain. The metabolic efficiency for N and P relates to the amount applied to agricultural land

corrected for plant availability. C is energetically reused via biogas use or sludge incineration for electricity and heat co generation, as quantified in layer 2 of the model. Putting this energy generation in relation to the TEP of C gives the metabolic efficiency. This extended energy balance gives a holistic picture of the energy flows along the urban water chain and allows assessing the metabolic efficiency: the degree of reuse in relation to the full energetic value of the resources. The metabolic efficiency is a useful measure to describe the degree of circularity within the system.

The clean cycle approach requires assessing not only the quantity, but also the quality of CNP recycling. Therefore, the extended energy balance is accompanied by the emission balance of the urban water chain. The emission balance includes the emissions of CNP from the flow streams (on site emissions) to all environmental compartments based on SFA results. While the effluent quality is the traditional focus of the water sector, a systemic view allows assessing additional indicators. This includes the C efficiency of bio-electricity generation and the comparison of on site $CO_2$ emissions and off site $CO_2$ emissions related to the energy consumption of external energy. In addition, a model substance (PFOS) is included in this study to discuss the problem of anthropogenic micropollutants (AMPs).

The strength of the developed framework lies in three main aspects. Firstly, this study is unique in focusing on the material flows on elemental level, tracing the flows of the major elements of the biosphere CNP, which are also the main constituents of wastewater, along the full urban water chain. The emission balance includes a systemic perspective on all compartments. Secondly, the TEP factors applied to the SFA allow inclusion of the energetic value of resources and the metabolic efficiency. This extends the usual approach to the analysis of energy flows. The extended energy balance developed in this study, captures both: the energy required for handling of flow streams (external energy flows) as well as the energetic value of the resources CNP (internal energy flows). The framework captures the double role of CNP: they are resources with an energetic value but act as pollutants when misplaced to the environment. Thirdly, the problem of AMPs is taken into account, albeit only exemplified by one model substance.

*Results: Status quo of the urban water chain in Germany*

In the first part, the status quo of the urban water chain in Germany is assessed as baseline for comparison. While the focus lies on the post use side of the urban water chain, the scope of the study includes the full pathway of water through settlements: from sourcing of water to treatment and provision of tap water for water use in households, to transport and treatment of wastewater and sludge. Different cases are investigated. For water supply, sourcing and treatment from groundwater (30% of population served) and from surface water (70%) are included. For wastewater and sludge disposal, three technical setups are defined: the advanced case (anaerobic digestion of sludge followed by incineration, 52% of population served), the medium case (with land use of sludge, 23%) and the basic case (simultaneous aerobic sludge stabilization followed by land use, 25%). The weighted average from these cases represents the German average. An additional case reflects the best available technology with optimized biogas and incineration processes.

For each step of the urban water chain the external energy flows: electricity from grid, thermal energy from natural gas, fuels for transport are accounted. With the wide system boundaries, this gives a detailed picture of the gross energy consumption associated to the pathway of water through settlements.

While the current system works reliably and fulfills its main functions for public health and protection of water resources, the results show the low metabolic efficiency of the urban water chain today. For C, it averages 15%. Even for facilities employing biogas combustion and sludge incineration, the metabolic efficiency for C is below 25% and with optimized biogas and incineration processes below 40%. Large parts of the C resources are lost for energetic reuse during aerobic treatment. The non reused energetic potential is large compared to brut consumption. In theory, bio-electricity from C resources can fully supply the electricity demand of the urban water chain. While a full energetic reuse may not be technically or economically feasible, the results underline the importance of concepts for increased metabolic efficiency of the urban water chain, in addition to efforts in energy efficiency.

For N and P, the metabolic efficiency of the urban water chain is also low: 20% for P and 4% for N. The concept for reuse of N and P currently employed is the application of sludge generated during wastewater treatment on agricultural land. While the sludge contains considerable amounts of nutrients, especially P, agricultural reuse is decreasing in Germany due to concerns about AMPs. Thus, besides the quantity of recycling flows of N and P, the quality of recycling related to AMP emission is an issue with the current system.

Due to the extent and complexity of the system under analysis, there are important limitations. In this study, three cases were used to represent wastewater and sludge treatment in Germany. In reality, every WWTP is different and there are many particularities in process design and associated energy consumption and generation.

Data for the individual stages of the urban water chain were compiled from statistics and various sources in literature. The most recent data available was compiled, to reflect the present situation as accurately as possible, but often data refers to different years. The analysis also has to rely on assumptions and values from other theoretical studies for some stages of the urban water chain, especially for sludge handling, as data availability from official sources is low. A detailed account is found in the methods section.

An SFA was used to assess flows of CNP within the system and their fate. The SFA method is inherently subject to uncertainties. Influent loads and partitioning factors are average empirical values, which are subject to large variations in reality. Partitioning factors can only reflect tendencies of elemental behavior within a complex system. In combination with the TEP factors for CNP, the SFA also shows the internal energy flows and the metabolic efficiency. The theoretical energy potentials of CNP derived in this study reflect the energetic value of resources. They mark the upper limit of energy harvesting from CNP, constrained only by resource characteristics. They provide no information about the technical feasibility of increased energy harvesting from flow stream resources (technical potential), and the related costs (economic potential), which are reserved for future studies. Despite the limitations, the applied methodology provided a holistic picture of the status quo of the urban water chain in Germany.

*Results: Algae systems for increased metabolic efficiency*

Based on the status quo as baseline for comparison, a concept for increased reuse of CNP is assessed in the second part of this study: the integration of algae systems at wastewater treatment plants (WWTPs). A technical setup to integrate algae systems in the existing treatment steps of WWTPs is proposed. It relies solely on the resources available on site, with no external input of fertilizer, water or $CO_2$ required. The whole algae process chain, from cultivation to production of bio-electricity, takes place on site of the WWTP. While algae systems receive much attention in literature, this is the first detailed study of the integrated system, including an SFA of the major nutrients CNP (see also Menger-Krug *et al.* 2012). Based on this SFA, the implications for energy and emission balance of the WWTP are elaborated. A scenario analysis highlights the key factors for the performance of algae systems at WWTPs.

The results show that the metabolic efficiency is considerably improved by algae systems. It is feasible from a flow stream perspective to produce enough bio-electricity from algae systems, to run the WWTP energy-neutral during the vegetation season or even turn them into net energy producers. This requires ~6 $m^2$ area per person served by the WWTP. It can be achieved with nutrients from wastewater, without any external resource input. C *resp.* $CO_2$ availability is the limiting factor for yield with the proposed process design *i.e.* in absence of external $CO_2$ sources.

While intensive nutrient recycling in algae systems considerably improves the energy balance, it also impacts on effluent quality. While limit values for C (usually measured as chemical oxygen demand COD), N and P are met, the load in effluent increases, mainly *via* the contribution of non-harvested biomass. The harvesting efficiency is identified as a technical key parameter at the crossroads of energy balance and effluent quality. Post treatment is highlighted as an opportunity to reliably meet effluent limit values. Additionally, post treatment also improves the effluent quality in terms of AMPs.

Due to the prospective nature of the system under analysis, there is no empirical data for many key parameters, such as nutrient uptake efficiencies, areal productivity, harvesting efficiency and anaerobic digestibility. Instead, the analysis had to rely on data from pilot applications and laboratory studies, which remain to be confirmed or rejected in practice. Ranges of values from literature were used in a scenario analysis highlighting the key factors for the performance of algae systems at WWTPs. The influence of algae systems on the energy demand of other processes at the WWTP was assessed based on SFA results and the validity of the applied proxies remains to be proven in practice.

While this study has shown the feasibility of the concept from a flow stream perspective, many other aspects require analysis on the way to implementation. This includes acceptance and social aspects, as well as political and economical aspects. For the latter, the future developments of energy costs - for fossil and renewable energy which again depend on the political framework - are important aspects to consider.

*Results: Connecting the urban water chain to urban metabolism*

To put the potential improvement by integration of algae systems in context of the urban metabolism, the scope of the study is extended in the third part. The extended perspective includes the flows that represent the connection points between urban water chain and the full urban metabolism: the daily household consumption of water, energy, food and detergents. The associated flows of CNP are quantified for a semi hypothetical model city.

The study traces the pathways of CNP: the input, the transformations during human metabolism, the transfer to wastewater and organic waste infrastructures and the fate during the treatment processes – with and without algae. While information on the bulk flows is available from official sources and statistics, this is the first study to quantify household consumption in Germany on the level of CNP flows, albeit only for a semi hypothetical model city with high uncertainties. While far from a complete analysis of urban metabolism, the extended perspective is useful to understand the relevance of the urban water chain –with and without algae systems - in context of urban metabolism.

The analysis showed that clean cycles in algae systems can contribute to a sustainable urban metabolism in three important aspects. Firstly, in context of the urban energy balance, the consequent energetic reuse of resources from wastewater in algae systems can considerably contribute to electricity production from renewable sources on an urban scale. Bio-electricity is an important pillar for sustainable energy systems as it covers base loads. The net consumption of the urban water chain on a per person base is rather low compared to total electricity consumption in households. On the other hand, the energy consumption of the urban water infrastructures show a high spatial concentration compared to household energy consumption. The spatial concentration of energy flows and the potential contribution to bio-electricity production make the urban water chain with algae systems an important player for the transition towards a sustainable urban energy system.

Secondly, in context of the urban nutrient flows, the recycling of the resources of N and P in wastewater is highly relevant for a sustainable urban metabolism. These flows represent a large share of the urban nutrient flows and are also important with a wider perspective on food supply. But as the urban water chain is also a major pathway of AMPs, clean cycles are required. Urban flows of AMPs are the third important aspect to consider. The reduction of AMP emissions from the urban water chain is highly relevant for a sustainable urban metabolism. To discuss the problem of AMPs, a model substance is chosen for this study: PFOS.

Including this notorious AMP into the analysis of CNP recycling, serves as a starting point for discussion. But for a full picture many different AMPs with different use patterns, biochemical characteristics and toxicological end points – as well as the effect of mixtures - need to be included. But the results of this study highlight two important aspects. Firstly, algae systems can provide bio-electricity without emission of AMPs during biomass cultivation. Thus, they can be characterized as a clean cycles, in contrast to „open" application of sludge in agriculture. Secondly, processes during algae growth can increase the elimination of AMPs from effluent. The potential of algae systems to reduce the load of AMPs to the environment even makes them cleaning cycles: they reduce the emissions of AMPs during recycling. In that sense, the area designated to algae systems has a double function: to produce biomass for electricity generation – with a higher output per unit area than other energy crops - and to allow for a long hydraulic retention time for AMP elimination from wastewater. If the increased elimination of AMPs works reliably under operating conditions, this provides a strong additional incentive for WWTPs to integrate of algae systems. Besides the fate of AMPs in algae systems, there are other important research needs including the possible formation of algae toxins during biomass growth.

*Conclusion*

In short, algae-to-energy systems can provide a double benefit: more bioenergy from otherwise wasted resources and lower emissions of AMPs to the environment. This characterizes algae systems as clean cycles or even cleaning cycles. On the downside, algae systems have a large area demand and increase the load of CNP in effluent, while meeting limit values.

The integration of algae systems has positive effects on the urban energy balance, the urban flows of nutrients and the urban flows of AMPs. With the combination of these effects, algae systems integrated in the urban water chain can contribute to a sustainable urban metabolism. All three topics: sustainable energy systems and bio-electricity, the cycle oriented management of flow streams including nutrients, and the chemical pollution problem, are important today and their importance will likely further increase in the future.

For algae-to-energy systems integrated in the urban water chain, the results of this analysis warrant further research on the scale of pilot applications. The developed model of substance and energy flows of the integrated system provided information on energy flows, on nutrient recycling within the system, on loads to the individual treatment steps and on loads to the environment. This information is useful to design pilot projects. Data gathered from pilot projects can in turn refine the model. In that sense, the model presented in this study can be used as a tool for system design and optimization.

For this study, a framework for analysis of water infrastructures was developed: the extended energy balance. It proofed a useful tool to analyze reuse oriented concepts for urban water infrastructures, as it assesses the internal energy flows and the metabolic efficiency in addition to the external energy flows. As required by the clean cycle approach, it includes an emission balance covering all environmental compartments. Independent of the case study on algae-to-energy systems in this study, this framework for analysis of water infrastructures can be useful for evaluation of other reuse oriented concepts.

# Content

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# Abbreviations

| | |
|---|---|
| ABM: | Algae biomass |
| AD: | Anaerobic digestion |
| AMP: | Anthropogenic micropollutants |
| AS: | Activated sludge |
| ATT: | Arbeitsgemeinschaft Trinkwassertalsperren e. V.; Association of Drinking Water from Reservoirs |
| ATP: | Adenosintriphosphate |
| BAT: | Best available technology |
| BOD: | Biological oxygen demand |
| BWT: | Biological wastewater treatment |
| C: | Carbon |
| CNP: | Carbon, nitrogen and phosphorus |
| CHP: | Combined heat and power |
| COD: | Chemical oxygen demand |
| DOC: | Dissolved organic carbon |
| DNA: | Desoxyribonucleic acid |
| EROI: | Energy return on investment |
| EU: | European Union |
| dw: | dry weight |
| GHG: | greenhouse gas |
| GWP: | Global warming potential |
| H: | Hydrogen |
| HH: | household |
| HRT: | Hydraulic retention time |
| HRAP: | High rate algae pond |
| IPCC: | Intergovernal Panel on Climate Change |
| JRC: | Joint Research Centre of the European Commission |
| MC: | Model city |
| MFA: | Material Flow Analysis |
| N: | Nitrogen |
| O: | Oxygen |
| OW: | Organic waste |
| P: | Phosphorus |
| PF: | Partitioning factor |
| PBT: | Persistent, bioaccumulative and toxic |
| PFOS: | Perfluoro octane sulfonate |

| | |
|---|---|
| POP: | Persistent organic pollutants |
| PS: | Primary sludge |
| PTW: | Primary treated wastewater |
| REACH: | Registration, Evaluation, Authorisation and Restriction of Chemicals |
| RNA: | Ribonucleic acid |
| SCST: | Sanitation concepts for separate treatment |
| SFA: | Substance Flow Analysis |
| SRT: | Sludge retention time |
| SS: | Secondary sludge |
| TEP: | Theoretical energy potential |
| UM: | Urban metabolism |
| UWC: | Urban water chain |
| UWIS: | Urban water infrastructures |
| vPBvT: | Very persistent, bioaccumulative and very toxic |
| VSS: | Volatile suspended solids |
| WFD: | Water Framework Directive |
| WWTPs: | Wastewater treatment plants |

# 1 Introduction

## 1.1 Motivation

Human activity is concentrated in urban areas representing 3% of land surface, but home to more than half of the world's population, to 75% of natural resources consumption and to 60-80% of GHG emissions (UNEP 2012). There is a compelling need to understand the flows of materials and energy in urban areas (Kaye *et al.* 2005, Grimm *et al.* 2008, Weisz *et al.* 2010), as they represent the physical base for sustainability and resilience of the astysphere (Norra 2009). The term urban metabolism is used to describe the sum of these flows (Wolman 1965). The notion of a metabolism highlights the transformation processes: in the same way as a reactor, a city transforms resource inputs into emissions and wastes. Today, the flow streams are managed mainly in a linear way (Figure 1, top), characterized by large resource inputs: energy, water, food, products and materials; and large output of wastes in gaseous, liquid and solid form. For disposal of wastes an individual settlement depends mainly on its immediate surroundings (Decker *et al.* 2000). Many cities have a halo of pollution around them, with gradients from urban to rural measurable for many substances in different media (Diamond and Hodge 2007). But the impact of a city extends far beyond its immediate surroundings, with a global reach for supply of resources, with imported fuels, food and products and for emissions especially greenhouse gases. In that sense, one may speak of the global hinterlands of cities (Decker *et al.* 2000).

Many environmental problems witness that the capacity of the global hinterlands is reaching its limits. Human activities increasingly influence the global climate (IPCC 2007) and ecosystems (UNEP 2005). The dominant impact of human activities on the Earth System has led to the postulation of a new epoch, the Anthropocene (Crutzen 2002). The planetary boundaries, which define the safe operating space for humanity, are transgressed or close to exceedance (Rockström *et al.* 2009). Transgression may trigger non-linear, abrupt environmental change. The nine planetary boundaries include the cycling of the major elements of the biosphere, Nitrogen (N), Carbon (C)[1] and Phosphorus (P) and the chemical pollution.

The chemical pollution problem is defined as planetary boundary. But it cannot be quantified due to large uncertainties and the multitude of substances involved (Rockström *et al.* 2009).

[1] Climate change as planetary boundary with $CO_2$ concentration in the atmosphere <350 ppm

In the EU, 100 000 different chemicals are used for industrial applications, but also for everyday and household activities (Schluep *et al.* 2006). Substances for household applications include flame retardants, impregnation agents, biocides and pharmaceuticals. Some of them have dangerous properties such as persistent, bioaccumulative and toxic, or endocrine disrupting substances. In this study, these substances are referred to as anthropogenic micropollutants (AMP) (see chapter 1.3.3).

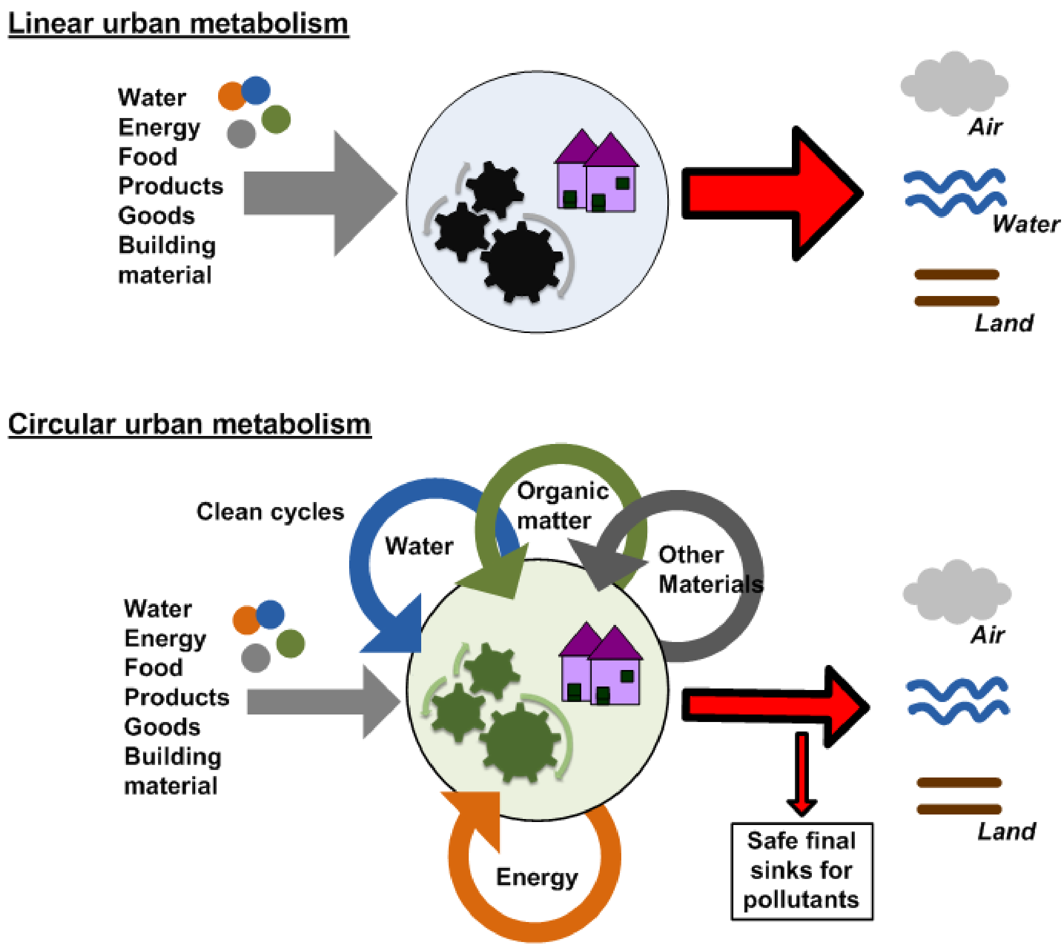


Figure 1-1: The urban metabolism as linear and circular system

Legend: Conceptualization of urban metabolism highlighting the transformation processes turning resources into wastes. (Top) Present form of (mainly) linear urban metabolism (UM) with large volumes of resources consumed and wasted emitted. (Bottom) Ideal form of circular urban metabolism with clean cycles for water, organic matter, materials and energy which decrease the amount of resources consumed and wastes emitted. Anthropogenic micro pollutants AMP which cannot be degraded are disposed of in safe final sinks. Adapted and extended from Rogers *et al.*, 1997, Kral, Kellner and Brunner 2012, Girardet 2010.

Opposed to the prevailing linear system, the ideal management of flow streams mirrors natural ecosystems, using material and energy in cascades thereby decreasing resource requirements and emissions (Girardet 2004 and 2010, Kral, Kellner and Brunner 2012). Reuse of material and energy flows before they exit the astysphere increases the metabolic efficiency i.e. the degree of circularity of material and energy flows (Browne *et al.* 2009).

While a high metabolic efficiency is desirable, a purely quantitative approach to recycling is not sufficient. The quality of recycling regarding the fate of AMPs needs to be considered. Due to the concentrated resource consumption and the accumulation in urban stock, urban areas are hot spots for AMPs. With AMP present in urban areas, *clean cycles* are required for a circular metabolism (Kral, Kellner and Brunner 2012).

As most of the material and energy flows of urban areas are managed by infrastructures, this raises the question, how these systems can be reengineered towards clean cycles. The focus of this study is the urban water infrastructures in Germany and their potential contribution to a more circular urban metabolism (UM).

The urban water infrastructures mediate the pathway of water through settlements: from sourcing of water to treatment and provision of tap water for water use in households, to transport and treatment of wastewater and sludge. This urban water chain (UWC) is an important part of the urban metabolism, mainly for three reasons.

Firstly, it has an indisposable essential function for hygiene and public health. Dense human settlements require some form of management for provision of drinking water and disposal of feces.

Secondly, the urban water chain is a hot spot for anthropogenic emissions. The flow streams on the post use side contain large amounts of the nutrients carbon (C), nitrogen (N) and phosphorus (P). CNP mainly originate from human consumption of food but also from detergents and other products. They act as pollutants if misplaced in the ecosphere (misplaced resources). The flow streams also contain AMPs originating from household use of products or from run off from urban surfaces collected in mixed sewer systems. This makes the urban water chain an important pathway for many AMPs from urban areas[2]. Due to their persistence, AMPs are not (fully) degraded with the current treatment technologies. They remain in

2 Urban wastewater has been described as a major source for many AMPs; compare Schluep *et al.* 2006, Diamond and Hodge 2007, Zimmerman *et al.* 2008, Fatta-Kassinos *et al.* 2011, Ferrari *et al.* 2004a+b, Muñoz *et al.* 2009a+b, Bolong *et al.* 2009, Menger-Krug *et al.* 2011, Mathan *et al.*2011

effluent or are transferred to sludge or to air, depending on their biochemical characteristics. Thus, the emissions from the urban water chain mirror the wide application of chemicals in modern society.

Thirdly, urban water infrastructures are large energy consumers, yet harboring large resources for energetic reuse. While the handling of flow streams consumes considerable amounts of energy, CNP in the flow streams have an energetic value (internal resources). Energetic reuse of internal resources can reduce the consumption of external resources. Via anaerobic processes (biogas) or direct incineration, C resources can provide electricity and thermal energy. N and P resources can gain *indirect* energy credits when substituting energy intensive fertilizers in agricultural applications (indirect energetic reuse). The presence of AMP in the flow streams is a challenge for energetic reuse.

Before this background, the integration of algae-to-energy systems in the urban water chain is a promising concept. Algae systems essentially represent a *closed system* for recycling of nutrients from wastewater. The elements carbon (C), nitrogen (N) and phosphorus (P) are the main ingredients of wastewater, and the key nutrients for algae growth. Algae can be cultivated in flat ponds or closed photo bioreactors (Wijffels and Barbosa 2010). With $CO_2$ addition algae systems can reach high areal productivity. Wastewater can be used as growth medium and the biomass can be harvested for bioenergy generation (Lundquist *et al.,* 2010, Park and Craggs, 2011). Furthermore, algae remove nutrients from wastewater during growth, thus gaining energy credits for (partial) wastewater treatment (Sturm and Lamer 2010, Campbell *et al.,* 2011; Lundquist *et al.,* 2010, Colosi and Clarens 2009, Clarens *et al.* 2012).

The closed system is a large advantage compared to open agricultural applications, as no AMPs are transferred from wastewater to the environment during cultivation. Furthermore, algae systems can also improve the effluent quality by removing AMPs from wastewater and accumulating them in biomass.

Thus, algae-to-energy systems can provide synergies that can contribute to a sustainable metabolism: produce bioenergy from otherwise wasted resources and lower the emissions of AMPs to the environment. This study tries to quantify these effects and to assess their relevance in context of the urban metabolism.

## 1.2 Aim and structure of study

The aim of this study is to analyze the integration of algae systems in the urban water chain in terms of energy balance and emission balance, compared to status quo, and to assess the relevance of the urban water chain – with and without algae systems - in context of the urban metabolism (Figure 1-2).

This study has three parts. It starts with a detailed description of the status quo of the urban water chain in Germany. The focus of the analysis is the metabolic efficiency i.e. the degree of reuse of resources CNP. Before the background of the current situation, a concept for increased metabolic efficiency is evaluated in the second part: the integration of algae systems at wastewater treatment plants (WWTPs). In the third part: Connecting the urban water chain to urban metabolism, the scope of the analysis is extended to assess the relevance of the urban water chain - with and without algae systems - in context of the urban metabolism. Three important aspects are considered: the role of the urban water chain in context of the urban energy flows, the urban nutrient flows and the urban flows of anthropogenic micropollutants (AMPs).

Methodologically, a systems analysis is employed combining a conventional energy balance with a substance flow analysis (SFA) and the assessment of the energetic value of the resources. The underlying model includes three layers. The technical setup builds the foundation of the model (layer 1). Different technical setups (cases) are defined to describe the status quo of the urban water chain, including the water supply, the wastewater and the sludge management; and the integration of algae systems. For each case, the gross consumption for handling and treatment of flow streams is compiled: electricity, thermal energy and fuel consumption. Together with the own generation of energy from biogas use or sludge incineration, these energy flows represent the external energy flows of the system (layer 2).

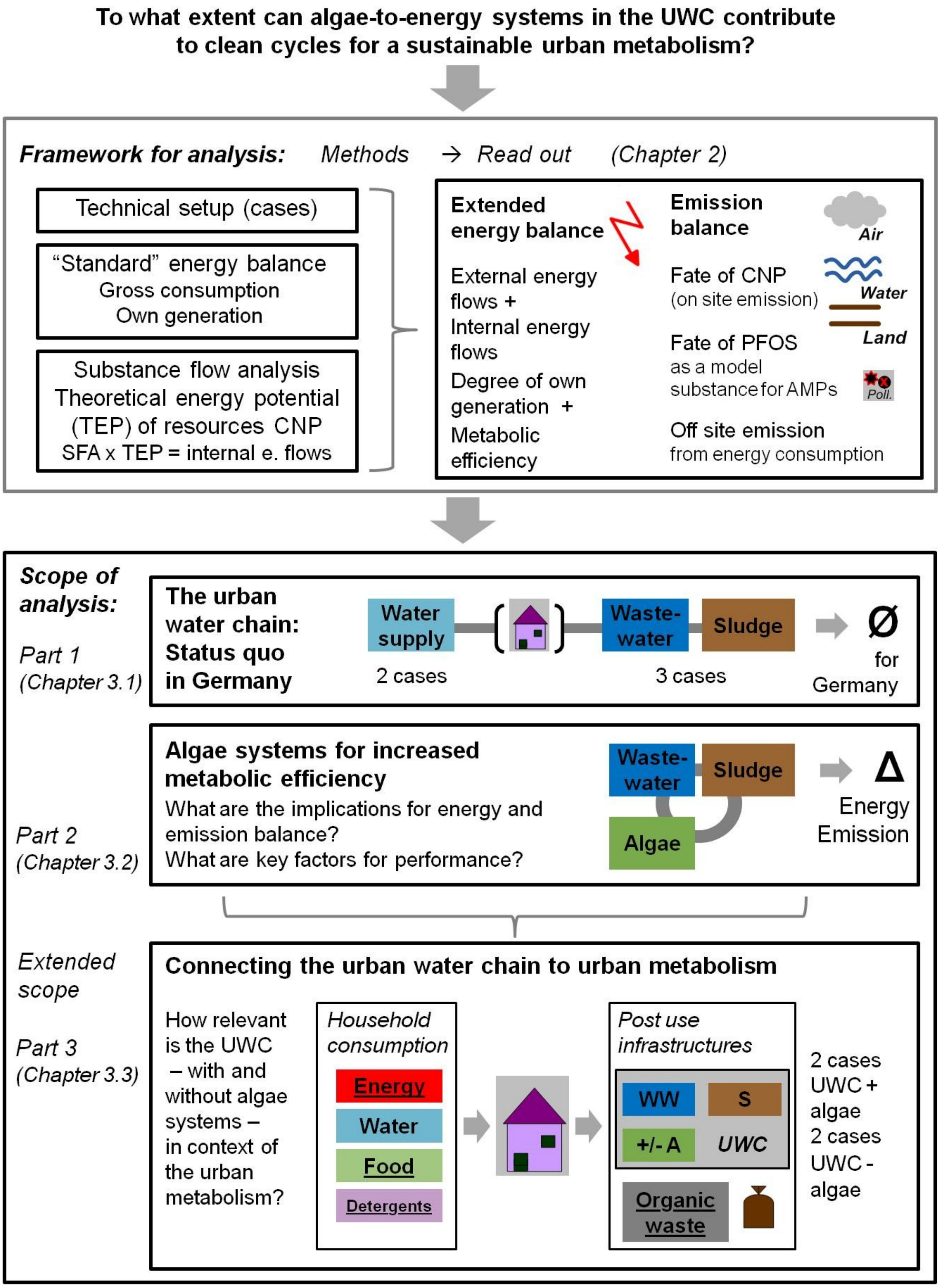


Figure 1-2: Structure of study

To extend the usual approach to energy balances, the analysis also assesses the energetic value of the resources *in* the flow streams, termed internal energy flows. The theoretical energy potential (TEP) is assigned to the resources CNP. It relates to the chemical energy of C and the “grey” energy of the nutrients N and P. By applying the respective TEP factors [kWh/kg] to the SFA, the internal energy flows associated to CNP are traced along the urban water chain. The metabolic efficiency for N and P relates to the amount applied to agricultural land corrected for plant availability. C is energetically reused via biogas use or sludge incineration for electricity and heat co generation, as quantified in layer 2 of the model. Putting this energy generation in relation to the TEP of C gives the metabolic efficiency. This extended energy balance gives a holistic picture of the energy flows along the urban water chain and allows assessing the metabolic efficiency: the degree of reuse in relation to the full energetic value of the resources. The metabolic efficiency is a useful measure to describe the degree of circularity within the system.

The clean cycle approach requires assessing not only the quantity, but also the quality of CNP recycling. Therefore, the extended energy balance is accompanied by the emission balance of the urban water chain. The emission balance includes the emissions of CNP from the flow streams (on site emissions) to all environmental compartments based on SFA results. While the effluent quality is the traditional focus of the water sector, a systemic view allows assessing additional indicators. This includes the C efficiency of bio-electricity generation and the comparison of on site $CO_2$ emissions and off site $CO_2$ emissions related to the energy consumption of external energy. In addition, a model substance (perfluoro octane sulfonate PFOS) is included in this study to discuss the problem of anthropogenic micropollutants (AMPs).

The strength of the developed framework lies in three main aspects. Firstly, this study is unique in focusing on the material flows on elemental level, tracing the flows of the major elements of the biosphere CNP, which are also the main constituents of wastewater, along the full urban water chain. The emission balance includes a systemic perspective on all compartments. Secondly, the TEP factors applied to the SFA allow inclusion of the energetic value of resources and the metabolic efficiency. This extends the usual approach to the analysis of energy flows. The extended energy balance developed in this study, captures both: the energy required for handling of flow streams (external energy flows) as well as the energetic value of the resources CNP (internal energy flows). The framework captures the double role of CNP: they are resources with an energetic value but act as pollutants when misplaced to the environment. Thirdly, the problem of AMPs is taken into account, albeit only exemplified by one model substance.

In the first part of the study, the status quo of the urban water chain in Germany is assessed as baseline for comparison (chapter 3.1). While the focus lies on the post use side of the urban water chain, the scope of the study includes the full pathway of water through settlements: from sourcing of water to treatment and provision of tap water for water use in households, to transport and treatment of wastewater and sludge. To capture the current situation in Germany, two the technical setups for water supply: from groundwater and from surface water, and three technical setups for wastewater and sludge disposal are defined. The weighted average from these cases represents the German average.

Based on the status quo as baseline for comparison, a concept for increased reuse of CNP is assessed in chapter 3.2: the integration of algae systems at wastewater treatment plants (WWTPs). A technical setup is proposed in this study. It relies solely on the resources available on site, with no external input of fertilizer, water or $CO_2$ required. This is the first study to assess the implications for the energy and emission balance arising from integration of algae systems at WWTPs. The applied method with the detailed SFA model allows assessing the impacts of algae systems on the internal cycling of CNP and thus on the energy balance. To assess also the quality of recycling, the implications for the emission balance are taken into account.

Due to the prospective nature of the system under analysis, there is no empirical data for many key parameters, such as nutrient uptake efficiencies, areal productivity, harvesting efficiency and anaerobic digestibility. Instead, data from pilot applications and laboratory studies are used. A scenario analysis is performed to show the influence of variations in key factors on the energy and emission balance of algae systems at WWTPs.

To evaluate the potential improvement in context of the urban metabolism, the scope of the study is extended (chapter 3.3). The extended perspective includes the flows that represent the connection points between urban water chain and the full urban metabolism: the daily household consumption of water, energy, food and detergents. The associated flows of CNP are quantified for a semi hypothetical model city. The flows and fate of PFOS is included to discuss the problem of anthropogenic micropollutants (AMPs). While far from a complete analysis of urban metabolism, the extended perspective is useful to understand the relevance of the urban water chain –with and without algae systems - in context of urban metabolism.

## 1.3 Background

This background chapter describes the current state of the urban water chain in Germany and approaches for increased sustainability. The double role of carbon (C), nitrogen (N) and phosphorus (P) as potential resources and pollutants is discussed. The problem of anthropogenic micropollutants (AMPs) is reviewed. Then, algae systems for CNP recycling are introduced. The background chapter closes with an outline of the concept of urban metabolism.

### *1.3.1 The urban water chain in Germany*

The urban water chain is an essential part of urban metabolism due to its function for hygiene and public health. The urban water chain includes the drinking water supply, the wastewater and sludge disposal. In Germany, 99% of the population or 82 million people are connected to the drinking water supply infrastructures. Drinking water is mostly sourced from groundwater with excellent quality (70%, ATT *et al.* 2011). Average water use in households comprises 110 l/p*d. There is a large variation in water use in households between different regions in Germany (Schleich and Hillenbrand 2008). The typical German household uses about 39% for personal hygiene (bathing, showering etc.), 30% for flushing toilets, 13% for laundry, 7% for dishwashing, 7% for room cleaning, washing cars and gardening and 4% for cooking and drinking (UBA 2007).

Used water is collected in the wastewater infrastructure. Today, 95% of the population is connected to the wastewater infrastructures. Average amount of wastewater generated per person is 250 l/d or 91 $m^3$/a (Haberkern *et al.* 2008). There is a large variation in wastewater generation between different regions in Germany, ranging from 140 to 310 l/p*d *resp.* 50-113 $m^3$/p*a (DWA 2011). Variations are due to user behavior and contribution of rainwater and extraneous water (net infiltration to sewer system).

Besides water as transport medium, household wastewater contains considerable loads of CNP, originating mainly from feces and urine (ultimately from foodstuff) and from cleansing products for body and household. While CNP are resources with an energetic value, they act as pollutants when emitted to the environment. The effluent quality is the traditional focus of the sector and protection of water resources from eutrophication is a major goal. For the effluent, limit values apply.

Different treatment steps are employed at wastewater treatment plants (WWTPs) to decrease the load of CNP in effluent and to comply with limit values.

The process typically employed is biological wastewater treatment (BWT), also called activated sludge treatment (AS). It requires intensive aeration and produces sludge that requires further treatment. The effluent is discharged, typically to rivers. The total elimination efficiencies from effluent average 95% for C, 81% for N and 91% for P, according to benchmarking studies at German WWTPs (DWA 2011, see also Table 2-9).

The sludge generated during biological wastewater treatment is stabilized, either with aerobic or with anaerobic processes. While the aerobic treatment requires energy intensive aeration, the anaerobic treatment produces biogas that can be used for electricity generation. While the anaerobic treatment is favorable from an energy perspective, it requires large mass flows to amortize the investment in digester and generator in due time (economies of scale). Thus, while technically feasible also for small WWTPs, it is rarely employed by smaller WWTP with less than 20 000 p.e.[3] (Haberkern *et al.* 2008).

In Germany, 75% of the population is served by WWTPs employing anaerobic sludge stabilization, the remaining 25% by WWTPs with aerobic sludge stabilization (ATT *et al.* 2011, Haberkern *et al.* 2008). The latter group typically represents small plants.

After stabilization, sludge is dewatered to a typical solids content of 20-30%. Dewatering releases sludge water (process water), which is typically rich in nutrients. The sludge water treatment contributes considerably to energy consumption of wastewater treatment. The processes for wastewater treatment and the underlying flows of CNP are described in the following chapter (chapter 1.3.2).

A holistic perspective of the urban water chain needs to include the end use of sludge. Sludge generated during wastewater treatment amounts to ~2 million tons dry weight (dw) per year in Germany. 52.5% of sludge is incinerated, 30% used for agriculture and 15% for landscaping. Land filling of sludge is prohibited (ATT *et al.* 2011, UBA 2012). Both land use and incineration have advantages and disadvantages. During sludge incineration, electricity can be produced, but the demand for thermal energy for the drying of the substrate can scavenge the energy gains. Furthermore, a sludge incineration facility requires large mass flows to produce enough waste heat for drying of the sludge before incineration. It is favorable to co incinerate sludge with coal in large power plants, providing high efficiencies for electricity generation

[3] The size of WWTPs is expressed in population equivalents (p.e.) (Haberkern *et al.* 2008). One p.e. refers to an certain load of C in wastewater that equals the average C load from one person. Wastewater from commercial operation and indirect dischargers are thus fitted to “person equivalents (p.e.)” via the C content.

and large amounts of waste heat. But capacities are limited due to transport distances and concerns about the quality of the (mixed) ashes and slag, which are usually used for cement production (Haberkern *et al.* 2008, UBA 2012)

Due to the large scale of incineration facilities, the transport distances from the WWTP can be very far with high associated costs, energy consumption and $CO_2$ emissions. Also for large facilities, the costs are considerably higher than for land application of sludge. The combustion gas from sludge incineration requires treatment, again consuming energy. Despite the treatment, the combustion gas from sludge incineration can contain anthropogenic micropollutants (AMPs). While organic based AMPs are degraded during incineration, other AMPs like mercury (Hg) and dioxins[4] can be transferred to combustion gas. Ashes contain many inorganic AMPs and need to be deposited[5]. On the other hand side, if sludge is used in agriculture, these AMPs are transferred to soils. For sludge containing AMPs, incineration reduces the emission of AMPs to the environment compared to the alternative.

The current system, as described above, fulfills the primary functions related to hygiene and public health and protection of receiving waters from eutrophication very reliably in Germany. But there are fundamental critics related to the sustainability, as listed below. This chapter gives a short overview of the critics and the proposed solutions, an extensive review can be found in DWA (2008).

*High water consumption and high quality water supply for all purposes.*

Large amounts of water with drinking water quality are required to transport feces through the sewer system. While water availability is no fundamental concern for Germany with ample water resources in most parts of the country today, many regions of the world employing the same system suffer from water scarcity. The problem may aggravate in the future due to climate change or chemical pollution of water resources by AMPs.

*Mixing different water qualities and high dilutions.*

The different uses of water: for cleaning, for washing and for toilet flushing (transport of feces and urine) generate different qualities of wastewater: with different loads of CNP, patho-

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4 Most organic AMPs (including PFOS) are degraded during incineration. But dioxins are a notable exception. They are produced during incineration of biomass and many other combustion processes if chlorines are present and the temperature meets the „dioxine window (see http://www.umweltbundesamt.de/chemikalien/dioxine.htm)

5 Ashes also contain large amounts of P. There are approaches to extract metals and P from ashes from sludge incineration, but without large-scale implementation (UBA 2012).

gens and AMPs, and with different temperatures. Often, urban run off is also collected in the same sewer systems. Mixing the flow streams and the resulting dilution of resources and pollutants makes reuse of resources, as well as elimination of pollutants difficult. Therefore more differentiated systems with two to four separate flows of wastewater, often in combination with decentralized or semicentralized treatment are advocated to increase sustainability of the urban water chain (see below).

*Quality of effluent and sludge.*

The urban water chain is a hot spot for emission of CNP as well as AMPs to the environment. AMPs from the urban stock are transferred to wastewater. As the technologies currently employed are inefficient for their removal, they are emitted to the environment via effluent and sludge. While the sludge generated during wastewater treatment contains considerable amounts of nutrients, especially P, agricultural reuse is decreasing in Germany due to concerns about chemical pollution of soils (UBA 2012). AMPs in effluent are also a growing concern for WWTPs in Germany. The large-scale implementation of advanced effluent treatment (4$^{th}$ treatment stage), as recently introduced in Switzerland, is discussed. But most technologies for advanced effluent treatment, such as activated carbon treatment or ozonation, have a considerable energy demand, resulting in a trade off between energy balance and effluent quality.

Based on these critical points, different alternative systems have been developed and implemented in Germany and internationally (for an overview, refer to DWA 2008). Their common point is the focus on resource reuse: water, thermal energy, energy from C resources, fertilizer or soil conditioner. This includes projects such as the DEUS project (Hillenbrand 2009), NOVAQUATIS (Larsen und Lienert 2007), KOMPLETT (Hansen *et al.* 2007), SCST (sanitation concepts for separate treatment, Peter-Fröhlich *et al*. 2008), Hamburg Water Cycle (Schonlau *et al*. 2008) and SANIRESCH (Winkler *et al*. 2011).

The concept evaluated in the present study: the integration of algae systems, can add to the toolbox of technologies for an increased sustainability of the urban water chain, as described in the references listed above. The integration of algae systems requires no separation of flow streams for reuse of CNP, but has large area requirements.

### *1.3.2 Carbon, nitrogen and phosphorus: resources and pollutants*

Carbon (C), nitrogen (N) and phosphorus (P) are the major elements of the biosphere and essential nutrients for all organisms (Schlesinger and Bernhardt 2013). CNP are the major constituents of biomass (besides water or Oxygen (O) and Hydrogen (H), *respectively*) and fulfill important functions in all biomolecules. The main “building blocks” of biomass are C atoms. In addition, C is also the “fuel” for the majority of organisms. Autotrophic organisms use solar energy to build reduced C substrates (biomass) and heterotrophic organism gain the energy for metabolism by oxidizing these C substrates. N is an essential constituent of proteins, while P is important for the energy metabolism of cells (adenosin-triphosphate ATP), cell membranes (phospholipids) and the storage and transcription of genetic material (desoxyribonucleic acid DNA and ribonucleic acid RNA).

While CNP are essential nutrients for all organisms they act as pollutants, if misplaced in the environment (misplaced resources). C and especially N and P cause eutrophication of fresh and salt water. Eutrophication is the process of ecological response to the enrichment of growth-limiting nutrients (especially N and P) with increased primary production, decreased biodiversity and subsequent hypoxia of water. Gaseous C species ($CO_2$, $CH_4$) and a variety of N gases contribute to the greenhouse effect. N gases also contribute to tropospheric ozone pollution, or stratospheric ozone destruction (Gruber and Galloway 2008).

As CNP are the major elements of the biosphere, their global biochemical cycles are coupled. All biota need CNP to build their tissues and the specific elemental stoichiometries[6] of CNP in biomass determine the coupling of CNP cycling (Sterner and Elser 2002, Gruber and Galloway 2008). For the urban water chain the coupling is important as the C:N:P ratio is an important factor for the efficiency of the biological wastewater treatment (BWT). While the cycles of CNP are coupled, the biogeochemistry of the elements is profoundly different (Schlesinger and Bernhardt 2013).

Due to human activities, bioavailable N has nearly doubled and bioavailable P tripled in the environment (Howarth and Ramakrishna, 2005). The human impact on the global biochemical cycles of CNP is surmounting, and the planetary boundaries are transgressed for N and C, and close to exceedance for P (Rockström *et al.* 2009). The majority of N and P are used as ferti-

[6] In the ocean, the C/N ratio of the autotrophic phytoplankton responsible for nearly all marine photosynthesis varies remarkably little (Redfield ratio), whereas the C/N ratio of terrestrial plants is substantially more variable and also tends to be larger than that for marine phytoplankton (Sterner and Elser 2002, Gruber and Galloway 2008).

lizer for agricultural production. N fertilizer can be produced from the abundant atmospheric nitrogen gas ($N_2$) through the Haber–Bosch process, which is only limited by the high energetic costs. In contrast, P is a limited mineral resource. P exists in the earth's crust in the form of phosphate rock, and naturally it is mobilized into terrestrial systems only through the slow processes of weathering and leaching. The production of P fertilizer relies on the availability of phosphate rock. The rate of phosphate rock extraction is much higher than the rate at which it is replenished (Smil, 2000). The annual extraction-to-reserve ratio of phosphate rock, known as reserve-life, has been therefore estimated at 50-100 years (Tilman *et al.*, 2001). Therefore, the increase in the anthropogenic mobilization of P has raised concerns on both input (limited resource) and output ends (Smil, 2000; Cordell *et al.,* 2009).

CNP can be regarded as resources with an energetic value. C, in dependency of its oxidation state, is an energy carrier. Biologically, the chemical energy of C fuels the heterotrophic organisms. Technically, the chemical energy of C can be exploited either directly via combustion, which requires a dry substrate, or via anaerobic processes and subsequent biogas combustion. The nutrients N and P have an indirect energetic value. They gain energy credits when substituting energy intensive mineral fertilizers. To recap, the benefits of P recycling go beyond the energy perspective, as P is a limited resource with an essential function for food production.

CNP are also the main constituents of wastewater. CNP enter the urban water chain on level of households. They originate mainly from human consumption of food, but also from detergents and other products used in households. Due to their negative impact on aquatic ecosystems, CNP in effluent are pollutants and limit values for effluent apply in Germany.

Different treatment steps are employed at wastewater treatment plants (WWTPs) to decrease the load of CNP in effluent and to comply with limit values. The processes and the underlying transformations of CNP are shortly reviewed here (for an extensive review refer to DWA 2008; DWA 2007, Bischofsberger *et al.* 2005).

The C load of incoming wastewater, measured as chemical oxygen demand[7] (COD) is removed from effluent typically by activated sludge process (AS). A diverse community of heterotrophic microorganisms metabolizes biodegradable C substrates. For maximized biodegradation, this process requires intensive aeration. The aeration is one of the most energy inten-

[7] Factor from C to COD assumed with 3 (range 2.8-3.2, Henze *et al.* 2000)

sive treatment steps. The microorganisms use C for their energy metabolism (catabolism) and for biomass growth (anabolism), resulting in the production of $CO_2$ and biomass.

The not readily biodegradable C substrates, typically around 5% of the incoming C load, remain in the effluent. This fraction also includes organic AMPs. It is estimated, that 10-50% of the C effluent load can be attributed to AMPs (Schluep *et al.* 2006, see the following chapter 1.3.3).

During the activated sludge process, the biomass forms flocs, which are retained in the system. Retention of biomass flocs decouples the hydraulic retention time from the sludge retention time and allows an effective removal of biodegradable C substrates from wastewater. Biomass is ultimately removed from the activated sludge process as sewage sludge.

N and P are also removed from effluent by the activated sludge process, but their elimination is limited by the availability of C substrates which are required to fuel heterotrophic metabolism. The C:N:P ratio of wastewater is an important parameter for a well functioning activated sludge process. While the ideal ratio is 63:5:1 (ATV 2000, Knerr 2012), the average wastewater is considerably C depleted. With an average load of 120 g COD, 11 g N and 2 g P (DWA 2008, see also chapter 2.2), the C:N:P ratio is 19:6:1.

As N and P removal from wastewater by the activated sludge process is limited by C availability, different processes are available for enhanced nutrient elimination. For N, most of these processes are energy intensive. Simultaneous nitrification-denitrification is often employed. Nitrification is the biological oxidation of nitrogen from ammonia ($NH_4^+$) to nitrate ($NO_3^-$). It is followed by denitrification, the reduction of nitrate to nitrogen gas ($N_2$) under anoxic conditions. Nitrogen gas is released to the atmosphere. The nitrification-denitrification process holds the risk to release also other N species to air, among them $N_2O$, a very potent greenhouse gas (GHG). Also with enhanced N elimination, N cannot be completely removed from effluent. The N load remaining in effluent is in the range of 20% (DWA 2011).

For P, enhanced elimination typically involves precipitation with iron and alum based precipitants. This process is less energy intensive than for N and reaches elimination efficiencies >90%. The plant availability of the precipitated P is low.

The sludge generated during the activated sludge process can be stabilized by anaerobic digestion (AD). The sludge is composed mainly of cells of microorganisms containing nucleic acids, proteins, carbohydrates and lipids, their decay products and non-metabolized organic material, e.g. cellulose (Manara and Zabaniotou 2012). Under anaerobic conditions, a community of microorganisms (acetogens and methanogens) produces biogas, a mixture of $CH_4$

and $CO_2$ from organic substrates. Substrates cannot be completely used. The typical anaerobic digestibilities of sewage sludge range from 35% to 55% (DWA 2007, Haberkern *et al.* 2008). Unused organic substrates, as well as inorganic compounds, remain in stabilized sludge. After digestion, stabilized sludge is dewatered to a typical solids content of 20-30% (DWA 2007, Haberkern *et al.* 2008). Dewatering releases sludge water (process water), which is typically rich in nutrients, especially N. Due to high loads, the sludge water treatment contributes considerably to energy consumption of wastewater treatment.

As a general trend, improved effluent quality in terms of lower loads of CNP comes at cost of increased energy intensity. In parallel to lower loads in effluent, the transfer of CNP to other compartments increases. With higher level of wastewater treatment, C and N are increasingly transferred to air and P to sludge. While the sludge generated during wastewater treatment contains considerable amounts of nutrients, especially P, agricultural reuse is decreasing in Germany due to concerns about AMPs and chemical pollution of soils (UBA 2012). The problem of AMPs is discussed in the following chapter.

### 1.3.3 The problem of anthropogenic micropollutants

While the technologies employed in the current system for wastewater management work reliably and eliminate large parts of carbon (C), nitrogen (N) and phosphorus (P) from effluent, they are ineffective for elimination of many anthropogenic micropollutants (AMPs). Due to the concentrated human activity in urban areas, there are many potential sources for AMP to wastewater.

The pool of potential AMP is large. 100 000 different chemicals are used in the EU, 30 000 of them in amounts larger than 1 ton per year. For the majority of chemicals (90%) the data base for assessment of harmful properties is insufficient (Schluep *et al*. 2006). While the data base for individual substances is low, the knowledge about the concerted chronic effects of chemical cocktails e.g. mixtures of substances is virtually non-existent.

The undefined nature of the chemical pollution problem: neither the substances nor the exact human and eco toxicological effects can be identified, hinders the integration in sustainability assessments. This undefined nature is also reflected in the classification as risk type "Pandora" with high uncertainty related to probability and impact (WBGU 1999). The problem with persistent substances lies in the irreversibility: once persistent substances are emitted to the environment, remediation is virtually impossible. So whatever the effects of these persistent substances on ecosystem and human health, these effects are irreversible. Therefore, Rockström *et al.* (2009) defined the chemical pollution problem as one of the planetary boundaries.

In this study, the term AMP is used to refer to all substances that contribute to the chemical pollution problem defined as one of the planetary boundaries. With this wide definition, AMPs include organic pollutants and also inorganic pollutants e.g. heavy metals. The exact pathways of AMPs through the anthroposphere are complex and largely unknown (Figure 1-3). The emission from urban areas gathers more attention, since many industrial sources have been regulated. The urban water chain is an important pathway for many AMPs from urban areas[8]. The load of AMPs in wastewater mirrors the common use of chemicals in modern society. The exact pathways of AMP transfer to wastewater are largely unknown. In households, AMPs are present in trace amounts in many products, such as impregnated carpets and

[8] Urban wastewater has been described as a major source for many AMPs; compare Schluep *et al.* 2006, Diamond and Hodge 2007, Zimmerman *et al.* 2008, Fatta-Kassinos *et al.* 2011, Ferrari *et al.* 2004a+b, Muñoz *et al.* 2009a+b, Bolong *et al.* 2009, Menger-Krug *et al.* 2011, Mathan *et al.*2011

clothes, electronics, cleaning products for house and body, biocides for house and garden, paints and plastics. Together with the built environment, these products and goods represent the urban stock of AMPs. They may be released by abrasion, by cleaning with water or by out gassing. AMPs may accumulate in household dust. The dust may be transferred to wastewater during cleaning. Run off from urban surfaces collected in mixed sewer systems[9] also contribute to the AMP load of the urban water chain. This includes traffic related AMP emissions, such as heavy metals from tire abrasion, as well as AMPs deposited from the air (dust, wet and dry deposition, long range transport) or AMPs leaching from surfaces and litter, or from secondary sources such as urban soils and sediments.

AMPs in wastewater are hardly degraded with the current treatment technologies. They remain in effluent or are transferred to sludge or to air, depending on their biochemical characteristics. The wide range of AMPs persisting in effluent after conventional treatment includes inorganic compounds, heavy metals, persistent organic pollutants like endocrine disrupting compounds, pharmaceutically active compounds, disinfection by-products, and many other complex compounds (Fatta-Kassinos *et al.* 2011). It is estimated, that 10-50% of the C effluent load can be attributed to AMPs with 100 to more than 1000 different AMPs in concentrations in the ng to µg range (Schluep *et al.* 2006). Thus, the chemical “cocktail” problem is of special concern for the receiving waters.

While the AMP problem is of high importance for the urban water chain in context of urban metabolism, the complexity of the issue with potentially more than 1000 different substances involved hinders the integration into sustainability assessments. In this study, one substance is included which is a prime example to illustrate the chemical pollution problem: perfluorooctanesulfonate (PFOS). Including this notorious AMP into the analysis of CNP recycling serves as a starting point to discuss the AMP problem; acknowledging that for a full picture many more AMPs with different use patterns, biochemical characteristics and toxicological end points – as well as the effect of mixtures - need to be included

[9] 60% of sewer system in Germany is mixed i.e. receives wastewater from households and commerce as well as run off from urban surfaces (ATT *et al.* 2011).

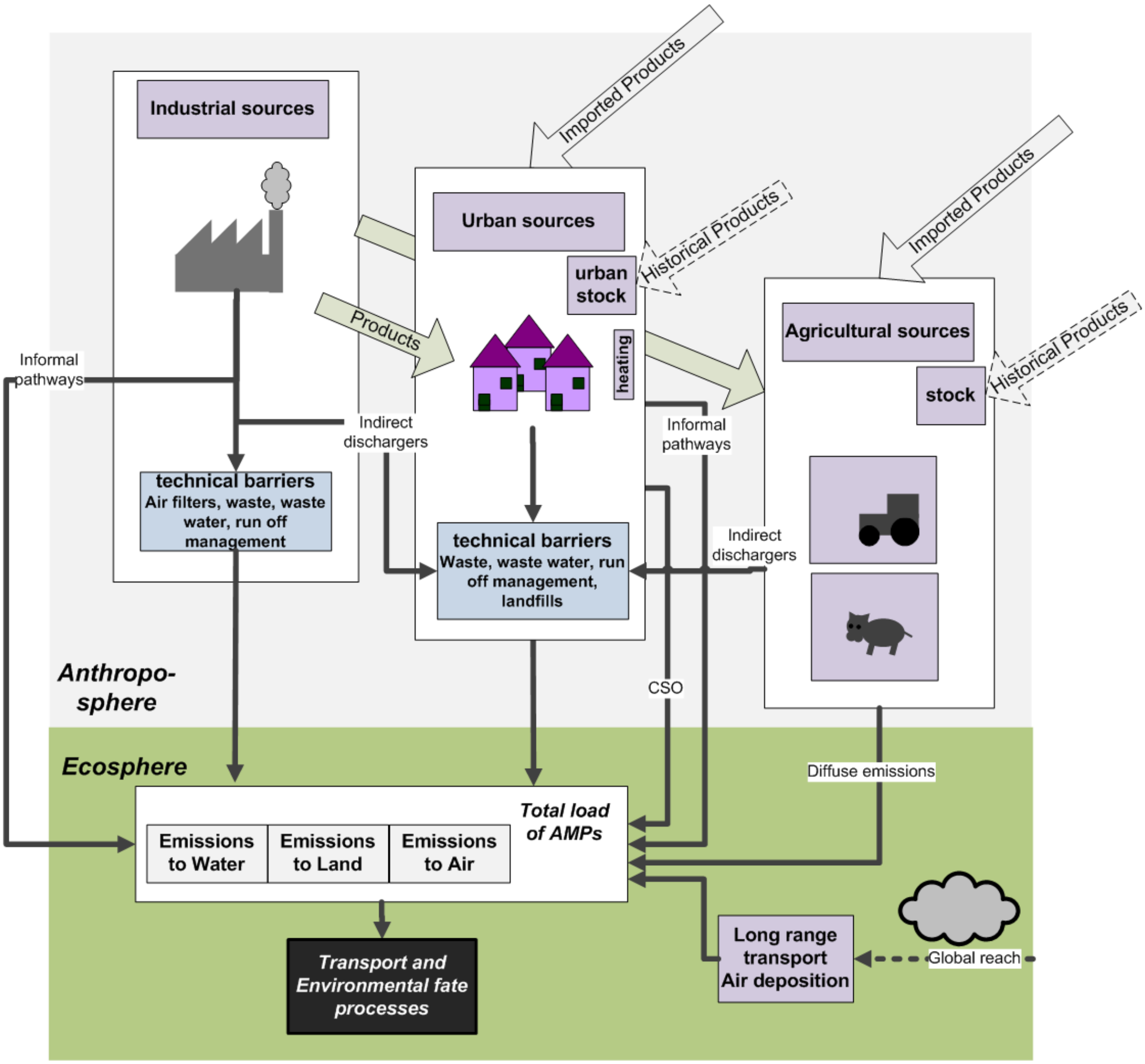


Figure 1-3: Overview of pathways for anthropogenic micropollutants

Legend: Sources of AMPs include industrial sources, urban sources and agricultural sources. The urban stock also contributes to the flow of AMPs. Besides the infrastructure systems (termed technical barriers), which also include the urban water chain, also informal pathways can be important. They include accidental or criminal discharge, littering or wrongly connected sewer pipes. While the focus of this study is the urban water chain, for a holistic strategy for AMPs, all pathways have to be included.

In this study PFOS is used as a model substance to discuss the problem of AMP in context of the urban water chain and the urban metabolism. PFOS is a perfluorinated substance, which is exclusively of anthropogenic origin and not formed in nature (UBA 2007). It has been used in a range of industrial and consumer applications and products since the 1950s. It is a surface-active substance repelling grease, dirt, as well as water. It is stable in industrial processes even under harsh conditions.

PFOS keeps its unique properties that make it valuable for industrial and consumer applications also after emission to the environment. There are no known degradation mechanisms under environmentally relevant conditions (UBA 2007, Buser and Morf 2009). During 60 years of use, PFOS has achieved a worldwide distribution, even in remote areas like the arctic, as many studies have reported. They are found in wildlife such as fish, birds and marine mammals, as well as in human blood samples (Buser and Morf 2009).

PFOS is classified as vPBvT-substance (very persistent, bioaccumulative and very toxic) under REACH[10] and is also included in the list of persistent organic pollutants (POPs) under the Stockholm Convention. PFOS is banned in the European Union (EU) and most industrial uses are in the process of phasing out. But PFOS was also widely used in household products including impregnated carpets, leather/apparel, textiles/upholstery, paper and packaging and household cleaning products. For example, carpets manufactured before the ban can contain large amounts of PFOS as impregnation agent. Despite the ban, PFOS is still emitted from long lived products in the so called urban stock.

In an EU wide survey of rivers, the Joint Research Centre of the European Commission (JRC) detected PFOS in more than 95% of samples, underlining the ubiquitous distribution of this AMP. A related study found a load of 27 µg/day (10 mg/year) in EU rivers, amounting to ~5 t/a for the EU[11]. The contribution of wastewater to the load in rivers is unknown, but studies in Switzerland and Germany found that wastewater treatment plants (WWTPs) are the major source (Huset *et al*. 2008 and Becker *et al.*2008).

[10] REACH: Registration, Evaluation, Authorisation and Restriction of Chemicals

[11] EU population in 2008 approximately 497.5 million, Lanzieri (2008)

### 1.3.4 Microalgae systems

Microalgae as a source of bioenergy have evoked interest in the economic and scientific community, due to their potential high energy yields (Wijffels and Barbosa 2010). "Microalgae" is a generic term to describe aquatic photosynthetic organisms of different families and species. It includes prokaryotic cyanobacteria and eukaryotes green algae (Lundquist 2010). Microalgae can be produced in flat ponds or reactors and do not require arable land.

There are no large-scale applications for energy production yet, but some species are commercially produced for nutritional supplement such as *Chlorella* and *Spirulina*, or for high value constituents such as *Dunaliella salina* for beta-carotene and *Haematococcus pluvialis* for astaxanthin (Lundquist 2010).

Compared to other energy crops, microalgae have a higher energy output per unit area and do not require arable land (Miller 2011). But they have high upstream burdens for water, nutrient and $CO_2$ provision. Therefore, the sustainability of microalgae systems has been questioned in recent LCA-based studies (Murphy and Allen 2011, Colosi and Clarens 2010). Integrating microalgae systems and wastewater treatment is often recommended for improved sustainability.

The idea of integrating microalgae systems and wastewater treatment dates back to the 1950s (Oswald *et al.* 1953, Oswald and Golueke 1960) and offers many potential synergies. In theory, all resources that are needed for algae growth are available at WWTPs (see Figure 1-4). Wastewater provides a growth medium rich in macro and micro nutrients (US DOE 2010, Sturm and Lamer 2010, Pittmann 2010, Rawat 2010, Christensen 2011), $CO_2$ can be supplied from flue gas on site (Lundquist *et al.,* 2010; Kadam *et al.*, 2002). Algae systems at WWTPs receive water, nutrients and $CO_2$, with no upstream burdens and no competing uses. Another synergy is the energy offset from (partial) wastewater treatment, as algae remove nutrients from wastewater during growth (US DOE 2010, Sturm 2010, Pittmann 2010, Rawat 2010, Christensen 2011). Harvested biomass can be used energetically for production of biofuels, or for electricity generation via biogas or direct combustion (Sturm *et al.* 2011, Sialve *et al.,* 2009, Lundquist *et al.,* 2010, Colosi and Clarens 2010).

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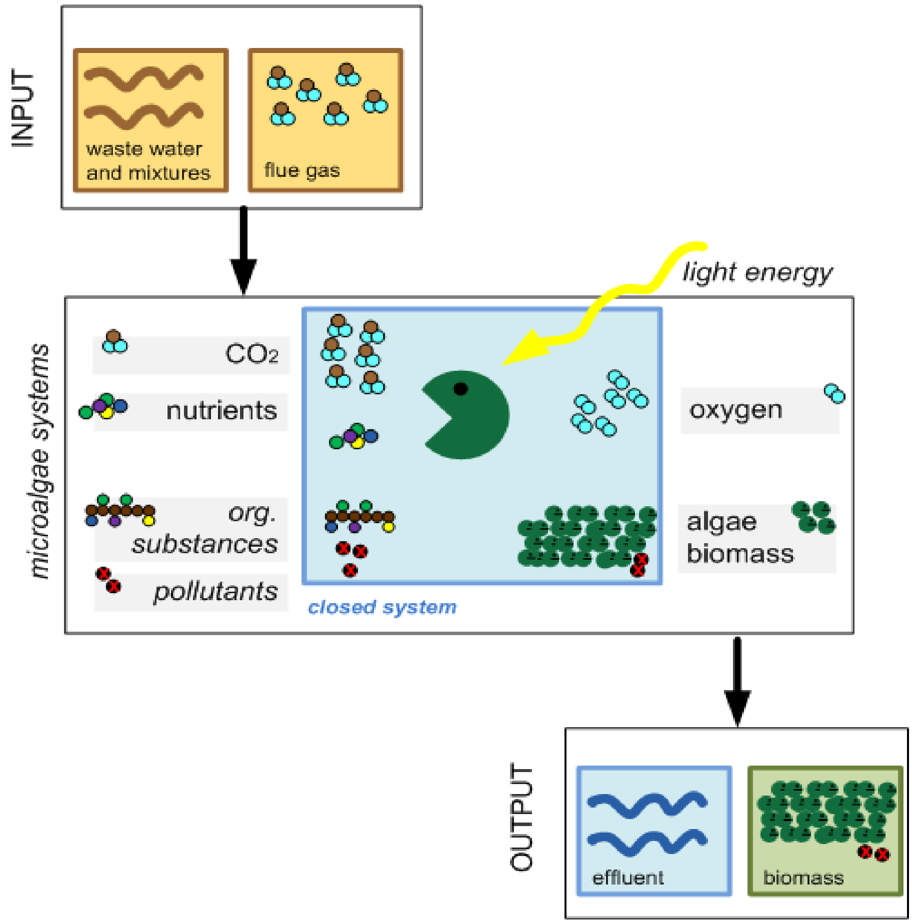


Figure 1-4: Algae systems for nutrient recycling in "closed" systems

Despite these advantages and potential synergies, only a few pilot projects of microalgae systems running with wastewater have been described, mainly located in the US (Sturm and Lamer 2010, Lundquist *et al.* 2010) and New Zealand (Park and Craggs 2010, 2011a-c). They confirm the technical feasibility of the concept.

The pilot plant in New Zealand is an open raceway ponds with additional $CO_2$ provision (high rate algae pond HRAP), running on (partially diluted) effluent from anaerobic digestion of sewage sludge. It achieves average areal biomass productivities of ~20 g/m²*d (Park and Craggs 2010, 2011a-c). The authors provide data on biomass productivities and elimination efficiencies for COD and N, but do not include an energy balance.

The pilot plant in the USA with a different process design (no additional $CO_2$ and primary treated waste water as growth medium) achieves ~10 g/m²*d (Sturm and Lamer 2010). This study provides no data on elimination efficiencies for COD and N. Instead, the authors assume a 90% elimination of N and P. This estimate is based on a laboratory study (Shi, 2007), which used a special process design, called twin sheet, for maximum elimination rates on lab scale. It is questionable, if these values can also be obtained in large-scale open ponds, especially for N. The authors provide a basic energy balance, assuming full substitution of wastewater treatment and not taking into account the energy required for energetic reuse of biomass (for example anaerobic digestion or drying and incineration.

Algae systems not using wastewater have been the subject of several life cycle based studies. Clarens *et al.* (2010) performs a life cycle analysis of different energy crops including algae and finds that algae have a lower total land use and eutrophication potential than conventional crops. But conventional crops have lower energy use and water use. The poor performance of algae is mainly due to high demands for nutrients, which are supplied in the form of mineral fertilizer in the study of Clarens *et al.* (2010). The authors conclude that algae perform more favorably in these impact categories if nutrients from wastewater are used, than with nutrients from mineral fertilizer.

Another study compares maximum areal energy output and minimum N requirements of different energy crops (Miller, 2010) and finds that algae have higher areal energy output than other energy crops, but also higher N requirements. If energy intensive mineral fertilizer is used, this trade off limits the energy output of the system. The authors conclude that using nutrients from wastewater avoids this trade off.

While algae systems receive much attention in literature, the combination with wastewater treatment is less often investigated. So far, there is no study of the integrated process of the WWTP and the algae systems. This study aims to fill this gap. A process design for integration of algae systems is proposed which relies solely on resources from wastewater, with no external input of water, fertilizer or $CO_2$. The whole algae process chain, from cultivation to production of bio-electricity, takes place on site of the WWTP.

This study is unique as it investigates the integrated system on level of substance flows of the major nutrients CNP. The model derived in this study includes an SFA of CNP. It allows assessing the energy and emission balance with a high level of detail. The SFA shows the impacts of algae systems on the internal cycling of CNP. Based on the nutrients provided to the algae systems combined with the uptake efficiencies and the stoichiometric requirements, the amount of biomass is calculated. From the amount of biomass generated, taking into account the harvesting efficiency and the anaerobic digestibility, the additional biogas generation is calculated. The SFA model also shows changes in loads to different treatment steps arising from rerouting of internal flows to algae systems. These changes in loads are used as proxies to calculate the changes in energy consumption of the WWTP.

The SFA is also the basis for the assessment of the emission balance. This is the first study to investigate effluent quality of algae systems at WWTPs, including the nutrients incorporated in the non-harvested biomass. Emissions to air and land are also taken into account. The emissions of $CO_2$ on site reflect the C efficiency of bio-electricity generation.

Another aspect that is often overlooked but very important from the perspective of the urban water chain is the implication for the flows of anthropogenic micropollutants (AMPs). To recap, the large advantage of algae systems is their closed nature. In contrast to "open" agricultural systems, algae systems can reuse nutrients without emission of AMP to the environment during cultivation. Furthermore, processes such as bio-oxidation, bio-sorption or bio-assimilation in algae systems themselves can contribute to elimination of anthropogenic pollutants, supported by a long hydraulic retention time of 3-6 days in an aerated environment. Elimination of heavy metals and persistent organic pollutants is described for laboratory studies, but remains to be proven in pilot projects. A list of substances for which elimination by microalgae was demonstrated is compiled in Table 1-1. Among these substances is also PFOS, the AMP chosen as a model substance in this study (see chapter 1.3.3).

The main processes are sorption and bioaccumulation, which transfer AMPs from effluent to algae biomass. The biomass is unsuitable for application as animal feed or fertilizer, but suitable for energetic use, for example co digestion and co incineration with sludge. Due to the small size, microalgae have a large surface area per gram biomass. Due to a variety of functional groups on the cell surfaces, they have effects on many different AMPs (cross substance effect, Monteiro *et al.* 2012, Subashchandrabose *et al.* 2013). For certain dyes, it has been demonstrated that microalgae can reach elimination efficiencies comparable to activated carbon (Aksu *et al.* 2005).

Table 1-1: Overview of substances for which elimination by microalgae was demonstrated

| Organic substance | Species (Reference) |
|---|---|
| Organic substances of industrial origin. | |
| PFOS | Mixed green algae (Liu et al. 2009) |
| Phenol | Ochromonas danica (Semple and Cain, 1996) |
| Tributyltin (TBT) | Chlorella vulgaris Chlorella sp. (Tsang et al., 1999)<br>Chlorella miniata (Tam et al., 2002) |
| Benzo[a]pyrene (BaP) | Selanastrum capricornutum (Warshawsky et al., 1988)<br>S. capricornutum (Schoeny et al., 1988) |
| Phenanthrene (PHE) | S. capricornutum (Chan et al., 2006) |
| Naphthalene | Agmenellum quadruplicatum (Cerniglia et al., 1979)<br>Chlorella vulgaris (Todd et al., 2002) |
| 1-Naphthalenesulfonic acid | Scenedesmus obiquus (Kneifel et al., 1997) |
| 1-Methylnaphthalene<br>2-Methylnaphthalene | A. quadruplicatum, Oscillatoria sp., Anabaena sp. (Cerniglia et al., 1983) |
| | A. quadruplicatum, Oscillatoria sp. Anabaena sp. (Cerniglia et al., 1983) |
| 2,4,6-Trinitrotoluene | Anabaena sp. (Pavlostathis and Jackson, 1999) |
| Dibenzofuran | Ankistrodesmus sp. (Todd et al., 2002) |
| Dibenzo-p-dioxin | Scenedesmus sp. (Todd et al., 2002) |
| Bisphenol | Chlorella fusca (Hirooka et al., 2005) |
| Bisphenol A | Pseudokirchneriella subcapitata, Scenedesmus acutus<br>Coelastrum reticulatum (Nakajima et al., 2007) |
| Biphenyl | Oscillatoria sp. (Cerniglia et al., 1980) |
| Dimethyl phthalate | Closterium lunula (Yan and Pan, 2004) |
| Sinapic acid | Stichococcus bacillaris (DellaGreca et al., 2003) |
| Azo compounds | Chlorella vulgaris (Jinqi and Houtian, 1992) |
| aromatic pollutants | algae–bacteria mixed culture (Borde et al. 2003) |

Table 1-1 (continued): Overview of substances for which elimination by microalgae was demonstrated

| *Organic substance* | *Species (Reference)* |
|---|---|
| **Organic substance of agricultural origin.** | |
| DDT | Aulosira fertilissima; (Lal et al., 1987); Chlorococcum sp.; Anabaena sp.; Nostoc sp.; (Megharaj et al., 2000) |
| γ-Hexachlorocyclohexane (Lindane) | Anabaena sp.; Anabaena sp. (pRL634); (Kuritz and Wolk, 1995 |
| Methyl parathion | Chlorella vulgaris; Scenedesmus bijugatus; Nostoc linckia,; N. muscorum; Oscillatoria animalis; Phormidium foveolarum; (Megharaj et al., 1994) |
| | Anabaena sp.; (Barton et al., 2004) |
| Metflurazon Norflurazon | Chlorella fusca; (Thies et al., 1996) |
| Norflurazon Desmethyl derivative | Chlorella fusca; (Thies et al., 1996) |
| Fluometuron | Ankistrodesmus cf.; Nannoselene, Selenastrum; capricornutum; (Zablotowicz et al., 1998) |
| Atrazine Diethyl | Ankistrodesmus sp.; Selenastrum sp.; (Zablotowicz et al., 1998) |
| α-Endosulfan | Scenedesmus sp.; Chlorococcum sp.; ; Scenedesmus sp.; (Sethunathan et al., 2004) |
| Diclofop-methyl (DM) | Chlorella vulgaris; C. pyrenoidosa; Scenedesmus obliquus; (Cai et al., 2007) |
| Dichlorprop-methyl (2,4-DCPPM) | Chlorella pyrenoidosa,; C. vulgaris; Scenedesmus obliquus; (Li et al., 2008) |
| Fenamiphos | Pseudokirchneriella; subcapitata; Chlorococcum sp.; (Cáceres et al., 2008a) |
| **Metals** | |
| $Cu^{2+}$ | Doshi et al. 2007; Deng et al., 2007; Vijayaraghavan et al., 2006 |
| $Ni^{2+}$ | Al-Rub et al., 2007 |
| $Zn^{2+}$ | Monteiro et al, 2011; Monteiro et al., 2009 b; Romera et al., 2007; Senthilkumar et al., 2006 |
| $Cd^{2+}$ | Aksu and Dönmez, 2006; Tüzün et al., 2005; Monteiro et al., 2009a; Monteiro et al., 2009 b; Romera et al., 2007 |

### 1.3.5 The concept of the urban metabolism

The concept of the urban metabolism (UM) was introduced by Wolman (1965). He analyzed a hypothetical city by the quantification of inputs: water, food and fuel; transformation processes and outputs – sewage, solid refuse and air pollutants. He highlighted three "metabolic challenges": water supply management, sewage disposal and air pollution control. Since its introduction, the concept has received growing attention. A recent review of studies can be found in Kennedy *et al.* (2011), showing an accelerated interest in the last decade. Practical applications of the concept are emerging, especially as a basis for sustainable urban design and material flow management (Kennedy *et al.* 2011, Agudelo-Vera *et al*. 2012, Girardet 2012, Villarroel Walker 2010, Beck *et al* 2010).

To recap from the introduction, a sustainable urban metabolism requires restructuring the present linear metabolism of cities to a more circular one. Brunner (2007) categorizes cities as linear reactors whose metabolism remains open and vulnerable depending on the hinterlands for material supply and disposal. In essence, the linear pattern of production, consumption and disposal is different than nature's circular metabolism. Natural ecosystems are generally energy self sufficient, or are subsidized by sustainable inputs, and often approximately conserve mass, through recycling by detrivores.

On a predominantly urban planet, cities will need to adopt circular metabolic systems to assure their own long term viability as well as that of the rural environments on which they depend (Girardet 2010, Crutzen *et al.* 2007). Otherwise, cities will continue to be strong agents of environmental decline on a local to global level and at the same time be vulnerable to these changes (Grimm *et al.,* 2008). The local to global effects of the highly altered biogeochemical cycles in cities include: altered air quality (smog, aerosol load on local level, GHGs on global level), altered urban soils and vegetation, altered hydrological dynamics and altered water quality (urban stream syndrome[12], groundwater pollution), altered dynamics for pollutants (urban pollution halo). These (partly) interdependent factors make urban ecosystems distinct from natural ecosystems, calling for a "distinct urban biogeochemistry" (Kaye *et al*. 2006).

With a linear metabolism as today, resource consumption in urban areas is large. For a typical city in an average industrialized country, consumptions per capita per year are 150–400 GJ for

[12] The "urban stream syndrome" is a conceptual model to describe the consistently observed ecological degradation of streams draining urban landscapes (Walsh *et al.*, 2005). This degradation includes elevated nutrients, increased organic and inorganic contaminants, increased hydrologic flashiness, and altered biotic assemblages.

energy, and 15–25 tons for materials (Krausmann *et al*., 2008). Large portions of the flows are exported out of the urban system: in form of wastewater, solid waste and demolished construction materials. But others remain in the urban system as urban ‘stocks’ (Brunner, 2007). While throughput is large, also the material stock in urban areas is growing, as inputs typically outweigh the outputs. The “urban stock” grows by approximately 10 t/p*a[13]. While the bulk accumulation is from construction material and long lived goods, also AMPs accumulate in the urban stock. Emissions from urban stock are significant for present and future flows of AMPs.

The linear metabolism can be associated with two main problems. On the one hand, the high rate of resources consumption is related to resource depletion (e.g. water, fossil energy carriers, and P resources); on the other hand, massive disposal of gaseous, liquid and solid waste causes pollution. Pollution refers to misplaced resources (CNP), to chemical pollution[14] (AMPs), but also by chemically relative inert materials such as plastics.

With a circular metabolism, that includes recycling and reuse of the different urban flows, the problems of the input and output side can be negotiated. A circular metabolism resembles the metabolism of natural ecosystems, has a low consumption rate, and less impact on hinterlands. A circular metabolism may also reduce the dependency on imports of material and energy and thus enhance the resilience of cities (Agudelo-Vera *et al*. 2012). To recap, as also AMPs are present in urban areas, not only the quantity, but also the quality of cycles needs to be taken into account as required by the clean cycle approach (Kral, Kellner and Brunner 2012).

According to Baccini and Brunner (1991), four major urban activities are the drivers of material and energy flows: to nourish and recover; to clean; to reside and work; and to transport and communicate. The flows of water, food (biomass), materials (construction materials, goods and products), and energy can be regarded as the four fundamental components of urban metabolism (Decker *et al*. 2000, Kennedy 2010).

The urban metabolism can be analyzed with different complementary perspectives: from a mass balance approach (Material Flow Analysis (MFA), see Baccini and Brunner 1991, Baccini and Bader 1996, Brunner and Rechberger 2003), on level of substances or elements

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13 According to Moll, Bringezu and Schütz, the net accumulation in the EU is 10 t of materials per person and year. This value does not distinguish between urban (astysphere) and the anthroposphere in general.

14 As defined by Rockström *et al.* 2009

(Substance Flow Analysis (SFA), see Antikainen 2007) or on level of (direct and embodied) energy flows (Emergy analysis, see Odum 1983).

Drawing boundaries around cities is difficult, as they are tightly connected: via roads and other infrastructures, transport of goods and wastes and human travel and commuting. This global network of cities can be regarded as a part of the anthroposphere: the astysphere (Norra 2009). A full analysis of urban metabolism includes all commercial, industrial or agricultural activities that take place within the city boundaries, as well as export and import; and the household activities. Given the diversity of cities and their administrative boundaries; the urban metabolism can include any kind of production (Figure 1-5). The scope of the present study is limited to household consumption.

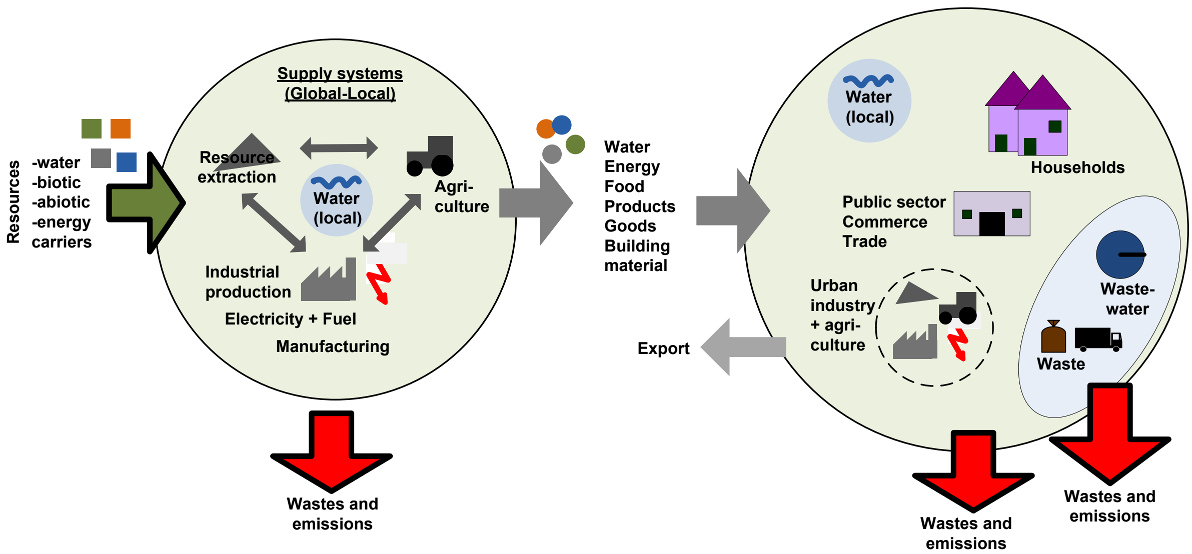


Figure 1-5: Flows and activities of the urban metabolism

Legend: Flows and activities of the urban metabolism include export and import; commercial, industrial, agricultural and household activities. The focus of this study is on the latter.

On the way towards a sustainable future, cities face many challenges. As a prerequisite for the physical sustainability, cities need to restructure their resource consumption and energy systems to negotiate the human impact on the ecosystems on which they ultimately depend. This includes a sustainable, C efficient and renewable energy supply, as well as recycling of resources from food and other consumption related urban flows in clean cycles. At the same time, cities need to find ways to minimize pollution of the environment, including AMPs. It is important for policy and decision makers to understand how “clean cycles” can be organized.

To contribute to this task, the present study focuses on urban water infrastructures and their potential contribution to a more sustainable urban metabolism with clean cycles. The urban water chain, as part of urban metabolism, manages large flows of water, food (biomass) and also the flows of some products of daily consumption, such as cleansing products. The flows are related to the activities: to nourish, to clean and also to reside, as water supply and sanitation are fundamental functions of a residence.

On an elemental level, the urban water chain includes large flows of CNP, originating from food and cleansing products. While CNP are resources with an energetic value, they can act as pollutants when emitted to the environment. Together with CNP, AMPs can enter the urban water chain. The urban water chain is an important pathway for AMPs to the environment. Thus, to evaluate concepts for recycling of CNP the quantity and quality of recycling needs to be considered.

This study assesses the status quo of the urban water chain and the potential for energetic recycling of CNP in algae systems. Finally, the perspective is extended to put this potential improvement in context of urban metabolism. To extent the perspective, the household consumption of energy, water, food and cleansing product for house and body (detergents) is quantified. These flows represent the connecting points of the urban water chain to the full urban metabolism. They cover large parts of the daily household consumption.

While the analysis of household consumption is far from a complete analysis of urban metabolism, it covers an important part of the system. The household consumption was identified as an important driver of material and energy flows and socially meaningful as unit of decision-making (Moll 2006, Fissore *et al* 2011, Villarroel Walker 2010).

Amongst the urban metabolism studies, there are some that focus on the flows of CNP through urban areas and their degree of reuse. But most of them neglect the problem of AMPs, thus following a purely quantitative approach to recycling. Studies of the urban metabolism with focus on the flow of nutrients have been performed for:

- Paris in France (Barles 2007: N flows in food sector),
- Linköping in Sweden (Neset *et al.* 2007: N flows in food sector, Neset *et al.* 2009: P flows in food sector, Neset 2005)
- Phoenix in the USA (P flows with the sectors food, forestry, municipal waste, Metson 2012)
- Minneapolis-St. Paul in the USA (N and P flows with focus on household consumption, Baker *et al.* 2007, Fissore *et al* 2011)

- on a watershed level: Upper Chattahoochee Watershed in the USA (C N and P flows with the sectors food, forestry, municipal waste, and energy,  Villarroel Walker 2010)
- on national level in Finland (Antikainen 2007, N and P flows, with the sectors food, forestry, municipal waste, and energy)

# 2 Methods and Data Base

## 2.1 Framework for analysis

Methodologically, a systems analysis is employed combining a conventional energy balance with a substance flow analysis (SFA) and the assessment of the energetic value of the resources. The underlying model includes three layers (Figure 2-1, left side). It is build in an excel spreadsheet. The technical setup builds the foundation of the model (layer 1). Different technical setups (cases) are defined to describe the status quo of the urban water chain, including the water supply, the wastewater and the sludge management; and the integration of algae systems.

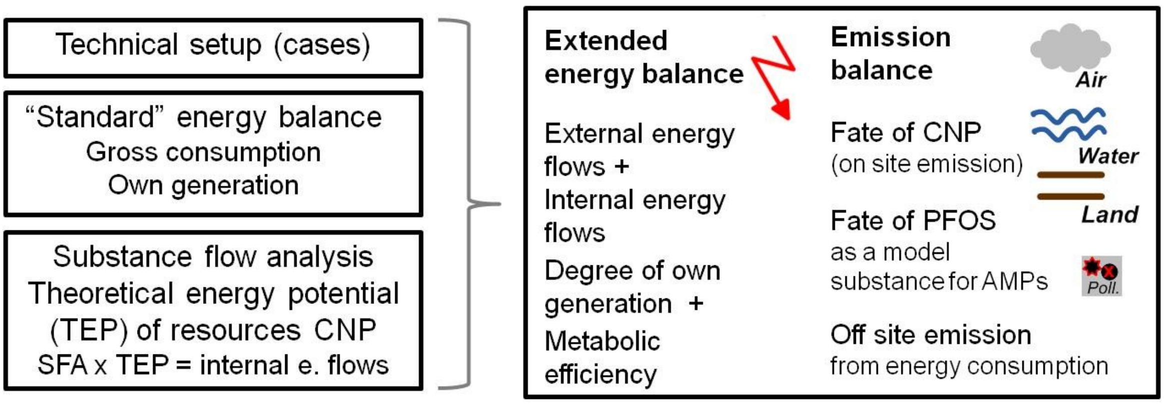


Figure 2-1: Framework for analysis

For each case, the gross consumption for handling and treatment of flow streams is compiled: electricity, thermal energy and fuel consumption for each step of the urban water chain. Together with the own generation of energy from biogas use or sludge incineration, these energy flows represent the external energy flows of the system (layer 2).

Table 2-1 provides an overview of the factors used for calculation of the external energy balance of the urban water chain. The derivation of the factors is described in detail in chapter 2.4.2. For the algae systems, the factors are adapted based on the results of the SFA to account for the altered flows of CNP (see chapter 2.5.4). The SFA shows changes in loads to different treatment steps arising from rerouting of internal flows to algae systems. These changes in loads are used as proxies to calculate the changes in energy consumption. The energy generation is also assessed based on SFA results. Based on the nutrients provided to the algae systems combined with the uptake efficiencies and the stoichiometric requirements, the amount of biomass is calculated. From the amount of biomass generated, taking into account the harvesting efficiency and the anaerobic digestibility, the additional biogas generation is calculated.

Table 2-1: Overview of factors for calculation of the external energy balance

For each step of the urban water chain, the energy consumption is assigned in kWh/p*a per person served and year of electricity EL, thermal energy TE or PE (fuels for transport)

| | *advanced* | | *medium* | | *basic* | | *All* | *References* |
|---|---|---|---|---|---|---|---|---|
| | *EL* | *TE* | *EL* | *TE* | *EL* | *TE* | *PE* | |
| Water supply | *26* | | *26* | | *26* | | | *HWW, 2007; RWW, 2007; Olsson 2012; Lingsten et al. 2008* |
| Wastewater transport | *5.5* | | *5.5* | | *5.5* | *0* | | *Olsson 2012; Lingsten et al. 2008; Hansen et al. 2007* |
| Wastewater treatment (BWT) | *28* | | *28* | | *40* | *5* | | *Haberkern et al. 2008; Agis 2001; Hansen et al. 2007; Olsson 2012; Lingsten et al. 2008* |
| Digester operation | *6* | *22* | *6* | *22* | | | | |
| Biogas co generation | *-9* | *-18* | *-9* | *-18* | | | | |
| Sludge dewatering | *2* | | *2* | | *2* | | | *Haberkern et al. 2008; Houillon et al. 2005; UBA 2012; Hong et al. 2008; Stillwell et al. 2010; MUNLV 2001, destatis 2006, Agis 2001; Hansen et al. 2007; Olsson 2012; Lingsten et* |
| Sludge storage | | | *3.5* | | *3.5* | | | |
| Sludge transport | | | | | | | *7-20* | |
| Sludge drying | *5* | *44* | | | | | | |
| Sludge incineration | *-12* | *-32* | | | | | | |
| Waste TE in | | *-8* | | | | | | |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Flue gas treatment | *2.4* | | | | | | | *al. 2008* |

Zooming in on level of CNP, the substance flow analysis (SFA) shows the flows of CNP along the urban water chain (layer 3). CNP enter the urban water chain at household level and follow complex pathways during wastewater treatment, even more complex when algae systems are involved.

In the SFA model, each flow is represented by a vector of 4 elements. Each treatment step in the different cases is represented by a matrix with partitioning factors (PF) to air, water and sludge (module). Thus, each module contains 3 x 4 = 12 partitioning factors (see Table 2-2). Adding up along the rows gives the sum of 1 for each of the vector elements CNPW, as the input to the module equals the output. The mathematical operations to calculate the flows are reported in chapter 2.4.4 for the urban water chain without algae and in chapter 2.5.4 for the urban water chain with algae.

Table 2-2: Overview of matrix for the SFA

Legend: The partitioning factors are named after the element CNPW, followed by the destination of the flow –a for air; -w to water; -s to sludge / biomass.

| PF for module X | X-A | X-W | X-S | Σ |
|---|---|---|---|---|
| C | c-a | c-w | c-s | **1** |
| N | n-a | n-w | n-s | **1** |
| P | p-a | p-w | p-s | **1** |
| W | w-a | w-w | w-s | **1** |

To extend the usual approach to energy balances, the analysis also assesses the energetic value of the resources. The theoretical energy potential (TEP) is assigned to the resources CNP. It relates to the chemical energy of C and the “grey” energy of the nutrients N and P. By applying the respective TEP factors [kWh/kg] to the SFA, the internal energy flows associated to CNP are traced along the urban water chain. Table 2-3 gives an overview of the factors for the theoretical energy potential (TEP) of CNP used in this study. The derivation of TEP factors is documented in chapter 2.3.

Table 2-3: Overview of TEP factors

| TEP of | Value used in this study | Value based on | References |
|---|---|---|---|
| C | 17.9 kWh/kg C (Chemical energy: direct combustion) | Measurements with bomb calorimeter (freeze-dried samples) | Heidrich *et al.* (2011) Shizas and Bagley 2004 |
| C | 13 kWh/kg C (Chemical energy: via biogas) | Chemical composition and calorific value of biogas | Sverdal-Kroiss (2012) Olsson (2012) |
| N | 16.4 kWh/kg N (Grey energy)[15] | Energy for fertilizer production | Lal 2004, Dockhorn 2008 |
| P | 7.9 kWh/kg P (Grey energy) | Energy for mining and processing | Maurer 2003 |

Based on the TEP factors, the metabolic efficiency can be assessed. The metabolic efficiency is the degree of reuse in relation to the full energetic value of the resources. For N and P it relates to the amount applied to agricultural land corrected for plant availability. The reuse of N and P can be expressed in energetic terms by applying the TEP factors to SFA results.

C is energetically reused via biogas use or sludge incineration for electricity and heat co generation, as quantified in layer 2 of the model. To assess the metabolic efficiency of C, the energy generation is put in relation to the TEP of C. Furthermore, the actual and potential electricity generation from C resources can be put into relation to the actual electricity consumption, as shown in Figure 2-2.

This extended energy balance is an important read out of the model (Figure 2-1, right side). It gives a holistic picture of the energy flows along the urban water chain.

[15] The term grey energy refers to energy saved by substituting a product i.e. fertilizer.The grey energy is “virtual” i.e. not directly usable within the system, in contrast to chemical energy or electrical energy.

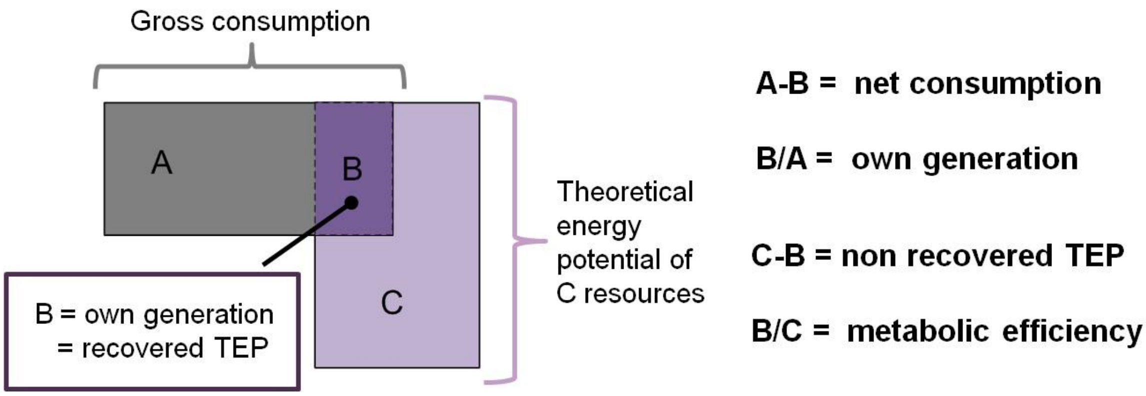


Figure 2-2: Elements of the extended energy balance

Legend: The extended energy balance shows external and internal energy flows. A: gross consumption of electricity for handling of flow streams (grey); B: own generation of electricity from the resources in the flow streams (dark purple); C theoretical potential for electricity generation based on the resources in the flow streams (light purple); B/C: metabolic efficiency.

To assess not only the quantity, but also the quality of CNP recycling, the extended energy balance of the urban water chain is accompanied by the emission balance. The emission balance includes the emissions of CNP from the flow streams (internal resources). Thus, the method captures the double role of CNP as potential energetic resources or as pollutants. The SFA is the basis for assessment of the emission balance. The mathematical operations for calculation of the emission balance are reported in chapter 2.4.4 for the urban water chain without algae and in chapter 2.5.4 for the urban water chain with algae.

Besides the on site emissions of CNP, also the $CO_2$ emissions related to the consumption of external energy are taken into account. While the focus of the study lies on the flows of CNP, the problem of AMPs is discussed using PFOS as a model substance. The inventory for the flows of PFOS is presented in chapter 2.6.9.

The emission balance is the second important read out of the model (Figure 2-1, right side). Together, the extended energy and emission balance of the urban water chain allow assessing the quantity and quality of reuse.

## 2.2 Input load

In all cases, the same input is assumed. The average daily load to wastewater generated per person and day is 120 g COD, 11 g N and 2 g P (ATV 2000, DWA 2008, and DWA 2013). Assuming an average factor from C to COD of 3 (range 2.8-3.2, Henze *et al.* 2000); the daily load COD load equals 38.8 g C. The amount of wastewater is assumed with 250 l per day and person served. In this analysis, only wastewater from households is considered. Wastewater from commercial operations is excluded.

## 2.3 Theoretical energy potential of CNP

*Carbon (C)*

Carbon (C), in dependency of its oxidation state, is an energy carrier. Wastewater contains a multitude of organic molecules, such as hydrocarbons, proteins and lipids. When organic molecules are oxidized to $CO_2$, e.g. during combustion, energy is released. The quantification of the theoretical energy potential TEP of organic carbon C is based on a study with bomb calorimeter. Heidrich *et al.* (2011) measured 7.6 kJ/l in a domestic wastewater with 0.115 g/l of organic C, using freeze-dried samples. For the model, the resulting TEP factor of 66 kJ/g C is used (17.9 kWh/kg).

Bomb calorimeters measure the calorific value of a dried substrate, therefore the TEP contains no corrections for water content. In practice, drying of substrate can easily consume more energy than contained in the substrate (see chapter 2.4.2). The TEP represents therefore the upper limit for energy harvesting from C in wastewater. Freeze drying of samples gave higher calorific values than oven drying, as losses during sample preparation were minimized (Heidrich *et al.* 2011).

In a mixed wastewater, including wastewater from commerce and business, higher values of 22.5 kJ/g COD in oven dried samples and 28.7 kJ/g COD in freeze dried samples were measured (Heidrich *et al.* 2011). Assuming an average factor from C to COD of 3 (range 2.8-3.2, Henze *et al.* 2000), calorific value lies between 66 and 86 kJ/g C. In an earlier study, (Shizas and Bagley 2004) measured lower values of 14.7 kJ/g COD or 45 kJ/g C with oven dried samples.

Besides energy harvesting from C *via* combustion processes, also a “detour” *via* biogas processes can be used. The TEP of C via biogas processes is quantified based on an assumed biogas composition of 65% methane (see Table 2-4), which is typical for biogas at WWTPs (Haberkern *et al.* 2008). With the molecular weights of the components, 1 mol of biogas weights 24 g and has a C content of 46%. Using a volume of 22.414 l/mol (ideal gases) for simplicity, this gives a density of 1.08 g/l and approximately 0.5 g C/l biogas. With a lower heating value of 6.5 kWh/m³, this gives 13 kWh/kg C or 48 kJ/g C. This is in the same range as the estimate presented by other studies (3.5-4 kWh/g COD, or 10.5-12 kWh/g C, Sverdal-Kroiss (2012) and Olsson (2012)).

This figure represents the upper limit for energy harvesting from C in wastewater via biogas, based on a 100% anaerobic digestibility of substrates. In practice, much of the substrate is resistant to biodegradation under anaerobic conditions and anaerobic digestibilities range from 35% for secondary sludge to 55-70% for primary sludge (DWA 2007). While TEP of biogas will always be below TEP of combustion, biogas has multiple advantages in practice. While combustion requires drying of substrate, anaerobic digestion (AD) uses wet substrates. For the WWTP, anaerobic digestion is a method for sludge stabilization, with biogas as co-product.

Table 2-4: Derivation of TEP for C via biogas

Legend: Derivation of TEP for C via biogas based on assumed biogas composition and molecular weights of biogas components.

| | $CH_4$ | $CO_2$ | $H_2O$ | $N_2$ |
|---|---|---|---|---|
| molecular weight [g] | 16 | 44 | 18 | 30 |
| biogas composition [%] | 65 | 28 | 6 | 1 |

*Nutrients NP*

Other than C resources, N and P in wastewater can not be used directly for generation of electricity and heat. But reuse on agricultural lands gains indirect energy credits by substitution of energy-intensive fertilizer production. Fertilizer production via Haber-Bosch requires 60 MJ/kg of N (Lal 2004, Dockhorn 2008) For P, energy intensity for mining and processing is estimated at 29 MJ/kg P (Maurer 2003). For N, there are no limitations in resource availability as $N_2$ is abundant in the atmosphere. But P resources are limited and energy demand for processing is expected to rise, as good quality resources decline. For the model, a TEP factor of nutrients of 60 kJ/g N (16.4 kWh/kg) and 29 kJ/g P (7.9 kWh/kg) is assumed. This represents the grey energy of nutrient provision.

## 2.4 Inventory for the current situation of the urban water chain

### 2.4.1 System description

The mass flows of the urban water chain are described in chapter 1.3.1. Based on the reported information (see Table 2-5), the cases required to describe the current situation are derived.

Table 2-5: Status quo: Technical setup for water supply, wastewater treatment and sludge treatment

| *Technical setup* | *Value* | *Important references* |
|---|---|---|
| ***water supply*** | | |
| groundwater | 70% | ATT *et al.* 2011 |
| surface water | 30% | |
| ***wastewater treatment*** | | |
| anaerobic stabilization of sludge | 75% | ATT et al. 2011, UBA 2012; Haberkern et al. 2008 |
| aerobic sludge treatment | 25% | |
| ***sludge treatment*** | | |
| incineration | 52% | ATT *et al.* 2011, UBA 2012; Haberkern *et al.* 2008 |
| land use in agriculture or landscaping | 48% | |

For the water supply side two cases are distinguished in this study with groundwater (70%) and surface water use (30%, ATT *et al.* 2011). For wastewater and sludge treatment, three cases are distinguished, which differ in their degree of energy harvesting from biogas and sludge. This includes facilities with anaerobic stabilization of sludge (75% of population served) and facilities with aerobic sludge treatment (25%) (ATT *et al.* 2011, UBA 2012, Haberkern *et al.* 2008). For fate of stabilized sludge, incineration (52% of population served) and land use in agriculture or landscaping (45%) is considered (ATT *et al.* 2011, UBA 2012). For simplicity, it is assumed that only facilities with anaerobic digestion conduct sludge incineration and the 3% other uses of sludge are added to land use. This gives three cases as shown in Table 2-6 (see also Figure 3-1 in the results section). The inventories for the external energy balance and for the SFA for these three cases are reported in the following chapters.

Table 2-6: Status quo: Three cases for wastewater and sludge management

| Technical setup | Biogas use | Sludge incineration | Population served |
|---|---|---|---|
| **Advanced case (1)** | ✓ | ✓ | 52% |
| **Medium case (2)** | ✓ | ✗ | 23% |
| **Basic case (3)** | ✗ | ✗ | 25% |

The advanced case (case 1) represents a medium to large WWTP employing anaerobic sludge stabilization with subsequent drying and incineration (52% of population served). The medium case (case 2) represents a medium to large WWTP employing anaerobic sludge stabilization with subsequent land use (23%). The basic case (case 3) represents a small WWTP with aerobic wastewater treatment and simultaneous sludge stabilization with subsequent land use (25%). All cases employ biological wastewater treatment (BWT, also referred to as activated sludge treatment AS). The weighted average of these three cases gives the average for Germany.

The advanced case (case 1) employs energy harvesting from biogas and incineration processes. As it reflects the current situation, there are considerable energy losses: due to flaring or thermal only use of biogas, due to a low electrical efficiency and due to inefficient mono incineration in older facilities. If these processes were optimized according to the best available technology, the energetic reuse can be considerably increased (best available technology case or case 1+). For the SFA, there are no differences compared to the advanced case (case 1).

### 2.4.2 External energy balance

*Drinking water supply*

For drinking water supply, an average of 110 l/p*d for domestic use and 6% distribution losses (ATT *et al.* 2011) are assumed, adding up to 41 $m^3$/p*a. For electricity demand for water sourcing, treatment and distribution, 0.6 kWh/$m^3$ is assumed. This estimate is based on information from two large German suppliers, reporting 0.57 kWh/$m^3$ (HWW, 2007) and 0.62 kWh/$m^3$ (RWW, 2007) for groundwater and 0.83 kWh/$m^3$ for surface water (RWW, 2007); taking into account that in Germany, 70% of public water supply is sourced from groundwater and 30% from surface water (ATT *et al.* 2011). This estimate for Germany is slightly higher than average reported for the Netherlands (0.47 kWh/$m^3$, Olsson 2012) and for Sweden (0.46 kWh/$m^3$, Lingsten *et al.* 2008).

The demand for thermal energy in the water sector is neglected in this study, as there is no statistical information available. As the treatment processes themselves do not require input of thermal energy, it leaves the heating of facilities and office rooms.

*Wastewater transport*

Transport of wastewater requires electricity for pumping. It is assumed that 91 $m^3$/p*a wastewater are generated in average, including rainwater and sewer infiltration. The electricity demand is assumed with 0.06 kWh/$m^3$, as reported as average for Sweden (Olsson 2012, Lingsten *et al.* 2008). This gives an electricity demand for wastewater transport of 5.5 $kWh_{el}$/p*a. This figure is in the same range as another estimate from Germany (5.8 $kWh_{el}$/p*a, Hansen *et al.* 2007).

*Wastewater treatment*

Data for electricity use for wastewater treatment processes from benchmarking studies is compiled in Haberkern *et al.* (2008). Average electricity demand is 35 $kWh_{el}$/p*a, with considerable differences between size classes: 75 $kWh_{el}$/p*a for size class 1, 55 $kWh_{el}$/p*a for size class 2, 44 $kWh_{el}$/p*a for size class 3, 34 $kWh_{el}$/p*a for size class 4 and 32 $kWh_{el}$/p*a for size class 5.

For the basic case (case 3) of the model, representing 25% of population served by smaller WWTPs with simultaneous aerobic sludge stabilization, the demand is assumed with

40 $kWh_{el}$/p*a. For the advanced case (case 1) and the medium case (case 2), representing 75% of population served by larger WWTPs employing anaerobic sludge stabilization, 33 $kWh_{el}$/p*a can be assumed The average over all three cases equals the German average of 35 $kWh_{el}$/p*a, as reported by Haberkern *et al.* (2008).

For facilities with simultaneous aerobic sludge stabilization (basic case 3), approximately 80% of the electricity are used for aeration, the rest for other purposes e.g. pumping of wastewater (Agis 2001). For facilities with anaerobic sludge stabilization (advanced case 1 and medium case 2), approximately 60% of the electricity is used for aeration during biological wastewater treatment and 20% for digester operation (Agis 2001).

The demand for thermal energy at WWTPs depends on the form of sludge treatment. With anaerobic sludge stabilization, digester operation requires input of thermal energy, which can be met in large parts by co generation of electricity and heat from biogas on site. Aerobic sludge stabilization requires less thermal energy than anaerobic sludge stabilization. But without biogas, there is no electricity or heat generated on site. According to an energy analysis of 21 WWTPs in Austria (Agis 2001), WWTPs with anaerobic sludge stabilization require a total of 22 $kWh_{thermal}$/p*a, with most of the demand covered by generation of thermal energy from biogas. WWTPs with aerobic sludge stabilization requires 2-5 $kWh_{thermal}$/p*a. In absence of AD, this has to be fully covered by external supply e.g. natural gas.

For the model, 22 $kWh_{thermal}$/p*a is assumed for case 1 and 2 with anaerobic sludge stabilization, and 5 $kWh_{thermal}$/p*a for case 3 with simultaneous aerobic sludge stabilization. For simplicity, it is assumed that thermal energy is provided by natural gas only.

*Electricity generation from biogas*

For Germany, average biogas generation per person at WWTPs is reported with 7.2 $m^3$/p*a, with ranges of 6.2-11.5 $m^3$/p*a resp. 16-32 l/p*d (Haberkern *et al.* 2008). Biogas from WWTPs averages 65% methane content and has a lower heating value of 6.5 kWh/$m^3$. This gives a lower heating value of biogas of 47 kWh/p*a. But only approximately 65% of biogas is used for electricity generation, with an average efficiency of 30%. The remaining 35% of biogas are flared or used for thermal applications only (Haberkern *et al.* 2008).

For case 1 and 2 of the model, 7.2 $m^3$/p*a of biogas generation is assumed, with 65% methane content and a lower heating value of 6.5 kWh/$m^3$. It is assumed that 65% of produced biogas is used for co generation. There are no energy credits for the remaining 35%. For co generation, 30% electrical efficiency and 60% thermal efficiency is assumed, thus yielding 9 $kWh_{el}$/p*a and 18 $kWh_{therm}$/p*a. For simplicity, it is assumed that the electricity generated is fully used on site, and the 14% of the electricity sold to grid is neglected (Haberkern *et al.* 2008).

For the best available technology case, it is assumed that the produced biogas fully contributes to electricity generation, i.e. no biogas is flared or used for thermal applications only. The electrical efficiency is assumed with 35%. Thus, electricity generation from biogas can be increased to 16 $kWh_{el}$/p*a in the best available technology case or case 1+.

*Sludge treatment*

After anaerobic or aerobic stabilization, sludge has a solids content of 2-5%, expressed as dry weight (dw). For volume reduction, sludge is usually dewatered. Technologies for dewatering of sludge include centrifuges or presses. The water content of sludge is divided into the following categories: free water, which can largely be removed by mechanical means, capillary water and water bound to particle’s surface, which can be removed by thermal drying and chemically bonded water molecules. Sludge is very hydrophilic due to a high organic content, composed mainly of cells of microorganisms containing nucleic acids, proteins, carbohydrates and lipids, their decay products and non metabolized organic material, e.g. cellulose (Manara and Zabaniotou 2012).

Depending on the individual characteristics of sludge, mechanical dewatering can raise the solids content to approximately 15-25% dw for aerobically stabilized sludge and 20-30% dw for aerobically stabilized sludge. Before dewatering, sludge is usually treated with flocculants, such as organic polyelectrolytes or salts of trivalent Al or Fe, to aggregate solids and help dewatering (DWA 2007). According to Haberkern *et al.* (2008), mechanical dewatering of sludge requires approximately 2 $kWh_{el}$/p*a and no thermal energy.

For sludge treatment, two pathways are considered: incineration (52% of population served, case 1) and land use (48%, case 2+3). For all cases, dewatering of sludge is assumed neglecting the direct use of liquid sludge e.g. on agricultural lands in vegetation season. For land use the scenario presented by Houillon *et al.* (2005) is adopted with spreading of pasty sludge with 20% dw after an average storage of 7 months. Electricity demand for storage under deodorizing conditions is 209 kWh/t dw (Houillon *et al.* 2005). Assuming 17 kg dw/p*a, electricity demand for storage is 3.5 $kWh_{el}$/p*a.

There is no information available for transport distances of sludge to reuse sites in Germany. To give an envelope for energy demand of sludge transport, an average transport distance of 20 km (MUNLV 2001) as lower and 600 km (Haberkern *et al.* 2008) as higher value is assumed. With 20% dw, the mass for transport is 85 kg/p*a. According to statistics, the energy demand (fuels) for transport is 0.532 kWh/tkm and the corresponding $CO_2$ emission is 0.135 kg $CO_2$-equ./tkm (destatis 2006).

For incineration of stabilized sludge (case 1), the main processes employed in Germany are mono incineration (43% of sludge incinerated) and co incineration in coal fired power plants (43%) (UBA 2012). For co incineration in coal fired power plants, drying is usually integrated with drying of coal in coal mills. Thermal energy for drying comes from waste heat. Sludge addition to coal is limited to 1-5%, due to ash contents, residues and capacity of coal mills (UBA 2012). For mono incineration, the processes most commonly employed are multiple hearth furnaces and fluidized bed furnaces (Stillwell *et al.* 2010).To generate enough heat for a stable process without large amounts of auxiliary fuels, mono incineration facilities usually receive sludge from several WWTPs. Remaining 14% of sludge are co incinerated in cement kilns, waste treatment facilities and others (UBA 2012).

The calorific value of dried sewage sludge is comparable to brown coal: sludge at 75% dw has a calorific value of 7.6 kJ/g; rising to 9.6 kJ/g at 90% dw; while brown coal has a calorific value of 8.65 kJ/g (Haberkern *et al.* 2008). After mechanical dewatering, sludge has a solid content of approximately 20-30% dw. Drying of sludge to higher solids concentrations requires considerable amounts of energy to remove capillary water and water bound to particle's surface. This is also reflected by the high energy demand of thermal drying, compared to mechanical means to remove water. For example, to dry 1 t dw of sludge from 30 to 90% dw, approximately 2 t of water need to be removed. On a theoretical base, to heat 1 l of water by 90°C and then transfer it to vapor, energy consumption is:

4.19 kJ/kg*K * 90 K + 2261 kJ/kg = 2638 kJ/kg (or 0.7 kWh/kg; Haberkern *et al.* 2008).

The minimum energy requirement for drying of 1 t dw of sludge from 30 to 90% dw is 1400 kWh of thermal energy. Due to process related energy losses, thermal energy consumption is higher in reality and additional electricity resp. mechanical energy is needed for processing e.g. transferring and mixing of sludge. Ideally, thermal energy for drying should be provided by renewable sources (solar) or by waste heat from combustion processes. Incineration process also requires electricity, mainly to treat flue gas to limits required by German law[16] and for transferring sludge.

The energy demand for drying and the energy generation from incineration, reported in various literature sources varies considerably. Haberkern *et al.* (2008) report per person heat and electricity demand for drying of sludge from 30% to 90% dw with 5 $kWh_{el}$/p*a and 45 $kWh_{therm}$/p*a. Assuming a sludge generation of 17 kg (dw)/p*a, energy demand is 0.3 $kWh_{el}$/kg dw and 2.7 $kWh_{therm}$/kg dw. Demand for thermal energy is almost twice the theoretical minimum. The authors do not report energy demand for incineration, but they report the calorific value of sludge with 90% dw with 2.7 kWh/kg.

Hong *et al.* (2008) report energy demand for drying with 0.2 $kWh_{el}$/kg dw and 1.6 $kWh_{therm}$/kg dw. Demand for thermal energy is close to the theoretical minimum. Incineration (fluidized bed) requires 0.3 $kWh_{el}$/kg dw and generates 0.9 $kWh_{el}$/kg dw. Assuming 17 kg (dw)/p*a sludge generation, energy surplus is 6.8 $kWh_{el}$/p*a. The authors do not report heat generation during incineration, but with 35% electrical efficiency and 60% thermal efficiency, there are 1.8 $kWh_{therm}$/kg dw available and a neutral balance of thermal energy is possible.

For drying and incineration taken together, Houillon *et al.* (2005) reports 0.4 $kWh_{el}$/kg dw for energy demand and 0.65 $kWh_{therm}$/kg dw for additional external supply of natural gas. Energy recovery is 6 MJ/kg dw. Assuming 30% electrical efficiency and 60% thermal efficiency, 0.6 $kWh_{el}$/kg dw and 1 $kWh_{therm}$/kg dw are recovered. Total demand of thermal energy including co generation and external supply, is 1.65 $kWh_{therm}$/kg dw. Demand for thermal energy is close to the theoretical minimum. Assuming 17 kg (dw)/p*a sludge generation, energy surplus from incineration is 2.9 $kWh_{el}$/p*a and additionally required external thermal energy is 7.2 $kWh_{therm}$/p*a.

Stillwell *et al.* (2010) report a net electricity generation of 0.9-1.3 $kWh_{el}$/kg dw with a neutral balance of thermal energy for drying and incineration taken together. Assuming 17 kg (dw)/p*a sludge generation, energy surplus from incineration is 15-23 $kWh_{el}$/p*a.

In a study from Germany from 2006, energy surplus over drying and incineration is estimated with 0.7 $kWh_{therm}$/kg dw resp. 12 $kWh_{therm}$/p*a and no surplus electricity (MUNLV 2001).

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[16] 17. BImSchV, see http://www.gesetze-im-internet.de/bundesrecht/bimschv_17/gesamt.pdf

Lacking statistical information on the energy demand for drying and energy generation from incineration in Germany, the following assumptions are made for the model:

For simplicity, it is assumed that 50% of total sludge incinerated is processed via mono incineration and 50% via co incineration in coal fired power plants, neglecting the 14% of sludge with other uses.

For drying of sludge, an energy demand with 5 $kWh_{el}$/p*a and 45 $kWh_{therm}$/p*a is assumed. In case of co incineration, thermal energy is fully supplied by waste heat from coal fired power plants. For energy generation from sludge, the calorific value of sludge is assumed with 2.7 kWh/kg at 90% dw. With 35% electrical efficiency, 16 $kWh_{el}$/p*a and 22 $kWh_{therm}$/p*a are generated.

For mono incineration, older facilities with a thermal only use of sludge, as reported by MUNLV (2001) and newer facilities with electricity generation have to be considered. Lacking detailed information on the distribution, it is estimated that 25% of sludge is treated in older mono incineration facilities and 25% in newer facilities. The older facilities use incineration for thermal energy only, generating 44 $kWh_{therm}$/p*a, but no electricity. The newer plants generate 16 $kWh_{el}$/p*a and 22 $kWh_{therm}$/p*a. To supply enough thermal energy for drying, they require an external supply of 23 $kWh_{therm}$/p*a. For mono incineration, flue gas requires treatment to meet the limits required by German law. The energy demand is estimated with 5 $kWh_{el}$/p*a. This is the same percentage (30% of the electricity generated) as reported by (Hong *et al.* 2008).

The weighted average for incineration (50% co incineration in coal fired power plants, 25% mono incineration in older and 25% in newer facilities) is consumption of 5 $kWh_{el}$/p*a and 44 $kWh_{therm}$/p*a for drying and generation of 12 $kWh_{el}$/p*a and 32 $kWh_{therm}$/p*a during incineration. Input of waste heat from coal fired power plants is 8 $kWh_{therm}$/p*a, and consumption of electricity for flue gas cleaning is 2.5 $kWh_{el}$/p*a.

For the best available technology case it is assumed that old facilities for mono incineration were replaced by newer facilities and the electricity generation from sludge incineration increases to 16 $kWh_{el}$/p*a.

There is no information available for transport distances of sludge to incineration facilities in Germany. Distances to coal fired power plants can be assumed to be further than for land use of sludge or mono incineration. To give an envelope for energy demand of sludge transport, the energy consumption for an average transport distance of 20 km (MUNLV 2001) as lower and 600 km (Haberkern, 2008) as higher value is reported.

### 2.4.3 Substance flow analysis

#### *2.4.3.1 Overview of partitioning factors for wastewater treatment plants*

The calculated overall partitioning factors from inlet (input to WWTP) to sludge, effluent and air (output from WWTP) for advanced and medium case WWTP with anaerobic sludge treatment are shown in Table 2-7. The sludge generated at WWTP is subsequently incinerated (advanced case) or applied to land (medium case). For the basic case: WWTP with aerobic sludge treatment, the calculated overall partitioning factors are shown in Table 2-8. The sludge generated at WWTP is subsequently applied to land.

The derivation of the partitioning factors is described in the following subchapters. The mathematical operations underlying the SFA are described in chapter 2.4.4.

Table 2-7: Status quo: WWTP partitioning factors for advanced and medium case

Rounded overall partitioning factors for advanced and medium case WWTP (case 1 and 2) including back load cycles. 1 = total load received by WWTP.

| | sludge | effluent | air (BWT) | air (biogas) |
|---|---|---|---|---|
| C | 0.29 | 0.05 | 0.37 | 0.29 |
| N | 0.19 | 0.19 | 0.54 | 0.07 |
| P | 0.90 | 0.10 | 0 | 0 |

Table 2-8: Status quo: WWTP partitioning factors for basic case

Rounded overall partitioning factors for basic case WWTP (case 3). 1 = total load received by WWTP.

| | sludge | effluent | air |
|---|---|---|---|
| C | 0.43 | 0.07 | 0.5 |
| N | 0.25 | 0.25 | 0.5 |
| P | 0.85 | 0.15 | 0 |

### 2.4.3.2 System description

The typical WWTP with anaerobic sludge stabilization (advanced and medium case) employs mechanical treatment (sedimentation), followed by biological treatment (activated sludge with simultaneous nutrient elimination) to reduce loads of dissolved organics (measured as chemical oxygen demand COD) and nutrients. Sewage sludge from mechanical (primary sludge PS) and biological (secondary sludge SS) treatment is anaerobically digested to generate biogas. Harvested biogas is used for co generation of electricity and heat on site in a combined heat and power plant. Stabilized sludge is dewatered to a solid content of 20-30%. Nutrient rich sludge water (backload) is recycled to biological treatment stage or is treated separately. Stabilized sludge from anaerobic digestion is transported off site for incineration (case 1) or for land use (case 2).

For the basic case (case 3), representing a WWTP with simultaneous aerobic sludge treatment, the technical setup underlying the SFA is less complex, with no backloads or biogas generation to consider. Sludge is stabilized simultaneously during aerobic treatment, then dewatered, stored and transported off site for land use.

The total elimination efficiencies for COD, N and P from effluent average 95% for chemical oxygen demand (COD), 81% for N and 91% for P, according to benchmarking studies at German WWTPs (DWA 2011). Elimination efficiencies are lower for small WWTPs (size class 1 and 2 with less than 10 000 p.e.), but the average in Germany is dominated by the large WWTPs.

Table 2-9: Status quo: Elimination efficiencies in % (DWA 2011)

| **Elimination efficiencies [%]** | **COD** | **N** | **P** |
|---|---|---|---|
| size class 1 | 91.5 | 71.3 | 70 |
| size class 2 | 92.9 | 76.7 | 70 |
| size class 3 | 94.5 | 82.7 | 80 |
| size class 4 | 95 | 81.2 | 91 |
| size class 5 | 94.2 | 80.1 | 94 |
| German average | 94.5 | 80.5 | 90.4 |

#### 2.4.3.3 *Primary treatment or sedimentation*

The first step in the model WWTP is primary treatment or sedimentation which removes settleable solids. Resulting primary sludge is transferred to anaerobic digestion (AD). Typical hydraulic retention times (HRT) are 1-2 h. Primary treated wastewater is fed to activated sludge treatment (AS). For the model WWTP the following partitioning factors are assumed, based on (DWA, 2010):

- 67% of C remaining in wastewater, 33% eliminated with primary sludge
- 90% of N remaining in wastewater, 10% eliminated with primary sludge
- 89% of P remaining in wastewater, 11% eliminated with primary sludge

#### 2.4.3.4 *Biological wastewater treatment*

The second step in our model WWTP is biological wastewater treatment or activated sludge treatment (AS) with nutrient elimination. The most common process in Germany, activated sludge treatment with nitrification- denitrification and chemical precipitation of P, is briefly explained below (DWA 2008; DWA 2007).

Activated sludge treatment (AS) relies on a community of mainly heterotrophic aerobic or facultative microorganisms to metabolize dissolved organic carbon (DOC), while releasing $CO_2$. To facilitate metabolism, wastewater is aerated. Resulting biomass, called secondary sludge (SS), is recirculated or fed to AD. Recirculation of activated sludge selects for micro-organisms that grow well in wastewater and easily form bioflocs that settle in a clarifier. After treated wastewater is clarified, it is discharged to the aquatic environment, mainly to rivers.

During AS, heterotrophic microorganisms use chemical energy stored in organic substances by oxidation with $O_2$, thereby releasing $CO_2$ (catabolism). Also, they incorporate C from organic substances in their biomass (anabolism). For C (measured as COD), typical elimination efficiencies (effluent *vs.* influent) are 90-95% (DWA 2011). Approximately 50% of C fed to AS is transferred to air as $CO_2$ (Ekama 2009), the remaining C is incorporated in biomass (SS).

The removal of N from wastewater often proceeds via simultaneous nitrification-denitrification. Nitrification is the biological oxidation of nitrogen from ammonia ($NH_4^+$) to nitrate ($NO_3^-$). It is followed by denitrification, the reduction of nitrate to nitrogen gas ($N_2$). Nitrogen gas is released to the atmosphere. Denitrification requires anoxic conditions. For N (as $N_{org}$, $NH_4^+$ and $NO_3^-$), typical elimination efficiencies (effluent *vs.* influent) are 80% and approximately 20% of incoming N remains in effluent (DWA 2011). Remaining N is transferred to air as $N_2$, and to a lesser extent incorporated in biomass (SS) (Ekama 2009).

For P (as $P_{org}$, $PO_4^{3-}$), typical elimination efficiencies (effluent *vs.* influent) are 90% (DWA 2011). P is removed from wastewater during microbial growth and incorporated in biomass (SS). Additionally, precipitation with iron or alum based flocculants is often employed. Precipitation mainly occurs on flocs of biomass and precipitated P is transferred to SS. For P, there is no environmentally important gas phase.

For the model WWTP, the following simplified partitioning factors over the whole described process are assumed:

- 50% of C are transferred to air, 7% remain in effluent and 43% are transferred to sludge (i.e. incorporated in biomass)
- 49% of N are transferred to air, 18% remain in effluent and 33% are transferred to sludge (i.e. incorporated in biomass)
- 10% of P remain in effluent and 90% are transferred to sludge (i.e. incorporated in biomass), no P is transferred to air, as there is no environmentally important gas phase

#### *2.4.3.5 Anaerobic sludge stabilization*

Case 1 and 2 employ anaerobic digestion to stabilize sludge (PS and SS). The process is briefly explained below (DWA 2008; DWA 2007, Bischofsberger *et al.* 2005).Under anaerobic conditions, methanogens produce biogas, a mixture of $CH_4$ and $CO_2$ from organic substrates. Substrates cannot be used completely, typical anaerobic digestibilities of sewage sludge range from 35% (SS) to 55% (PS). Unused organic substrates, as well as inorganic compounds, remain in stabilized sludge. After digestion, stabilized sludge is dewatered to a typical solids content of 20-30%. Dewatering releases sludge water (process water), which is typically rich in nutrients, especially N. During anaerobic digestion and dewatering, most N is transferred to sludge water ($N_{org}$, $NH_4^+$), remaining N is transferred to gas phase ($NH_3$) or remains in stabilized sludge. In contrast, most P remains in stabilized sludge, only a small part is transferred to sludge water. To recap, there is no environmentally important gas phase for P.

Partitioning factors for anaerobic digestion and especially loads to sludge water are very variable, as they depend on technical characteristics and process parameters, such as pH and

Temperature. Therefore, they are subject to high uncertainties. For the model WWTP, the following simplified partitioning factors over the whole described process are assumed:

- C is transferred to biogas as $CH_4$ and $CO_2$ according to anaerobic digestibilities of 55% for PS and 35% for SS
- 10% of C in PS and SS remains undigested in water phase (sludge water), rest of C remains undigested in stabilized sludge
- 25% of incoming N is transferred to biogas as $NH_3$, 45% to water phase, remaining 30% to stabilized sludge (PS and SS)
- 90% of incoming P remains in stabilized sludge, 10% are transferred to water phase (PS and SS)

#### *2.4.3.6 Sludge water treatment*

Stabilized sludge is dewatered to a solid content of 20-30% dw and nutrient rich sludge water (backload) is recycled to biological treatment stage or is treated separately. Backload from nutrient rich sludge water represents a considerable additional load (especially of N) and generates additional secondary sludge. Additional secondary sludge is recycled back to anaerobic treatment. For the model, it is assumed for simplicity that all sludge water is treated in the main stream. Back load cycles are calculated until recovery rates of >99% (output *vs.* input) are reached (3 cycles). Including backload, 65% of incoming C is transferred to anaerobic digestion and 29% of incoming C is transferred to biogas.

#### *2.4.3.7 Sludge incineration*

After dewatering, sludge is usually transported to sites for end use: incineration (case 1) or land use (case 2 and 3). For incineration (case 1), it is assumed that C and N in sludge is fully transferred to air and P fully remains in the ashes (solid waste product).

#### 2.4.3.8 *Land use of sludge and plant availability*

For case 2 and 3 of the model, sludge is transferred to land. 66% of the sludge is used in agricultural applications, the remaining in landscaping and other applications. The average plant availability of nutrients in sludge is assumed with 61% for N and 70% for P, according to (Bengtsson *et al.* 1997, Houillon and Jolliet 2005). To calculate energy credits for substitution of fertilizer, load of N and P in sludge applied to agricultural land is multiplied with the derived TEP factor and corrected for plant availability. For the purpose of this study C in sludge is considered as energetic loss. From perspective of ecosystems, organic C fuels the metabolism of soil communities, but it is lost for direct energetic reuse in form of electricity generation. C in sludge applied to agricultural land will eventually be biodegraded and released to air as $CO_2$. Due to the nature of the organic substrate with many cell wall components and residual compounds, stabilized sludge is not easily biodegradable. Rosso and Stenstrom, (2008) estimate that 2 to 3 years are necessary to biodegrade stabilized sludge applied to agricultural land.

#### 2.4.3.9 *Basic case*

For case 3 of the model, the WWTP with simultaneous aerobic sludge treatment slightly lower elimination efficiencies than for the larger facilities (case 1 and 2) can assumed: 93% for C, 75% for N and 85% for P (DWA 2011).

For a similar facility with a high sludge age of 30 days, (Ekama 2009) reports that 60% of C are transferred to air, 7% remain in effluent and 33% are transferred to sludge (i.e. incorporated in biomass). For N, the authors report that 60% of N is transferred to air, 13% remain in effluent and 27% are transferred to sludge (i.e. incorporated in biomass).

For the purpose of this study, these partitioning factors are adapted to typical elimination efficiencies in Germany and a shorter sludge age by shifting load from air to sludge for C *resp.* to sludge and effluent for N. For P, it can be assumed that 15% remain in effluent, and rest is transferred to sludge.

The following overall partitioning factors from inlet of WWTP (input) to sludge, effluent and air (output) are assumed for case 3 of the model:

- 50% of C are transferred to air, 7% remain in effluent and 43% are transferred to sludge (i.e. incorporated in biomass)

- 50% of N are transferred to air, 25% remain in effluent and 25% are transferred to sludge (i.e. incorporated in biomass)
- 15% of P remain in effluent and 85% are transferred to sludge (i.e. incorporated in biomass)

### 2.4.4 Mathematical operations

This chapter describes the mathematical operations underlying the model. Figure 2-3 gives an overview of the flows for a WWTP with anaerobic sludge stabilization (advanced and medium case). The model includes nine flows (termed F1-F9) and employs three modules:

- sedimentation [Sed]
- biological treatment: activated sludge with simultaneous nutrient elimination [AS]
- anaerobic digestion [AD]

From anaerobic digestion [AD], the nutrient rich sludge water is recycled to biological treatment stage [AS], sludge from biological treatment stage (secondary sludge SS) is recycled to anaerobic digestion [AD] (backload cycle).

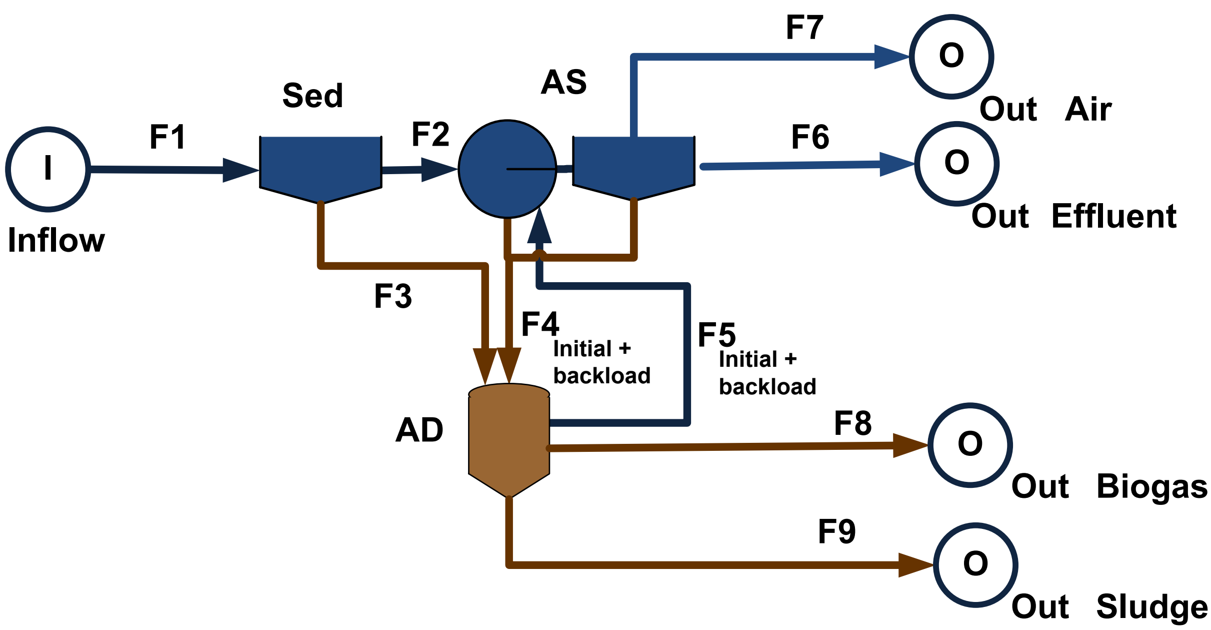


Figure 2-3: Status quo: Overview of SFA model

As the WWTP is a steady state equilibrium, this backload cycle between AD and AS (flows F4 and F5) has to be calculated by the model. The calculation is stopped when more than 99% of the input is transferred to the output (3 cycles for the WWTP without algae).

While primary sludge PS (F3) and secondary sludge SS (F4) are digested together, the anaerobic digestibilities differ, Therefore there are two modules. [AD1] for primary sludge PS with a higher anaerobic digestibility and [AD2] for secondary sludge SS. The calculation steps for the flows F1-F9 are reported in Table 2-10.

Table 2-10: Status quo: Calculation of flows for the SFA

| Calculation | Remarks |
|---|---|
| F1 = F6 + F7 + F8 + F9 | Inflow = Outflow |
| F2 = F1 x Sed-W | Primary treated wastewater PTW |
| F3 = F1 x Sed-S | Primary sludge PS |
| F4 = F4i + F4b + F4(b+n) | Backload cycle to consider |
| F4i = F2 x AS-S | Secondary sludge: Initial flow |
| F5 = F5i + F5b + F5(b+n) | Backload cycle to consider |
| F5i = [F3 x AD1-W] + [F4i x AD2-W] | Sludge water: Initial flow (AD1 for primary sludge, AD2 for secondary sludge –F4) |
| F4b = F5i x AS-S | Backload cycle 1: Secondary sludge |
| F5b = F4b x AD2-W | Backload cycle 1: Sludge water |
| F4(b+n) = F5i x [AS-S]^n | Backload cycle n: Secondary sludge |
| F5(b+n) = F4b x [AD2-W]^n | Backload cycle n: Sludge water |
| F6 = [F2 x AS-W] + [F5 x AS-W] | Output: Effluent |
| F7 = [F2 x AS-A] + [F5 x AS-A] | Output: Air 1(AS) |
| F8 = [F3 x AD1-A] + [F4 x AD2-A] | Output: Air 2 (Biogas) |
| F9 = [F3 x AD1-S] + [F4 x AD2-S] | Output: Sludge |

The calculations presented above show the internal cycling of CNP and the output to air, water and sludge. The SFA is the basis for the emission balance. The emission balance is calculated from the overall partitioning factors to air, water and sludge. It is expressed as mass flow by multiplying the overall partitioning factors with the input load of CNP per person and year (I).

Table 2-11: Status quo: Calculation of the emission balance from SFA

| | **[WWTP without algae]** |
|---|---|
| Air | Em (A) = (F7 + F8) * I |
| Water | Em (W) = F6 * I |
| Sludge | Em (S) = F9* I |

## 2.5 Inventory for algae systems

### 2.5.1 System description

The proposed process design for integration of algae systems is shown Figure 3-8. Based on a WWTP with 20 000 p.e. and anaerobic sludge stabilization (top), several internal flows are rerouted to support algae cultivation.

Algae cultivation takes place in open raceway ponds supplied with $CO_2$, as described by previous studies (high rate algae ponds (Park and Craggs 2010, 2011a-c). $CO_2$ is delivered to cultivation systems from sources on site the WWTP: combustion gas from biogas based co generation and gaseous emission from biological wastewater treatment (BWT).

For $CO_2$ supply, three scenarios are distinguished. First scenario ("algae light") uses 60% of available $CO_2$ for algae cultivation, representing daytime use of $CO_2$ with only small storage capacities. The other two scenarios: "algae medium" (80% of available $CO_2$) and "algae full" (100% of available $CO_2$) exploit also $CO_2$ generated during the night time, when algae do not require $CO_2$. These scenarios require storage capacities for $CO_2$ in addition to capture and supply infrastructures.

In all scenarios, sludge water is completely diverted to algae systems for nutrient supply. Additional nutrients are supplied by primary treated wastewater (PTW). PTW is added according to the nutrients required to fully exploit the $CO_2$ available in the different scenarios, not according to required volume. It is assumed that any additional water demand of algae systems, which depends on precipitation, evaporation and hydraulic retention time, is met by recycling water after harvest.

The system boundaries for analysis include the processes for wastewater and sludge treatment, including biogas use on site (water and sludge pathway). For the WWTP with integrated algae systems, system boundaries additionally include the process steps cultivation, harvest and co-digestion of harvested biomass (algae pathway).

The WWTP and the WWTP with algae systems receive identical input: raw wastewater with a typical composition and a daily load of 36 g C, 11 g N, 2 g P and 250 l of water per person served (as reported in chapter 2.2). As in the assessment of the current situation of the urban water chain only wastewater from households is considered, no commercial or industrial wastewater.

### 2.5.2 Substance flow analysis

#### *2.5.2.1 Algae cultivation and harvest*

Algae cultivation takes place in open raceway ponds mixed with paddlewheels. The design is based on prior studies and technical feasibility is proven in pilot scale applications with wastewater as growth medium (Park and Craggs, 2011a, Lundquist *et al.,* 2010). Depth of cultivation ponds is 0.2-0.4 m. It was observed, that depths shallower than 0.3 m limit pond size, due to pond hydraulics and $CO_2$ out gassing and show large diel temperature fluctuations. Greater depths have the disadvantage of lower biomass concentrations and larger volumes of water that need to be handled, e.g. pumped to harvesting facilities, but improve temperature regime and $CO_2$ storage (Lundquist *et al.,* 2010).

Hydraulic retention time (HRT) is 3-6 days. $CO_2$ is provided from flue gas from on site use of biogas (Lundquist *et al.,* 2010; Kadam *et al.*, 2002) and from biological treatment of wastewater. $CO_2$ *resp*. flue gas is supplied by countercurrent sumps within the ponds. $CO_2$ delivery is controlled by pH, as described by Park and Craggs*,* 2011a and Lundquist *et al.,* 2010.

As base case performance of algae systems at WWTPs, the total biomass productivity is assumed with 18 g/m²*d during vegetation season. Compared to the literature, the total biomass productivity assumed in this study is in the lower range. Many other studies use 25-30 g/m²*d (Campbell *et al.,* 2011; Lardon *et al.,* 2009; Collet *et al.,* 2011; Lundquist *et al.,* 2010; Stephens *et al.,* 2010). This conservative assumption of biomass productivity is based on values reported from a pilot project in New Zealand with microalgae systems using sludge water from a WTP as growth medium (Park and Craggs*,* 2011a).

The cultivation and harvesting module of the SFA model is based on the following parameters:

- partitioning factor of CNP to *harvested* biomass
    - nutrient uptake efficiencies (CNP from growth medium, and C from flue gas)
    - stoichiometric composition of biomass
    - harvesting efficiency
- partitioning factor of CNP to air
    - unused $CO_2$ from flue gas
    - unused $N_2$ from flue gas
    - unused $CO_2$ from DOC in growth medium

- partitioning factor of CNP to effluent
  - *non-harvested* biomass in effluent (containing CNP from growth medium, and C from flue gas)
  - free nutrients in effluent (unused)

For the harvesting efficiency, 88% is assumed for the model. Values for harvesting efficiencies in the literature range from 65%-83% for settling cones, depending on the size of flocs (Park and Craggs, 2011) to 95% for underground continuous flow clarifiers (Lundquist *et al.,* 2010). The authors point out the large uncertainty associated with these values. Therefore the harvesting efficiency is included in the analysis of parameter variations (see chapter 3.2).

Nutrients (CNP) are supplied to algae systems via:

- growth medium: sludge water and PTW (C as dissolved organic carbon DOC, N, P)
- $CO_2$ from flue gas and from biological wastewater treatment

Nutrient uptake efficiencies are important parameters for algae systems at WWTPs, as they define the maximum amount of biomass that can grow with a given amount of nutrients available. Also, they influence effluent quality. Table 2-12 gives an overview of the nutrient uptake efficiencies assumed in this study, differentiating between uptake efficiencies of total biomass and uptake efficiencies of *harvested* biomass.

Table 2-12: Algae systems: Nutrient uptake efficiencies of biomass

Total nutrient uptake efficiencies and uptake efficiencies of *harvested* biomass (gm: growth medium, fg: flue gas). 1= load received by algae systems.

| | **C (gm)** | **C (fg)** | **N** | **P** |
|---|---|---|---|---|
| uptake efficiencies of *total* biomass | 0.50 | 0.75 | 0.75 | 0.80 |
| uptake efficiencies of *harvested* biomass | 0.43 | 0.64 | 0.64 | 0.68 |

For N, the nutrient uptake efficiency is assumed with 75%. This assumption is based on values reported from a pilot project in New Zealand (Park and Craggs, 2011a). The authors measured 59% reduction in total Nitrogen at a harvesting efficiency of 69% and gaseous losses of 5-9% (Park and Craggs, 2011a). Therefore, total N uptake by biomass (harvested and non harvested) would be approximately 72-78%. In laboratory studies, also higher uptake efficiencies of 95% were measured (Shi *et al., 2007*). For P uptake efficiencies, there is no data available from pilot projects. In laboratory studies, 90% were measured (Shi *et al., 2007*). In the model a 80% P uptake efficiency is assumed. As P is the limiting nutrient in many aquatic ecosystems, microalgae developed efficient mechanisms to assimilate available P. Also, luxury uptake of P has been described (Powell, 2008). Combined with the assumed harvesting efficiency of 88%, nutrient uptake efficiencies of *harvested* biomass are 64% for N and 68% for P.

C is supplied to algae systems by two ways, via flue gas as $CO_2$ and via growth medium (sludge water mixed with primary treated wastewater) as dissolved organic carbon (DOC). For C from $CO_2$, Lundquist *et al,* (2010) approximate uptake efficiencies with 75% for flue gas and 85-90% for pure $CO_2$, based on prior estimates (e.g. Benemann *et al.,* 1982; Weissman *et al.*, 1987). The uptake efficiency for $CO_2$ from flue gas is assumed with 75% for total biomass and 64% for *harvested* biomass. This means that 25% of C from flue gas is lost to atmosphere, while 9% is incorporated in non-harvested biomass and remains in effluent of microalgae systems.

Algae can use DOC in growth medium after transformation to $CO_2$ by oxidation with free oxygen from photosynthesis or by bio-oxidation by accompanying heterotrophic microorganisms, as well as during mixotrophic growth (Lundquist *et al.,* 2010, Bhatnagar *et al.,* 2011). It was observed in a pilot project in New Zealand with microalgae systems using sludge water from a WTP as growth medium, that heterotrophic biomass accounts for 15-55% of total biomass, depending on HRT (Park and Craggs, 2011a). $O_2$ for heterotrophic metabolism is supplied by algae (autotrophs), while $CO_2$ for autotrophic metabolism is supplied by heterotrophs. According to Lundquist *et a*l. (2010), 1.55 g $O_2$ is produced with every g of algae biomass, while 1.1 g $O_2$ is required to oxidize 1 g of BOD (biological oxygen demand, typical ratio BOD to COD is 2). Therefore, with 1 g of algae biomass produced, accompanying heterotrophic biomass removes approximately 1.4 g of BOD or 0.7 g of COD or 0.2 g of C (DOC).

It is assumed that over the described biochemical processes: mixotrophic growth of algae, growth of heterotrophic biomass with $CO_2$ production and growth of autotrophic biomass with $O_2$ production, 50% of C delivered to algae systems as DOC in growth medium is incorporated in total biomass. Combined with the harvesting efficiency of 88%, uptake efficiency of *harvestable* biomass is 44%. For air emissions, it is assumed that 43% of C delivered to algae systems as DOC in growth medium is transferred to air as $CO_2$. This accounts for unused amounts of $CO_2$ due to elimination efficiency (75-90%) and release of $CO_2$ during night time by heterotrophs and autotrophs. The rest of C delivered to algae systems as DOC in growth medium remains in effluent: 7% as non-metabolized DOC plus 6% as non-harvested biomass.

For the stoichiometric composition of biomass, 36.7% C, 6.1% N and 0.81% P on a mass base are assumed (Collet *et al.,* 2011). C and N content is based on experimental data (Ras *et al.*, 2011), while P content is calculated using C/P ratio of *Chlorella vulgaris* as proxy (Lardon *et al.*, 2009). This is in the medium range of algal biomass composition reported by Lundquist *et al.* 2010: 45 -50% C, 4-10% N, and 0.3 -1.2% P.

The reported nutrient uptake efficiencies are used to calculate the amount biomass generated with the nutrients provided in different scenarios (light/medium/full). Biomass generation is limited by stoichiometric composition i.e. by the limiting nutrient. The amount of primary treated wastewater delivered to algae systems is chosen to meet the nutrient demand for full exploitation of $CO_2$ supplied in different scenarios. Therefore, the limiting nutrient is C. The limiting nutrient sets the ceiling for biomass generation. It is assumed that surplus nutrients are not incorporated in biomass: there is no luxury uptake of N or P.

Emissions to air from microalgae systems include unused $CO_2$ from flue gas (25% of C supplied by flue gas), unused $N_2$ from flue gas (100%), unused $CO_2$ from DOC in growth medium (43% of C supplied by growth medium) and gaseous losses of N (as $NH_3$, $N_2$ or $NO_x$). According to Park *et al.,* 2011, gaseous losses of N can be kept low by feeding growth medium shortly after carbonization sump, when the pH is controlled by $CO_2$ addition. Park *et al.,* 2011 approximates losses with 5-9%, while Roesch *et al.,* 2011 and Lundquist *et al.*, 2010 use 5%. For the model, 9% gaseous losses of N from microalgae systems are assumed. It is noteworthy, that the air emissions of N are considerably lower than for activated sludge process.

Effluent loads of CNP include nutrients incorporated in non-harvested biomass remaining in effluent and unused amounts of nutrients. Non-harvested biomass contains CNP from growth medium and C from flue gas. 9% of C delivered to algae systems as $CO_2$ in flue gas are incorporated in non-harvested biomass and remain in effluent of algae systems. For C from growth medium (DOC), it is assumed that 93% are metabolized by the consortium (heterotrophic, mixotrophic and autotrophic microorganisms) and therefore 7% remain in effluent of algae systems. Additionally, 6% of C from growth medium (DOC) are incorporated in non-harvested biomass. In total, 13% of C delivered to algae systems as DOC remain in effluent of algae systems(7% as DOC/”free” nutrients, plus 6% incorporated in non-harvested biomass).

For N, it is assumed that 18% of N delivered to algae systems in growth medium remain in effluent of algae systems(7.5% ”free” nutrients, mainly $NO_3$- plus 10.5% incorporated in non-harvested biomass). For P it is assumed that 13% of P delivered to algae systems in growth medium remains in effluent of algae systems, mainly incorporated in non-harvested biomass. Effluent quality of algae systems is subject to high uncertainties. There are only few studies, which address the subject.

For the model WWTP with integrated algae systems, the following assumptions are made:

- simplified partitioning factors over the whole described process as shown in Table 2-13 for nutrients from growth medium and in Table 2-14 for nutrients from flue gas
- nutrients are incorporated in total biomass according to uptake efficiencies reported in Table 2-12 (giving the maximum possible uptake), limited by stoichiometric requirements (no luxury uptake of nutrients)
- non-harvested biomass remains in effluent and contributes to effluent loads (even though the nutrients are not “free” i.e. in ionic or dissolved organic form)
- harvested biomass is transferred to sludge, which is fed to AD
- nutrients from sludge water and flue gas from anaerobic digestion are recycled to algae systems

Table 2-13: Algae systems: Partitioning factors for growth medium

Partitioning factor to effluent includes the contribution of non-harvested biomass. 1= load received by algae systems from growth medium.

| Partitioning factors | sludge | wastewater | air |
|---|---|---|---|
| C | 0.44 | 0.13 | 0.43 |
| N | 0.66 | 0.25 | 0.09 |
| P | 0.70 | 0.30 | 0 |
| Water | 0.01 | 0.94 | 0.05 |

Table 2-14: Algae systems: Partitioning factors for flue gas

Partitioning factor to effluent includes the contribution of non-harvested biomass. 1= load received by algae systems by flue gas.

| Partitioning factors | sludge | wastewater | air |
|---|---|---|---|
| C | 0.66 | 0.09 | 0.25 |
| N | 0 | 0 | 1 |

### 2.5.2.2 *Anaerobic digestion of harvested biomass*

Harvested biomass is co digested with sewage sludge to produce biogas. Values for mono digestion of algae biomass in literature range from 40-60% (Clarens *et al.*, 2011, Collet *et al.,* 2011), to 66% (Sialve *et al.,* 2009) or 70-90% (Hernandez and Cordoa, 1993). Co digestion with sewage sludge potentially enhances digestibility of algae biomass (Lundquist *et al*., 2010; Sialve *et al.,* 2009; Samson and LeDuy 1983), due to reduced ammonia inhibition. Lundquist *et al.*, 2010 assumes anaerobic digestibility of harvested biomass with 70% as co substrate with PS. This value is adopted for the model. With 70%, the anaerobic digestibility of harvested biomass as co substrate is higher than for primary sludge (55%) and secondary sludge (35%).

Assuming 0.5 g C/l biogas at a typical $CH_4$ content of 65%, and 70% anaerobic digestibility of harvested biomass gives 1536 l biogas/kg C or 569 l biogas/kg volatile suspended solid (VSS) loaded to AD, or 370 l $CH_4$/kg VSS. A laboratory study (Samson and LeDuy 1983) reported 350 l $CH_4$/kg VSS for mono digestion of algae biomass and 700 l $CH_4$/kg VSS for co digestion of algae biomass with sewage sludge, but with a long sludge retention time (SRT). Clarens *et al.,* 2011 evaluated different studies and reports likeliest value with 490 l $CH_4$/kg VSS and a range from 180-990 l $CH_4$/kg VSS. The authors point out the large uncertainty associated with these values. Therefore, the anaerobic digestibility of harvested biomass is included in the analysis of parameter variations (see chapter 3.2).

For transfer of N to air, 25% is assumed. This is the same value as for PS and SS. Laboratory studies reported 16% (Samson and LeDuy 1983).

For transfer of CNP to sludge water, the same values as for PS and SS are assumed: 10% of C, 45% of N and 10% of P loaded to AD. These assumptions are conservative compared to values from literature. Laboratory studies reported N mineralization efficiency of algae biomass (*i.e.* transfer of $N_{org}$ to effluent of anaerobic digestion as $NH_4^+$) with 47-69% (Samson and LeDuy, 1983) and 68% (Ras *et al.*; 2011). For P, there is no experimental data available. In another SFA study, Roesch *et al.,* (2011), approximated P mineralization efficiency of algae biomass with 80%. Assumed partitioning factors for anaerobic digestion of biomass harvested from algae systems are shown in Table 2-15.

Table 2-15: Algae systems: Partitioning factors for anaerobic digestion

Partitioning factors for anaerobic digestion of biomass harvested from algae systems. 1= load received by anaerobic digestion via biomass harvested from algae systems.

| Partitioning factors | sludge | ww | air |
|---|---|---|---|
| C | 0.20 | 0.10 | 0.70 |
| N | 0.30 | 0.45 | 0.25 |
| P | 0.90 | 0.10 | 0 |
| Water | 0.10 | 0.90 | 0 |

### 2.5.3 External energy balance

For the WWTP with 20 000 p.e. that is used as example, the energy consumption is assumed with 55 $kWh_{el}$/p*a. This is higher than the german average reported by Haberkern *et al.* (2008) but typical for the considered size class.

The water pathway consumes 70% of the energy, especially for aeration and N elimination in BWT. 20% are consumed in the sludge pathway, especially for operation of digester and dewatering, and 10% for other uses. Energy demand of BWT depends mainly on volume of wastewater treated, on received loads of CNP and on C/N ratio. N backload in sludge water contributes considerably to energy demand (Haberkern *et al.* 2008). P load in contrast has a much lower impact on energy demand -unless post treatment is required to meet limit values- as P elimination requires no energy intensive aeration.

To assess the implications of integration of algae systems for energy consumption, changes in loading rates have to be considered. The changes in loading rates to different treatment steps are assessed by the SFA.

The integration of algae systems changes loading rates in water pathway, as sludge water is rerouted to algae systems together with a fraction of primary treated wastewater. Besides reducing volume and load, the diversion of N rich sludge water also favorably changes the C/N ratio in BWT. The reduction in N load to BWT compared to WWTP without algae (in %) is used as proxy for reduction in electricity consumption in the model. While the energy demand of the water pathway is reduced, additional loads increase energy demand of sludge pathway. Here, the increase in C load to anaerobic digestion (in % compared to WWTP without algae) is used as proxy.

The energy demand for algae cultivation and harvest is assumed with 70 $kWh_{el}$/ha*d, with a *total* biomass productivity of 18 g/m²*d during vegetation season (and 88% harvesting efficiency). The biomass productivity is based on results from a pilot project for microalgae systems running on wastewater (Park and Craggs, 2011a). This study reports no energy demand. Values in literature for energy demand and biomass productivity of algae systems include:

- 65 $kWh_{el}$/ha*d (30 g/m²*d, Campbell *et al.,* 2011)
- 50-70 $kWh_{el}$/ha*d (with 20-30 g/m²*d, Lardon *et al.,* 2009)
- 74 $kWh_{el}$/ha*d (with 25 g/m²*d, Collet *et al.,* 2011)
- 91 $kWh_{el}$/ha*d (with 25 g/m²*d, Lundquist *et al.,* 2010)
- 100 $kWh_{el}$/ha*d (with 20-50 g/m²*d, Stephens *et al.,* 2010)

Compared to the literature, the total biomass productivity assumed in this study is in the lower range and the energy demand is in the medium range. Therefore, energy demand per ha and total biomass productivity is included in the analysis of parameter variations.

The required area is calculated from the SFA results specifying the respective amount of biomass generated with the available nutrients CNP in the different scenarios. The reported energy consumption accounts for $CO_2$ supply, nutrient supply, mixing, harvesting and water recycling after harvest.

On the energy generation side, SFA results for biogas production are used to calculate bio-electricity generation. For biogas, a 65% methane content corresponding to a lower heating value of 6.5 kWh/m$^3$, and 35% electrical efficiency are assumed. There is no external supply of heat required for anaerobic digestion operation. Thermal energy is fully supplied by on site co-generation.

### 2.5.4 Mathematical operations

This chapter describes the mathematical operations underlying the SFA model. Compared to the WWTP without algae (see chapter 2.4.4), the flows for a WWTP with algae are more complex. In total, 13 flows (F1-F13) have to be considered (Figure 2-4). Algae systems receive four different inputs: rerouted primary treated wastewater (PTW, F2-2) plus sludge water (F4) as growth medium (green arrows in Figure 2-4) and recycled $CO_2$ from activated sludge process (F7) and combustion gas (F8) as $CO_2$ supply (purple arrows in Figure 2-4).

For the WWTP without algae, one backload cycle between AD and AS via the sludge water have to be considered. For the WWTP with algae, there are two backload cycles to consider: via sludge water (F4) and via combustion gas (F8) to harvested biomass (F10), back to sludge water and combustion gas.

To calculate the flow of harvested biomass (F10), two further aspects have to be considered. Firstly, the stoichiometric requirements have to be taken into account. The ratio of C:N:P of algae biomass in combination with the uptake efficiencies for CNP determines the ideal C:N:P ratio of nutrient supply. For C, the supply via $CO_2$ has to be added to the C supplied by growth medium. For the technical setup for integration of algae systems presented in this study, C is the limiting nutrient. N and P not assimilated due to the stoichiometric requirements remain in effluent of algae systems. They are termed unused nutrients (U). The amount of unused nutrients depends on the actual C:N:P ratio of nutrient supply in relationship to the ideal C:N:P ratio.

There are two algae modules distinguish between nutrients from growth medium and $CO_2$ supply. The module [AL1] is for the growth medium. The module [AL2] is for the $CO_2$ supply by combustion gas or by air emissions from AS. For the [AL1] module, the PF to biomass for N and P needs to be corrected by the unused nutrients (U). The amount of C in biomass is known from the calculation of F10. As C is the limiting nutrient, there is no correction for unused nutrients ($U_C = 0$). The flow of unused nutrients $U_N$ and $U_P$ can be calculated using the C/N ratio and the C/P of the ideal supply (Table 2-16).

Table 2-16: Algae systems: Calculation of unused nutrients

| | Ideal supply | Actual supply | U |
|---|---|---|---|
| **C** | $c_I = z$ | $c_A = z$ | 0 |
| **N** | $n_I = (z / a)$ | $n_A = (z / a) + U_N$ | $U_N = n_A - (z / a)$ |
| **P** | $p_I = (z / b)$ | $p_A = (z / b) + U_P$ | $U_P = p_A - (z / b)$ |

*Remarks:* The ideal supply of C is set as the actual supply, as C is the limiting nutrient. The ratio of C/N and C/P of the ideal supply is calculated:
a = C/N (ideal)
b = C/P (ideal)
Further it is known that there is surplus N and P. Thus the actual supply of N and P is higher than the ideal supply by a factor $U_N > 0$ and $U_P > 0$.
Substracting the ideal supply from the actual supply gives $U_N$ and $U_P$

Thus, N and P cannot be assimilated by the total biomass (F9) with the efficiencies reported by the PF of the module [AL1]. The unused N and P contribute to effluent load (F11). But also the CNP assimilated in biomass (F9) is not fully transferred to harvested biomass (F10). The harvesting efficiency (HE) has to be taken into account. The non harvested biomass remains in effluent (F11) and contributes to CNP load.

The module [AD3] describes the anaerobic digestion of harvested biomass (F10). Compared to the module [AD1] for primary sludge PS and [AD2] for secondary sludge SS, it has higher anaerobic digestibilities.

As the WWTP with algae is a steady state equilibrium, the described backload cycles (flows F4 and F8 plus F10) have to be calculated by the model. The calculation is stopped when more than 99% of the input is transferred to the output (6 cycles for the WWTP with algae).

The full load of CNP to effluent includes the effluent load of AS (reduced flows compared to WWTP without algae) and the effluent load of algae systems. The latter includes the nutrients not assimilated due to uptake efficiency plus those which are not assimilated due to C limitation (NP) and those incorporated in the non harvested biomass. It is noteworthy, that C from CO2 supply can contribute to the effluent load of algae systems via the non harvested biomass. The air emissions from the algae systems include: unused $CO_2$ from flue gas, unused $N_2$ from flue gas, unused $CO_2$ from DOC in growth medium and gaseous losses of N. The calculation steps for the flows F1-F13 are reported in Table 2-17.

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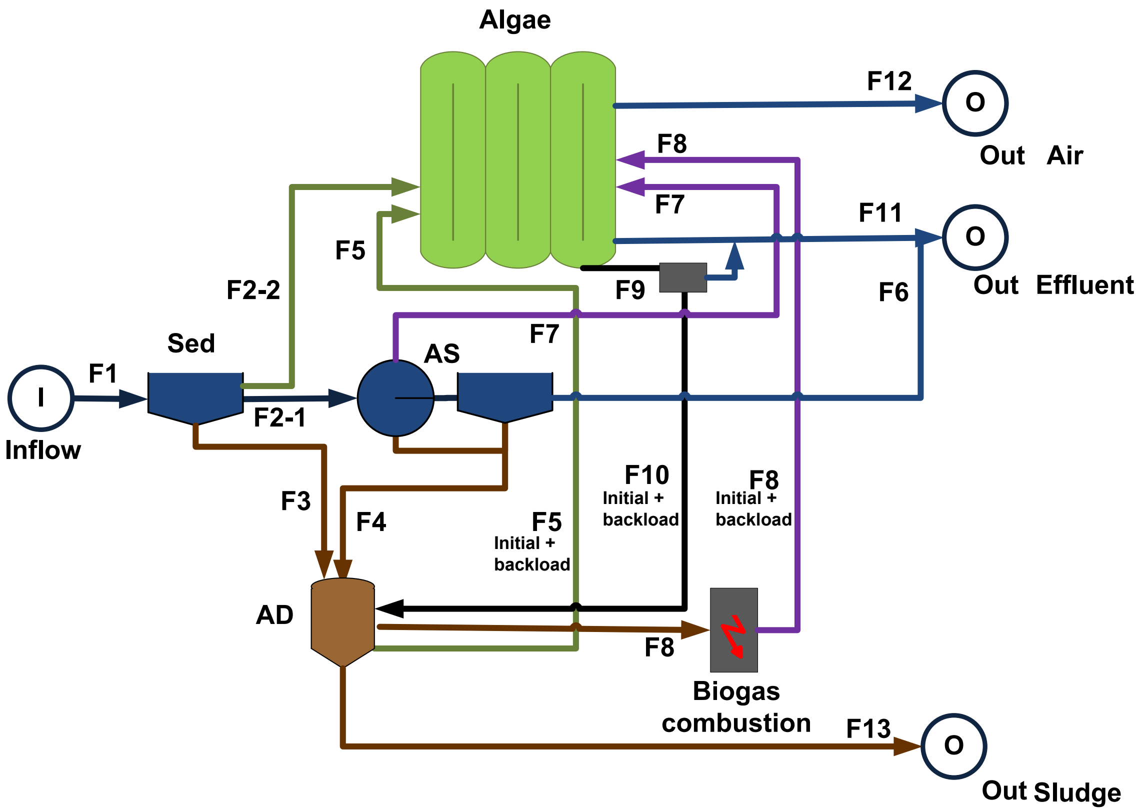


Figure 2-4: Algae systems: Overview of SFA model

Table 2-17: Algae systems: Calculation of substance flows for the SFA

| Calculation | Remarks |
|---|---|
| F1 = F6 + F9 + F12 + F13 | Inflow = Outflow |
| F2-1 = [F1 x Sed-W] * (1-x) | Primary treated wastewater PTW to Algae, with x <1 |
| F2-2 = [F1 x Sed-W] * (x) | PTW to AS → Δ in load to AS compared to WWTP without algae used for calculation of energy savings (E1) |
| F3 = F1 x Sed-S | Primary sludge PS |
| F4 = F2-1 x AS-S | Secondary sludge: no backload cycle to consider |
| F5 = F5i + F5b + F5(b+n) | Sludge water: Backload cycle to consider, input to algae systems |
| F5i = [F3 x AD1-W] + [F4i x AD2-W] | Sludge water: initial flow, AD1 for primary sludge –F3, AD2 for secondary sludge –F4 |
| F6 = [(F2-1) x AS-W] | **<u>Output:</u>** Effluent AS |
| F7 = [(F2-1) x AS-A] | Air emission from AS rerouted to Algae → no output |

Table 2-17 (continued): Algae systems: Calculation of substance flows for the SFA

| Calculation | Remarks |
|---|---|
| F8 = F8i + F8b + F8(b+n) | Combustion gas: Backload cycle to consider → F8 (biogas/combustion gas) for calculation of energy generation via biogas (E4) |
| F8i = [F3 x AD1-A] + [F4 x AD2-A] | Combustion gas: initial flow |
| **F9** = [F9i + F9b + F9(b+n)] | Total biomass: Backload cycle to consider |
| F9i = [[(F2-2 + F5i) x AL1-S/U] + [(F7 + F8i) x AL2-S]] | Total biomass: initial flow<br>U condition applies: stoichiometric requirements reduce PF for N and P for [AL1-S/U] |
| F10i = F9i * HE | Harvested biomass, HE harvesting efficiency |
| F5b = F9i x AD3-W | Sludge water: Backload cycle 1 |
| F8b = F9i x AD3-A | Combustion gas: Backload cycle 1 |
| F9b= [F5b x AL2-S/U] + [F8b x AL2-S] | Total biomass: Backload cycle 1; U condition applies |
| F10b = F9b * HE | Harvested biomass, HE harvesting efficiency |
| F5(b+n) = F9b x [AD3-W]^n | Sludge water: Backload cycle (b+n) |
| F8(b+n) = F9b x [AD3-A]^n | Combustion gas: Backload cycle (b+n); |
| F9(b+n) = [F5b x [AL2-S/U]^n] + [F8b x [AL2-S] ^n] | Total biomass: Backload cycle (b+n); U condition applies |
| F10(b+n) = F9(b+n) * HE | Harvested biomass: Backload cycle (b+n) |
| F11 = [(F2-2 + F5i) x AL1-W/U] + [F9 * (1-HE)] | **Output:** Effluent Algae<br>CNP not eliminated due to<br>uptake efficiency + C limitation (NP) + non harvested biomass |
| F12 = [(F2-2 + F5i) x AL1-A] + [(F7 + F8) x AL2-A] | **Output:** Air Algae<br>Air emission of CN from growth medium + $CO_2$-supply |
| F13 = [F3 x AD1-S] + [F4 x AD2-S] + [F10 x AD3] | **Output:** Sludge<br>CNP from primary sludge PS + secondary sludge SS + harvested biomass |

The calculations presented above show the internal cycling of CNP and the output to air, water and sludge. The SFA is the basis the emission balance. The emission balance is calculated from the overall partitioning factors to air, water and sludge. It is expressed as mass flow on a per person base, by multiplying the overall partitioning factors with the input load per person (I).

Table 2-18: Algae systems: Calculation of emission balance

| | [WWTP with algae] | [WWTP without algae] |
|---|---|---|
| Air | Em (A) = F12 *I | Em (A) = (F7 + F8) * I |
| Water | Em (W) = (F6 + F11) * I | Em (W) = F6 * I |
| Sludge | Em (S) = F13 * I | Em (S) = F9 * I |

For the WWTP with algae, the SFA is also the basis to assess the energy balance. The energy consumption and generation of the WWTP with algae is adapted to the altered internal flows of CNP by assigning factors E1-E4. These show:

- the reduced energy demand of AS (E1)
- the additional energy demand of AD (E2)
- the additional biogas for additional energy generation (E3)
- the additional sludge for additional energy generation (E4)

To derive the factors, the flows of the WWTP without algae (marked grey in Table 2-19) are compared to the flows of the WWTP without algae (*marked green*). For E1, the flow of N is used as a proxy, for the other factors, the flow of C is used as a proxy.

Table 2-19: Algae systems: Calculation of energy balance

| Calculation | Remarks |
|---|---|
| E1 = [ [F2 + F5] - *[F2-1]* ] / [F2 + F5] | **Reduced energy demand of AS**<br>*[energy consumption AS]* with algae = E1 * [energy consumption AS]<br>N as proxy, E1 < 1 → less N loaded to AS for the WWTP with algae |
| E2 = [ [F3+ F4] - *[F3 + F4 + F9]* ] / [F3+ F4] | **Additional energy demand of AD**<br>*[energy consumption AD]* with algae = E2 * [energy consumption AD] without algae<br>C as proxy, E2 > 1 → additional C loaded to AD for the WWTP with algae |
| E3 = [ [F8] - *[F8]* ] / [F8] | **Additional biogas**<br>*[energy generation biogas]* with algae = E3 * [energy generation biogas] without algae<br>C as proxy, E3 > 1 → additional C in biogas |
| E4 = [ [F9] - *[F13]* ] / [F9] | **Additional sludge for incineration**<br>*[energy generation sludge]* with algae = E4 * [energy generation sludge] without algae<br><br>*[energy consumption sludge handling]* with algae = E4 * [energy consumption sludge handling] without algae<br>C as proxy, E4 > 1 → additional C in sludge |

# 2.6 Inventory for model city

## 2.6.1 System description

The model city analyzed in this study has 20 000 inhabitants (corresponding to the size of the WWTP without and with algae systems analyzed in chapter 3.2). While model city is pictured in a rural setting with land resources available around the WWTP, gardening and urban agricultural activities (own production of food) are excluded from the present analysis. The focus of this analysis lies on the household consumption, and industrial or commercial activities are also excluded.

To extent the perspective, the flows representing the connection points of the urban water chain to the full urban metabolism are analyzed: the flows of energy, water, food and cleansing product related to daily household consumption. The input of water (chapter 2.6.2), energy (chapter 2.6.3), food (chapter 2.6.4) and cleansing products (chapter 2.6.6) to the household and the associated flows of CNP are quantified. The pathways of CNP are traced from the point where they enter the household until they leave the astysphere as emission to air, water and land/soil. This includes the transformations of CNP in the consumed food during human metabolism (chapter 2.6.5); on the post use side it includes organic waste treatment (chapter 2.6.8), in addition to the wastewater and sludge treatment (chapter 2.6.7). To discuss the problem of AMPs, PFOS is included as a model substance (chapter 2.6.9).

For the wastewater and sludge treatment (post use side of the urban water chain), four cases are assessed. Two of them include algae systems. All other analyzed flows of UM: energy, water, food and detergents consumption, human metabolism and organic waste disposal are assumed equal for these four cases.

## 2.6.2 Water consumption

For water use in households, the average values as described in chapter 2.2 are assumed: consumption of 41 $m^3/p*a$, with an associated energy consumption of 26 kWh/p*a for the supply side of the urban water chain.

### 2.6.3 Energy consumption

Energy consumption in households includes electricity and thermal energy. Electricity consumption in households in model town is assumed with 1300 kWh/p*a (Schmidt *et al.* 2011). This electricity is provided from German grid. It is produced from brown coal (24.6%), black coal (18.5%), nuclear energy (17.7%) and natural gas (13.6%). Other energy carriers, including oil account for 5.1%. Renewable energy sources contribute (20.3%) (year 2011, AGEB 2012).

The $CO_2$ emission factor for electricity is 566 g/kWh. This value refers to emission of $CO_2$ equivalents and includes the contribution of non C based GHG as for example $N_2O$. Without these contributions, $CO_2$ emissions (not $CO_2$ equ.) are 409 g/kWh (UBA 2010a). Converted to C using the molecular weights (factor 0.27 for $CO_2$ to C), this gives a C intensity of electricity production of 110 g C/kWh. This value is used to calculate the C flows associated with electricity production.

For consumption of thermal energy in households, an average value of 8000 kWh/p*a is assumed. This includes 1000 kWh/p*a for hot water preparation. Interestingly, most of the energy consumed for hot water preparation in households remains stored in wastewater as thermal energy. This provides an opportunity for reuse of thermal energy.

As a simplification, it is assumed that all thermal energy in model city is provided by natural gas. The emission factor is 0.2 kg/kWh [$CO_2$-equivalents] and the C intensity is 0.005 kg/kWh [C] (UBA 2010B probas). This simplification neglects other energy sources with a higher (e.g. oil or electricity, used by 30% of German population) or lower $CO_2$ emission factor (renewable heat sources e.g. solar thermal, used by 10% of German population). The average figures for energy consumption used in this study does not take into account that energy consumption varies considerably with household size, appliances used and user habits.

There are also N flows related to energy consumption, as energy carriers such as coal and natural gas contain a certain amount of N that is released during combustion mostly as $N_2$ (e.g. coal 3%, natural gas <1%). For this study, 0.5% of C content is assumed. Thus, the N flows N related to energy consumption amount to 2.9 kg/p*a. As the released $N_2$ adds to the non-reactive N pool in the atmosphere, its ecological relevance is low. But more ecologically relevant, the combustion processes may also transform this internal N or $N_2$ from air to $N_2O$. $N_2O$ has a high global warming potential (GWP) of 298 (IPCC 2006). The emission factors for $N_2O$ are 7.7 mg/kWh for electricity and 0.4 mg/kWh for thermal energy from natural gas (UBA 2010b).

### 2.6.4 Food consumption

For the purpose of this study, the magnitude of flows is exemplified for a semi hypothetical model city, based on statistics, scientific publications and own calculations. A full assessment of household food consumption requires an analysis on its own right, as there are large variations in consumption patterns i.e. between sexes, age, income, preferences and special diets. According to BMELV (2012), the average of the total food provision in Germany amounts to 682 kg/p*a (average 2005-2011). This list is adapted by grouping similar entries to simplified food groups.

Table 2-20 shows the simplified food groups and the average amounts per person and year (BMELV 2012).

This amount includes in house (household) and out of house consumption, as well as losses along the supply chain (food refused or wasted, excluding losses during agricultural production). In absence of data on the ratio of in house and out of house consumption, 100% is assigned to the household consumption for the purpose of this study. This simplification allows including the full food consumption per person. Also, journeys out of the boundaries of model city are neglected. These journeys basically represent a replacement of resource consumption and/or waste discharge to another urban area i.e. to a different sewer system and organic waste (ow) collection and treatment system.

The food provided per person is not equal to the actual food consumption. In a recent study, it was found that food waste amounts to 134 kg/p*a (Kranert *et al.* 2012). 52 kg/p*a of food waste are generated by commercial operators and 82 kg/p*a in households. In the household, losses include unavoidable losses during preparation: peels, inedible parts etc, and avoidable losses: food refused due to mismanagement, transgression of best-before-date or personal preferences. 65% of household food wastes are avoidable i.e. are attributed to the latter category (Kranert *et al.* 2012).

Using the presented data in conjunction gives the following picture of food flows in Germany: 682 kg/p*a of food are provided, 134 kg/p*a are transferred to organic waste and 548 kg/p*a are consumed by humans. Actual human consumption of food is 80% of the food provided. For the purpose of this study and in absence of more detailed data, it is assumed that food losses are distributed evenly over all food groups and also that the CNP content is reduced proportional to the CNP content of bulk food (see below).

The assessment of the content of CNP in the food flows is not straightforward: from the bulk consumption of food, to food losses to actual human consumption to content of CNP in consumed food, uncertainty becomes higher. For C content of food groups, 0.77 is assumed for fat and 0.43 for sugars (carbohydrates) (Baker *et al.* 2007). For the remaining food groups: animal and plant products, an average dry weight of 40% and a C content of 45% of dw are assumed. Based on these estimates, the food in model city contains 136 kg/p*a of C. This gives an average C content of 20% for the bulk food, which seems plausible.

This value is also in agreement with an estimate of C content of food from the wastewater side. For the dimensioning of treatment steps, an incoming load of 120 g/p*d of COD is assumed according to the technical standards (ATV 2000). It is estimated that 60% of COD originate from feces and urine (ultimately food consumption) and the remaining 40% from detergents (DWA 2008, DWA 2013). With a factor of 3 from C to COD, this amounts to 9 kg/p*a C transferred to wastewater from food consumption. Adding 100 kg/p*a for respiration losses (human metabolism, see chapter 2.6.5 below), gives 109 kg/p*a of C consumed via food. Including the 20% that are transferred to organic waste, gives 136 kg/p*a C in food.

N in food is almost exclusively incorporated in proteins (Fissore *et al.* 2011, González *et al.* 2011). Proteins have an average N content of 16%. Food groups high in protein include animal products, and some plant products, such as soy (leguminoses) or cereals. González *et al.* (2011) lists the protein content of several food groups (

Table 2-20, second column). Using these data in conjunction with the contribution of several food groups to total food provision in Germany (

Table 2-20, first column), gives a total of 6.7 kg/p*a of N in food (

Table 2-20, last column). Some food groups contribute over proportionally to the total N consumed with food (see also Figure 3-15).

With 20% transferred to organic waste (1.3 kg/p*a), the amount of N in food consumed by humans is 5.4 kg/p*a. If the losses during human metabolism are assumed with 10% (0%: Baker *et al.* 2007, Fissore *et al.* 2011; 10%: Villarroel Walker 2010, Antikainen 2007; see also chapter 2.6.5 below), a N load of 4.8 kg/p*a or 13.2 g/p*d can be expected from food consumption based on the “top down” calculation from the food side.

Table 2-20: Model city: Bulk food, protein content and amount of N in food

| | kg/p*a (bulk) | protein content [mg/g] | kg/p*a (N) |
|---|---|---|---|
| Fruits | 122 | 5 | 0.10 |
| Vegetables | 92 | 10 | 0.15 |
| Cereals | 91 | 100 | 1.45 |
| Rice | 5 | 66 | 0.05 |
| Legumes | 1 | 210 | 0.02 |
| Potatoes | 64 | 17 | 0.18 |
| Meat | 89 | 200 | 2.84 |
| Fish | 15 | 207 | 0.51 |
| Cheese | 15 | 249 | 0.60 |
| Milk products | 19 | 32 | 0.10 |
| Milk | 87 | 32 | 0.44 |
| Eggs | 13 | 126 | 0.26 |
| Fat (total) | 22 | | |
| Sugar | 49 | | |
| | | | |
| Sum | 682 | | 6.7 |

But calculating the amount of N in food "bottom up" with values reported from the wastewater side gives a different picture. For the dimensioning of treatment steps, an incoming load of 11 g/p*d of N is reported in the relevant norms (ATV 2000). It is estimated that 92% of N originate from feces and urine (ultimately food consumption) and the remaining 8% from other sources, e.g. detergents (DWA 2008, DWA 2013).This amounts to 3.7 kg of N transferred to wastewater from food consumption. This is considerably lower than the value from the "top down" calculation from the food side. As this discrepancy cannot be solved with the available data, 30% N losses during human metabolism are assumed for the model city to close the gap, acknowledging the uncertainty.

The P content of food in model city is assessed based on the N/P ratio. The N/P ratio of food is assumed with 9, which gives 0.75 kg/p*a of P in food. The N/P ratio of food reported in other studies is:

- 4.1-5.2 ( Villarroel Walker 2010)
- 5.9 (Antikainen 2007)
- 9.5 (Neset 2005)
- 10.5 (Baker *et al.* 2007, Fissore *et al.* 2011)

With the N/P ratio of 9, a 20% transfer rate of food to organic waste; and a 100% transfer rate from consumed food to wastewater; gives a load of 0.6 kg/p*a of P to wastewater in model city. This value is in agreement with an estimate of P content of food from the wastewater side. According to the relevant norms (ATV 2000), the load in wastewater is 0.75 kg/p*a. This includes 0.55 kg of P transferred to wastewater from food consumption (75%) and 0.2 kg from detergents (25%) (DWA 2008, DWA 2013).

### 2.6.5 Human metabolism

The food consumed by humans provides nutrients for human metabolism: for energy production (heat and chemical energy) and for "biomass growth" e.g. cell regeneration, protein building and other fundamental functions. As in all heterotrophic organisms, C from food is oxidized to $CO_2$, fueling the energy metabolism (catabolism). While anabolic processes run continuously, the human body can be considered as steady-state equilibrium. All CNP taken up is released to the environment (Villarroel Walker 2010, Baker *et al.* 2007, Fissore *et al.* 2011). Exemptions are phases of net biomass generation: growth in young people, pregnancy or lactation. The temporary stock of CNP in human biomass will ultimately be released upon death and decay. During lifetime, CNP is released to the environment via breath, via feces and urine or via sweat and losses of skin cells, hairs etc.

For this study, human respiration is estimated with 100 kg C/p*a (366 kg $CO_2$/p*a) in average (Villarroel Walker 2010, Baker *et al.* 2007, Fissore *et al.* 2011, Prairie and Duarte 2007). These 100 kg represent 92% of the C consumed with food. The remaining 8% are assumed to be transferred to wastewater via feces and urine. For N consumed with food, 30% losses are assumed, during human metabolism (see chapter 2.6.4). Other studies report higher transfer rates for N from consumed food to wastewater (90%: Villarroel Walker 2010; 100%: Baker *et al.* 2007, Fissore *et al.* 2011). For P, 100% transfer to feces and urine is assumed (Baker *et al.* 2007, Fissore *et al.* 2011, Villarroel Walker 2010).

### 2.6.6 Detergents

Cleansing products for house and body are consumed on a daily base in households. According to UBA (2012), private end users consume 1.3 million tons of detergents, including: 630 000 t of cloth washing products, 220 000 t of fabric softeners, 260 000 t of dish washing products and 480 000 t cleansing products for body and hair. The use of these products transfers a total of 630 000 t of active ingredients (7.7 kg/p*a), including 31 860 t of Phosphates to wastewater (UBA 2008).

Assuming a P content of 0.33, based on molecular weights, gives an annual load of 0.13 kg P/p*a. In addition, approximately 4 000 t of phosponates are transferred to ww. Phosponates are organic substances containing C-PO$(OH)_2$ or C-PO$(OR)_2$ groups, which are used to chelate metal ions which can disturb bleaching processes (UBA 2008). They are only slowly degradable in natural waters. Assuming an average P content of phosponates of 0.2, gives an additional load of 0.009 kg/p*a. Based on these estimates, the load of P in detergents for model city is assumed with 0.13 kg/p*a.

This is value is in good agreement with an estimate of P load from detergents from the wastewater side, albeit. It is estimated that 25% of the 2 g/p*d P load in wastewater, originate from detergents (DWA 2008). This adds up to an annual load of 0.18 kg of P per person from detergents. This value is slightly higher than calculated from the data from UBA (2008). For model city, the values based on the data from UBA (2008) is used (0.14 kg/p*a).

To calculate the C content of detergents, the C content of active ingredients (7.7 kg/p*a, UBA 2012) is estimated with 0.75. The resulting 5.8 kg C/p*a are in good agreement with the values given by other sources (DWA 2008, DWA 2013). From the wastewater side, the reported annual load from detergents is 5.7 kg of C per person[17].

For N content of detergents, it is estimated that from the 11 g/p*d in wastewater, 9% originate from detergents (DWA 2008, 2013). This adds up to an annual load of 0.4 kg of N per person from detergents. This value was adopted for the present study.

Detergents can also be transferred to waste e.g. left-over product in discarded packaging etc. This pathway was neglected in the present study.

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[17] 116 g/p*d of COD in wastewater, 40% from detergents (DWA 2008, 2013), factor 3 for C to COD (Henze 2000)

### 2.6.7 Wastewater and sludge treatment

For the model city, a mixed sewer system is assumed with 91 $m^3/p*a$ of wastewater generated in average, including rainwater and sewer infiltration. The electricity demand for wastewater transport is 5.5 $kWh_{el}/p*a$. Four cases for wastewater and sludge treatment are assessed; two of these cases include algae systems.

The first case “basic urban water chain” (Model city MC1) corresponds to the basic case as analyzed in chapter 3.1, without anaerobic digestion and with agricultural reuse of sludge which represents 25% of German population. It has a WWTP with simultaneous aerobic sludge stabilization followed by land use of sludge. The energy demand of the WWTP is assumed with 37 kWh/p*a. This is slightly lower than the average used for case 3 in chapter 3.1 (40 kWh/p*a), as the WWTP in the model city with 20 000 p.e. represents a larger plant of this category which mostly contains small plants with less than 10 000 population equivalents. In terms of energetic recycling, this represents a worst case scenario. For transport of sludge, short distances are assumed (20 km). Due to the minor contribution to the energy balance with low distances, the transport is neglected.

The second case “best available technology” (MC2) refers to an optimistic scenario with anaerobic digestion and sludge incineration. The energy balance for the best available technology case is well above the German average due to minimized energy losses and maximized electricity production (see best available technology case in chapter 3.1). It represents a best case scenario employing current technology. Compared to case 1 and 2, which represent mostly larger WWTPs, the energy demand is slightly increased (37 kWh/p*a *vs.* 33 kWh/p*a for case 1 and 2).

The third case “algae” (MC3) is based on the “best available technology” case and employs algae systems with full $CO_2$ recycling as described in chapter 3.2 (without post treatment). The required area is 6 $m^2/p$, totaling 12 ha around the WWTP. For the fourth case “algae +” (MC4), the algae systems receive additional $CO_2$ and the cultivation area can be extended to 9 $m^2/p$ (18 ha). For the „algae +” case, $CO_2$ from the composting facility is transferred to the algae systems during growing season (ca. 4 kg C). This additional source of $CO_2$ allows full exploitation of N and P available in wastewater. With the additional $CO_2$, post treatment is mandatory to achieve limit values for effluent (see chapter 3.2). Post treatment reduces the load of CNP in effluent, as well as the load of AMPs. The eliminated fraction is transferred to sludge. For the algae systems a vegetation season of 250 days is assumed. Table 2-21 shows the resulting energy balances.

Table 2-21: Model city: Electricity consumption and generation at WWTP

| WWTP | MC1 | MC2 | MC3 | MC4 |
|---|---|---|---|---|
| Energy use water pathway | -35 | -26 | -15 | -9 |
| Energy use sludge pathway | | -7 | -13 | -16 |
| Energy use others | -4 | -4 | -4 | -4 |
| Energy use algae pathway | | 0 | -9 | -14 |
| Energy use post treatment | | | | -5 |
| Energy generation sludge (PS) | | 10 | 12 | 12 |
| Energy generation sludge (SS) | | 6 | 4 | 2 |
| Energy generation algae | | 0 | 26 | 42 |
| | | | | |
| net consumption | -6 | 4 | 4 | 5 |
| brut consumption | -6 | -13 | -14 | -18 |
| own generation | 0 | 16 | 19 | 23 |

Compared to the algae full scenario in chapter 3.2, the algae scenarios used for model city have a better energetic performance. While the parameters for the algae systems were adopted from chapter 3.2, the WWTP processes are assumed to be more efficient. The WWTP processes are based on the best available technology case, which represents a best practice example for WWTPs of this size class. Therefore, energy consumption for BWT and anaerobic digestion are reduced compared to chapter 3.2, while the energy consumption of the algae systems and the energy generation from algae biomass (ABM) is the same.

For sludge treatment in model city, a modern mono incineration facility, as described in chapter 3.1 is assumed for all cases except for the basic case with land use of sludge. The transport is neglected. As the integration of algae systems slightly increases the amount of sludge, it also increases energy consumption for sludge treatment, as well as the energy generation (

Table 2-22).

Table 2-22: Model city: Electricity consumption and generation for sludge treatment

| | MC1 | MC2 | MC3 | MC4 |
|---|---|---|---|---|
| dewatering (EL) | -2,5 | -2,5 | -2,9 | -3,6 |
| storage (EL) | -4 | | 0 | 0 |
| drying (EL) | | -5 | -6 | -7 |
| incineration/flue gas cleaning (EL) | | -5 | -6 | -7 |
| incineration electricity gen. (EL) | | 16 | 19 | 23 |
| | | | | |
| drying (TE) | | 44 | 51 | 64 |
| incineration (TE) | | 29 | 33 | 42 |
| external te required (TE) | | 15 | 18 | 22 |
| transport (fuels PE) | 0,7 | 3,5 | 4,0 | 5,1 |
| | | | | |
| net consumption (EL) | -6 | 4 | 4 | 5 |
| brut consumption (EL) | -6 | -13 | -14 | -18 |
| own generation (EL) | 0 | 16 | 19 | 23 |

### 2.6.8 Organic waste treatment

Composting is the main process employed in Germany for organic waste treatment. 90% of organic waste is treated in composting facilities, only 10% in digestion facilities. For model city, a composting facility for organic waste treatment is assumed. As a particularity, this facility is in close distance to the WWTP to allow reuse of $CO_2$ emissions from composting in algae systems at the WWTP in the algae + case.

For the collection and transport to the facility, a distance of 20 km is assumed. For transport of 134 kg/p*a over this distance, the energy consumption (PE) is ~2 kWh/p*a (see chapter 2.4.2). The transport is neglected for the purpose of this study.

For the composting process, reported partitioning factors for losses to air are 30-60% for C and 10-40% for N (Villarroel Walker 2010, Hao *et al.*, 2004, Eghball *et al.*, 1997). For this study, 40% for C and 30% for N are used. Losses to water occur during leaching and depend on water content of composted materials and water input e.g. by precipitation. The organic waste is a relatively dry substrate for composting. The water content is approximately 70% (30% dw, Villarroel Walker 2010) and facilities are covered. For this study, no leaching and thus no losses to water are assumed.

For model city, the values reported by Kranert et al. (2012) are used: production of 134 kg/p*a of organic waste. As a simplification, a 100% transfer rate to the organic waste system is assumed. In reality, organics are also transferred to mixed waste, to own composting devices or directly to the environment (dumping).

### 2.6.9 PFOS

Estimating the load of PFOS for model city is not straightforward, due to lack of data and the multitude of potential sources and pathways. For the load of PFOS to water, a monitoring study of rivers in the EU (Pistocchi and Loos 2009) found 10 mg/p*a. This includes many potential sources for PFOS: household wastewater[18], leaching and erosion from land, atmospheric deposition, and emissions from landfills, direct discharge of urban run off and combined sewer overflow. WWTPs are often the main source of PFOS to water (Huset *et al.* 2008 Becker *et al.*2008).

The load of PFOS to water found in the monitoring study does not capture the emissions to other environmental compartments: to land and to air. Assuming that emissions to water represent 50% of total emissions, the load of PFOS[19] to all environmental compartments is estimated with 20 mg/p*a for model city.

The load of PFOS in wastewater is estimated with 14 mg/p*a, representing 70% of the total load. Due to its persistence, PFOS cannot be degraded by biological processes during "standard wastewater treatment". But it can be removed from effluent by sorption to sludge.

For the standard treatment, the partitioning factors reported by Buser and Morf (2009) are applied. Taking into account recent studies (Schultz *et al.* 2006, Sinclair and Kannan 2006, Loganathan *et al.* 2007, Heidler and Halden 2008, Huset *et al.* 2008); the authors estimated that 65% of the incoming PFOS are transferred to sludge.

The load of PFOS in sludge is 9 mg/p*a, representing 45% of the total load to the environment. If sludge containing PFOS is applied to land, the chemical quality of soils deteriorates and it may become a secondary source of emission to water. During incineration of sludge, the Carbon-Fluorine bond is broken and PFOS is degraded[20].

35% of the incoming PFOS remain in effluent resulting in an effluent load of 5 mg/p*a. The effluent from WWTP contributes 50% to the load of PFOS to water and 25% of the total load to the environment.

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18 Industrial applications of PFOS are banned in the EU

19 This includes the load of precursor substances that are degraded to PFOS during wastewater treatment, such as longer chained per- and poly-fluorinated substances

20 Other AMPs, especially heavy metals remain in the ashes or the air filter material.

This estimate is backed up by WWTP studies, which often show even higher loads per capita and by studies in Switzerland and Germany which found WWTPs the major source of PFOS to rivers (Huset *et al.* 2008, Heidler and Halden 2008, Becker *et al.*2008). But the input of PFOS to WWTPs and the fate during treatment is subject to large uncertainties. For the purpose of this study, the estimate serves to illustrate the problem of AMPs in wastewater in relation to CNP recycling and the potential improvement by integration of algae systems.

For the algae systems, it has been shown that processes during algae growth can increase the elimination of AMPs from effluent. Due to intense contact to cell surfaces capable of biosorption during a long hydraulic retention time of 3-6 days in an aerated environment, the transfer of AMPs to biomass and ultimately sludge can be increased compared to standard process. Elimination of heavy metals (Mallick 2002) and persistent organic pollutants (Munoz and Guieysse 2006, Borde *et al.* 2003 Arranz *et al.* 2008) is described for laboratory studies, but remains to be proven in pilot projects.

For model city, it is assumed that 85% of PFOS are adsorbed to biomass during algae cultivation, compared to 65% for the standard process. As this value is based on laboratory studies, it is subject to large uncertainties. Combined with an HE of 88%, elimination efficiency from effluent is 75%. The resulting effluent load is 3.5 mg/p*a, compared to 4.9 mg/p*a for the standard treatment. PFOS in sludge is degraded during incineration. Thus, sludge incineration is necessary to make algae systems clean cycles. But algae systems increase the amount of PFOS that is degraded during incineration.

Post treatment further reduces the load of PFOS in effluent, as the majority of the non-harvested biomass is removed from effluent. With the assumption that 95% of biomass is removed from effluent with post treatment, the elimination rate for PFOS from effluent increases to 80%. 2.7 mg/p*a remain in the effluent and 11.2 mg/p*a are transferred to sludge. The energy required for post treatment included in the energy balance of the algae + case.

# 3 Results and Discussion

## 3.1 Status quo of the urban water chain in Germany

For a holistic picture of the status quo of the urban water chain in Germany, first the external energy flows are presented and discussed, followed by the internal energy flows and the metabolic efficiency. Then, the extended energy balance is complimented by the emission balance.

To assess the current the current situation of the urban water chain, different cases are considered. For the water supply side, the average groundwater (70%) and surface water use (30%, ATT *et al.* 2011) is presented. For wastewater and sludge treatment, three cases are distinguished (Figure 3-1). The weighted average of these three cases gives the average for Germany.

Case 1 represents a medium to large WWTP employing anaerobic sludge stabilization with subsequent drying and incineration (52% of population served). Case 2 represents a medium to large WWTP employing anaerobic sludge stabilization with subsequent land use (23%). Case 3 represents a small WWTP with aerobic wastewater treatment and simultaneous sludge stabilization with subsequent land use (25%).

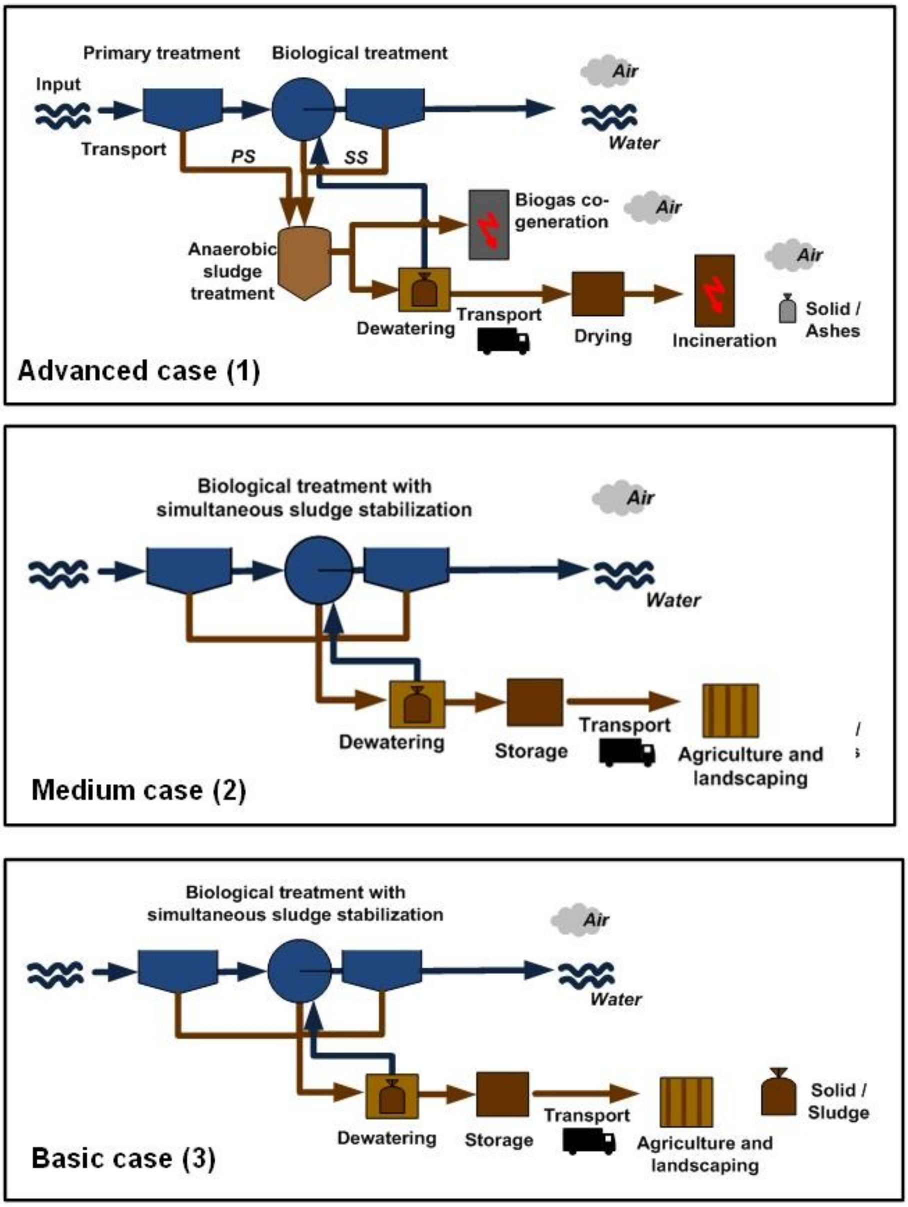


Figure 3-1: Status quo: Three cases for wastewater and sludge treatment

Legend: Basic, medium and advanced case of wastewater and sludge treatment. (Top) Case 1 represents a medium to large WWTP employing anaerobic sludge stabilization with subsequent drying and incineration (52% of population served). (Not shown) Case 2 represents a medium to large WWTP employing anaerobic sludge stabilization with subsequent land use (23%). (Bottom) Case 3 represents a small WWTP with aerobic wastewater treatment and simultaneous sludge stabilization with subsequent land use (25%).

The direct energy consumption and generation for the different stages of the urban water chain is shown in Figure 3-2. Red color represents electricity, orange color represents thermal energy and black represents primary energy in form of fuels for transport of sludge. The first stage of urban water chain, the extraction, treatment and distribution of drinking water requires 26 $kWh_{el}/p*a$ for 41 $m^3/p*a$. This value represents the weighted average of drinking water supply from groundwater (70%) and surface water (30%). It is noteworthy, that the

German value (0.3 kWh/m³ for sourcing and treatment) is low when compared to other countries (Plappally and Lienhard 2012), reflecting the outstanding quality and accessibility of water resources in Germany.

In general, the energy demand for drinking water supply depends mainly on the accessibility and quality of water resources, but also on the size of facilities and of distribution networks. Accessibility of water resources and extent of distribution network determines the required pumping energy and quality of water resources determines the required treatment. In water scarce regions, sourcing and treatment may much more energy. Plappally and Lienhard (2012) report up to 3 kWh/m³ for California.

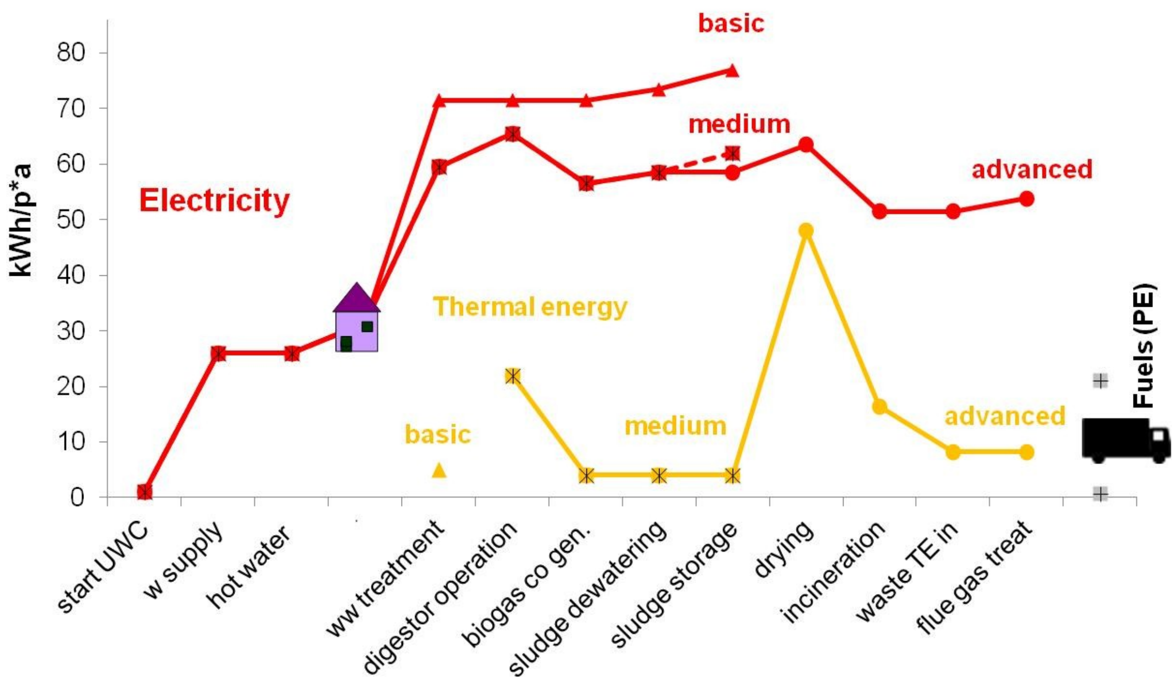


Figure 3-2: Status quo: External energy balance of the urban water chain (1)

Legend: Accumulated consumption and generation of electricity (red) and heat (orange) along the urban water chain for basic case (case 3), medium case (case 2), advanced case (case 1). Energy for transport (fuels, primary energy, low and high consumption) shown separately i.e. not accumulated.

For wastewater and sludge treatment, three cases are distinguished with different technical setups and different degrees of energy recovery. In all cases, transport of wastewater requires 5.5 $kWh_{el}$/p*a. Energy demand for wastewater transport is highly variable, as it depends on topography, characteristics of sewer system and amount of rainwater and extraneous water infiltrating the sewers. For Sweden, due to higher amounts of wastewater generated (mainly rainwater), energy demand per person in is much higher than in Germany: averaging 20 $kWh_{el}$/p*a with 313 $m^3$/p*a, with a range from 1-89 $kWh_{el}$/p*a (Olsson 2012, Lingsten *et al.* 2008).

Case 3 with the most basic technical setup, requires 40 $kWh_{el}$/p*a and 15 $kWh_{thermal}$/p*a for biological wastewater treatment and simultaneous sludge stabilization. Mechanical dewatering of sludge requires approximately 2 $kWh_{el}$/p*a. Land use of sludge requires storage with 3.5 $kWh_{el}$/p*a, but no other processing. Without anaerobic digestion or incineration, there is no energy recovery in case 3.

As there is no information available for transport distances of sludge to reuse sites in Germany, an average transport distance of 20 km (MUNLV 2001) as lower and 600 km (Haberkern, 2008) as higher value was chosen to give an envelope for energy demand of sludge transport. Energy demand is negligible for short distances with 0.7 $kWh_{primary}$/p*a, but increases to 21 $kWh_{primary}$/p*a for long distances. For land use of sludge, distances can be assumed to be at the lower end of the envelope.

Case 2 with anaerobic sludge stabilization requires 28 $kWh_{el}$/p*a for biological wastewater treatment. The energy demand for the biological wastewater treatment is lower than in case 3 described above. The simultaneous aerobic sludge stabilization requires additional aeration. Instead, sludge is stabilized anaerobically in case 2, requiring 6 $kWh_{el}$/p*a and 22 $kWh_{thermal}$/p*a for operation of digester.

The produced biogas has a lower heating value of 47 kWh/p*a. Due losses by flaring or thermal only use of biogas and a low average efficiency of electricity generation (30%), co generation currently recovers only 9 $kWh_{el}$/p*a and 22 $kWh_{thermal}$/p*a. Further processing of sludge follows case 3 described above.

Case 1 employs anaerobic sludge stabilization, as described for case 2 above, but with subsequent drying and incineration of stabilized sludge. For the model, a mixture of co incineration of sludge in coal fired power plants (50%), and mono incineration in older and newer facilities (each 25%) was assumed. In average, drying consumes 5 $kWh_{el}$/p*a and 44 $kWh_{therm}$/p*a and incineration generates 12 $kWh_{el}$/p*a and 32 $kWh_{therm}$/p*a. Input of waste heat from coal fired power plants for co incineration contributes additional 8 $kWh_{therm}$/p*a. Energy requirements for treatment of flue gas to German standards for mono incineration is 2.5 $kWh_{el}$/p*a. Taken together, sludge drying and incineration generate surplus electricity of 4.5 $kWh_{el}$/p*a and requires external supply of thermal energy of 2.2 $kWh_{therm}$/p*a. If no waste heat was used, the required external supply of thermal energy was 10.2 $kWh_{therm}$/p*a.

Figure 3-3 summarizes the brut and net consumption of electricity and thermal energy of the urban water chain. Brut consumption is similar in case 1-3 with 75-71-77 $kWh_{el}$/p*a. But electricity generation from internal resources varies: for case 1 with biogas use and incineration of sludge, 21 $kWh_{el}$/p*a are generated, covering 28% of brut consumption, for case 2 with biogas use without incineration of sludge, 9 $kWh_{el}$/p*a are generated, covering 13% of brut consumption. Net electricity consumption i.e. external supply required, is lowest in case 1 with 54 $kWh_{el}$/p*a, followed by case 2 with 62 $kWh_{el}$/p*a. For case 3, there is no energy recovery and net consumption equals brut consumption with 77 $kWh_{el}$/p*a.

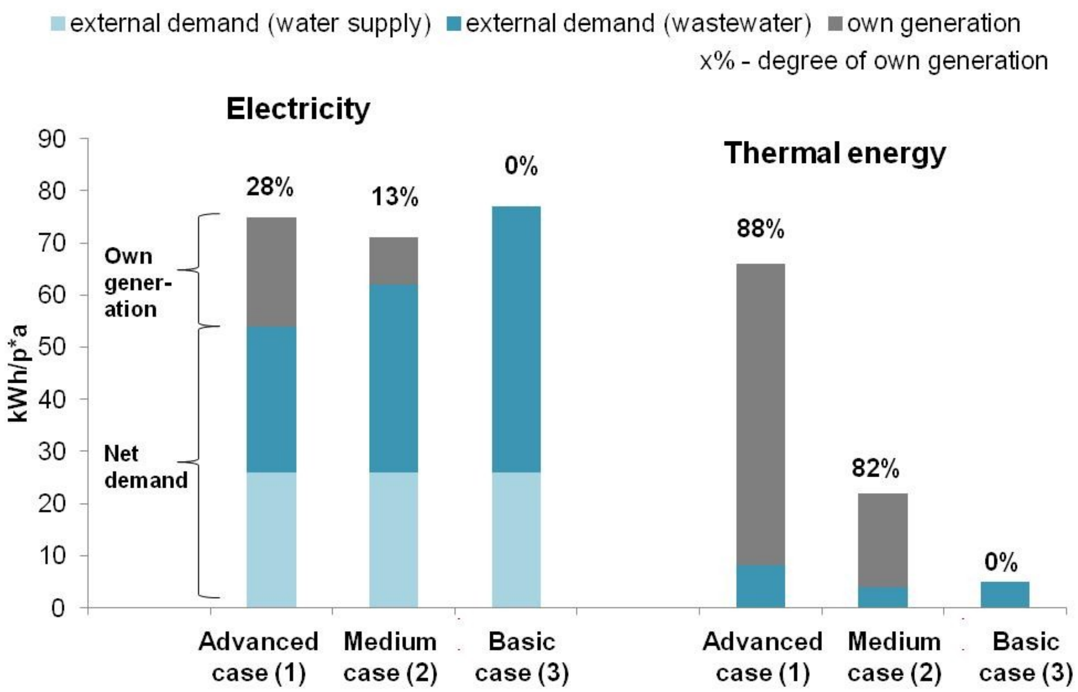


Figure 3-3: Status quo: External energy balance of the urban water chain (2)

Legend: Brut consumption of electricity and thermal energy for water and wastewater infrastructures (case 1-3), including consumption of energy generated from internal resources (own generation from C resources, grey) and external or net consumption on level of water (light blue) and wastewater infrastructures (dark blue). Percentages refer to current energetic reuse in relation to brut consumption

While case 1 employs energy harvesting from biogas and incineration processes, there are considerable losses. If the processes were optimized according to the best available technology, the energetic reuse can be considerably increased (case 1+). If all produced biogas contributed to electricity generation, i.e. no flaring or thermal only use of biogas occurred, and with improved electrical efficiency of 35%, electricity generation from biogas can be increased to 16 $kWh_{el}/p*a$. If old facilities for mono incineration were replaced by newer facilities, electricity generation from sludge incineration can also be increased to 16 $kWh_{el}/p*a$. With a total electricity generation of 32 $kWh_{el}/p*a$ due to optimized processes, 43% of the brut electricity demand of the urban water chain would be covered.

Looking at the thermal energy, the brut consumption of is highest in case 3 due to sludge drying. But 88% of brut demand is covered by internal resources or by waste heat in case of sludge co incinerated in coal fired power plants. The net consumption of thermal energy is small and very similar in case 1-3 with 5-8 $kWh_{thermal}/p*a$.

The weighted average for Germany over case 1 (representing 52% of population served), case 2 (23%) and case 3 (25%) is net consumption of 62 $kWh_{el}/p*a$ and 7 $kWh_{thermal}/p*a$. In average, current reuse covers 18% of brut electricity demand and 84% of brut demand for thermal energy.

On a primary energy base, net energy demand of infrastructures averages 189 $kWh_{primary}/p*a$ (Figure 3-4). Primary energy consumption includes the electricity consumption for water supply and wastewater and sludge treatment, the thermal energy consumption and the consumption of fuels for transport (low: 20 km distance, and high: 600 km distance) with the respective conversion factors.

The $CO_2$ emissions associated with the energy consumption average 36 kg/p*a ($CO_2$ equivalents). These originate mainly from electricity consumption for water supply and wastewater and sludge treatment with a low contribution of thermal energy consumption. With large transport distances for sludge, fuels can add considerably to $CO_2$ emissions (up to 6 kg for 600 km sludge transport).

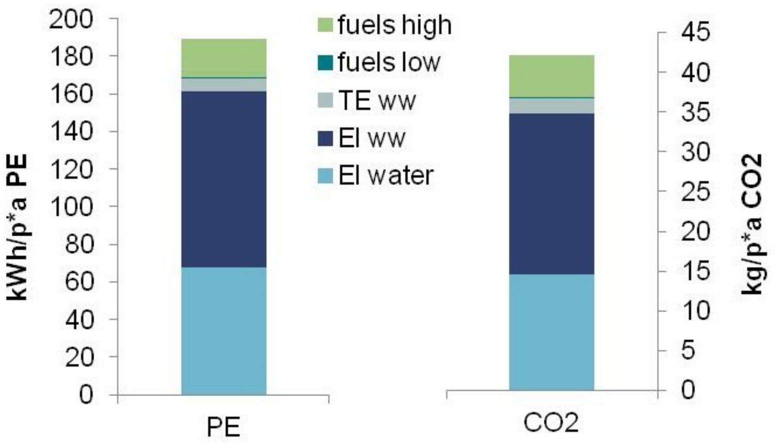


Figure 3-4: Status quo: Primary energy demand and $CO_2$ emissions

Legend: Average primary energy demand (kWh/p*a PE, left) and off site $CO_2$ emissions of the urban water chain (kg/p*a $CO_2$ equivalents, right). Contribution of net electricity consumption EL, consumption of thermal energy TE and fuels. Contribution of fuels for short and long transport distance of sludge (green). Weighted average from case 1-3 representing the average for Germany.

To extend the usual approach to energy balances as presented above, the internal energy flows are included. By quantification of the energetic value of CNP and application of the respective theoretical energy potential (TEP) factors to the SFA, the internal energy flows are traced along the urban water chain. Thus, the metabolic efficiency of the urban water chain can be assessed: the degree of energetic reuse of internal resources CNP (Figure 3-5).

For an input of 14 kg C/p*a (see chapter 2.2) and the derived TEP factor for C (chapter 2.3), the theoretical energy potential of C resources is 254 $kWh_{primary}$/p*a. With 4 kg N/p*a, TEP of N is 66 $kWh_{primary}$/p*a, which is lower by a factor of 4 than TEP of C. Again lower is TEP of P with 6 $kWh_{primary}$/p*a for 0.7 kg P/p*a.

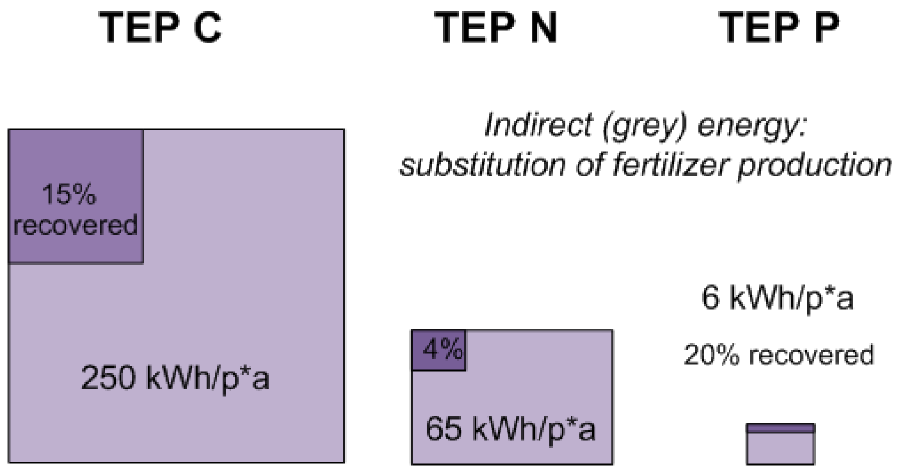


Figure 3-5: Status quo: TEP of CNP and metabolic efficiency

Legend: Theoretical energy potentials of CNP in [kWh/p*a primary energy]. Weighted average over the three cases. Percentages refer to current metabolic efficiency *i.e.* energetic reuse in relation to TEP on a primary energy base [%]

Energetic resources in wastewater differ not only in quantity, but also in quality. C resources have a high quality: they can be exploited for electricity and heat generation *via* biogas or incineration of sludge. Based on the TEP and the actual electricity generation from C resources (see Figure 3-3), the current metabolic efficiency for C is 15% in average (23% for case1, 9% for case 2 and 0% for case 3).

In contrast to direct energetic reuse of C, reuse of N and P only gains indirect energy credits by substituting energy intensive fertilizer. Taking into account the amount of N and P applied to agricultural land and the plant availability, reuse of N averages 4% of TEP N (0% for case1, 6% for case 2 and 8% for case 3). The reuse rate for P averages 19% (0% for case1[21], 30% for case 2 and 30% for case 3). Expressed in energetic terms, reuse of N recovers 2.8 $kWh_{primary}$/p*a and reuse of P recovers 1.2 $kWh_{primary}$/p*a. Despite the considerably smaller reuse rate of N compared to P, N contributes more than double in absolute terms to the recovered TEP.

Taken together, the energetic reuse of CNP averages 42 $kWh_{primary}$/p*a on a primary energy base with a metabolic efficiency of 13%. Thus, 283 $kWh_{primary}$/p*a of TEP CNP are not re-

[21] P remains in ashes and can theoretically be recovered. While the technologies have been developed, there is no large-scale application (UBA 2012).

covered. It is noteworthy that the non recovered energy potentials are higher than the primary energy demand of the urban water chain of 189 $kWh_{primary}$/p*a (see Figure 3-4).

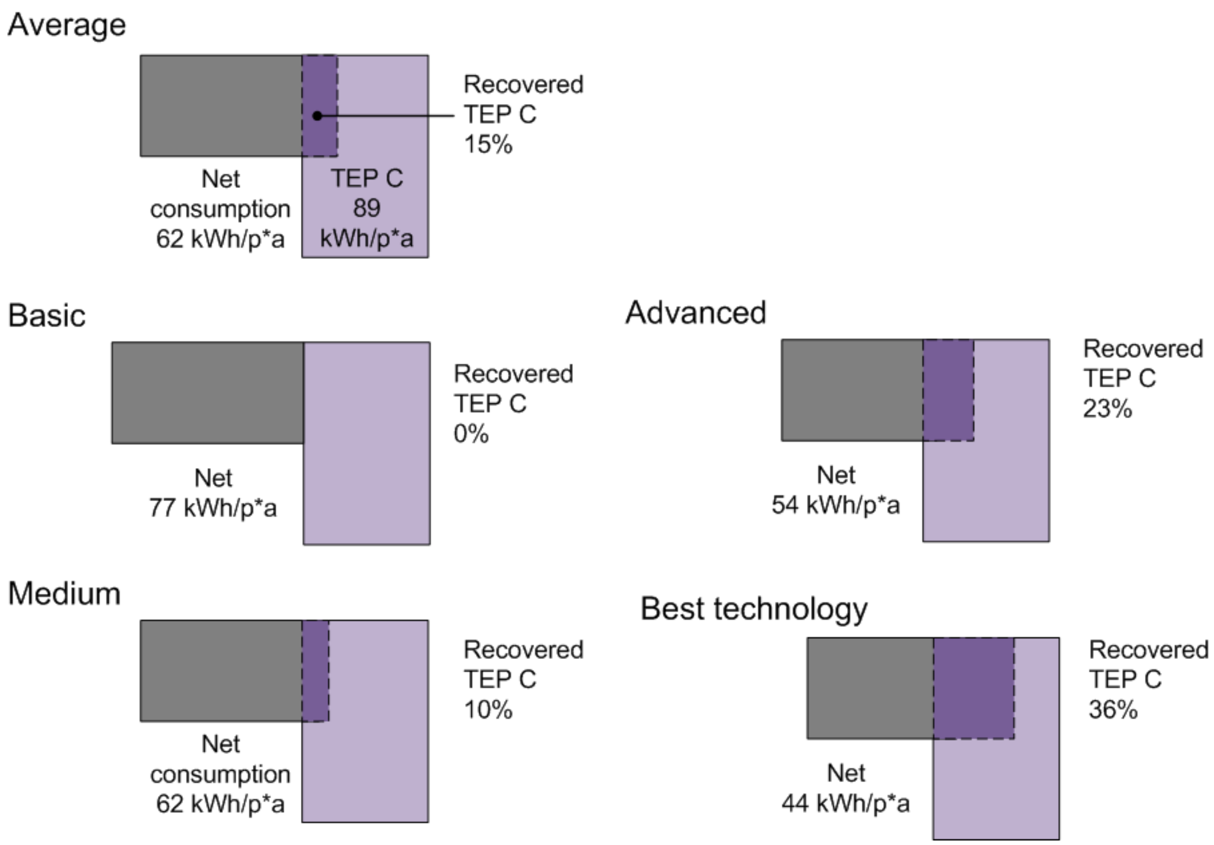


Figure 3-6: Status quo: Extended energy balance of the urban water chain

Legend: Extended energy balance in $kWh_{el}$/p*a for the German average, basic case (case 3), medium case (case 2), advanced case (case 1) and best available technology case (case 1+) showing the net electricity consumption. The metabolic efficiency or recovered TEP of C is shown as percentage of full TEP.

The C resources can be exploited for electricity generation. The maximum electricity generation, based on the TEP and 35% electrical efficiency, is 89 $kWh_{el}$/p*a. Thus, in theory electricity from C resources in wastewater can cover the current brut electricity consumption of the full urban water chain. The extended energy balance, comparing net electricity consumption, current electricity generation of from C resources and theoretical potential of C resources is shown in Figure 3-6. The average reuse rate for Germany i.e. the ratio of own generation to $TEP_{el}$ C, is 15%. For the three cases, reuse is 23% of TEP in case 1, 9% in case 2 and 0% in case 3. Thus, even with biogas generation and incineration of sludge (case 1), 77% of TEP C is currently not recovered. And even for the best available technology case with optimized biogas and incineration processes (case 1+, as described above) the metabolic efficiency for C is only 40%.

Looking at the SFA results give an indication of where the energy is lost. Following the C flows through the system shows that more than one third of incoming C is lost to air without energetic reuse during biological wastewater treatment (37%, Figure 3-7). But even in the steps with energy recovery, there are losses. This becomes obvious when comparing the substance flows of C with the actual energy recovery expressed as % of TEP. SFA results show, that 65% of incoming C is transferred to AD. Then, 29% of C is transferred to biogas *resp.* flue gas from biogas use. But the electricity generation from biogas combustion recovers only 9% of the TEP of C.

The sludge incineration receives 29% of incoming C and recovers 14% of TEP. While sludge incineration is slightly more efficient than biogas combustion in terms of TEP recovery, the picture is different when the external energy requirements are included (see Figure 3-2). For AD, electricity consumption is 6 $kWh_{el}$/p*a (9 $kWh_{el}$/p*a generated), while for dewatering, drying and flue gas treatment, electricity consumption is 9.5 $kWh_{el}$/p*a (12 $kWh_{el}$/p*a generated)[22]. Thus, net electricity gain is slightly higher for biogas processes than for sludge incineration.

22 As simplification, this calculation excludes the consumption of thermal energy and energy for transport, as shown in Figure 3-2

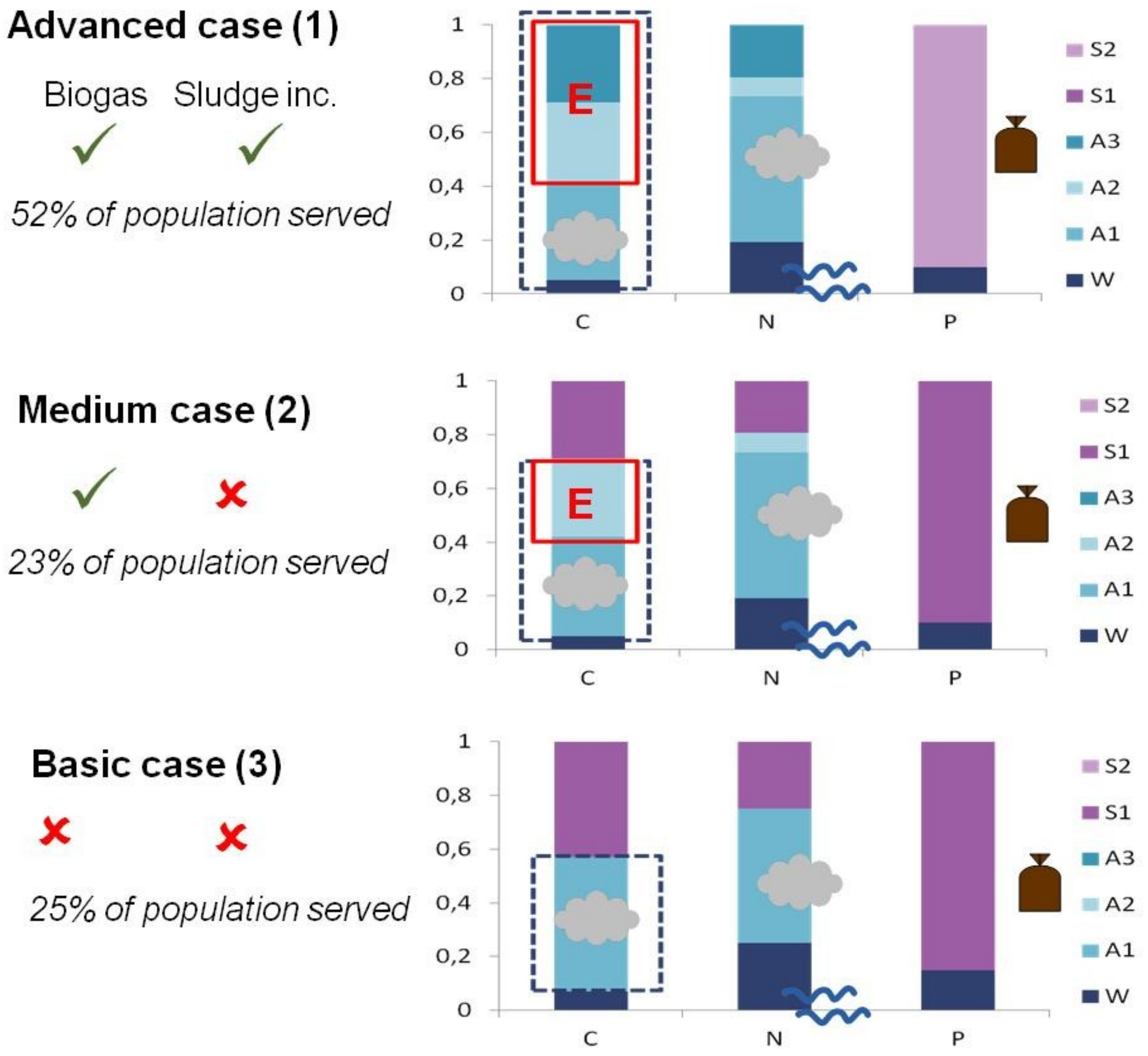


Figure 3-7: Status quo: Emission balance

Legend: Fate of C (left bar), N (middle bar) and P (right bar) in case 1 (top), case 2 (middle) and case 3 (bottom). Emission to water (W), air (A1: BWT, A2: AD, A3: Inc) and sludge resp. land (S1: stabilized sludge for land application, case 2 and 3, S2: ashes from sludge incineration, case 1).

In analogue to the metabolic efficiency, the $CO_2$ intensity of electricity production reflects the degree of reuse of internal C resources. On site emissions of $CO_2$ are considered renewable and are not accounted for in GHG reporting based on IPCC convention (Sahely *et al.* 2006, Rosso and Stenstrom, 2008). But the magnitude of $CO_2$ emissions from the flow streams of the urban water chain (internal resources) is noteworthy. These internal $CO_2$ emissions add up to 48 kg/p*a in case 1, 33 kg/p*a in case 2 and 25 kg/p*a in case 3 (see also Figure 3-7 for C emissions to air). This is in the same range as the $CO_2$ emissions originating from energy consumption off site (fossil C, average 36 kg/p*a $CO_2$ equ. see Figure 3-4). While the on site emissions cannot be avoided, as C is part of the flow streams, their magnitude underlines the importance of improving the $CO_2$ intensity of bio-electricity production.

If all C from the flow streams was degraded to $CO_2$, the emission would be 50 kg/p*a of $CO_2$. In combination with the TEP, this gives an ideal $CO_2$ intensity of 0.6 kg/kWh. The current $CO_2$ intensity of electricity from biogas is 1.9 kg/kWh. When the emissions during BWT are included, it rises to 3.2 kg/kWh. For sludge incineration it is 1.2 kg/kWh. The low metabolic efficiency is also reflected in the high (on site) $CO_2$ intensity of electricity generation.

Figure 3-7 summarizes the emission of CNP to water; air and land. The applied treatment technologies remove the majority of CNP from effluent. Elimination efficiency is 95% for C, 80% for N and 95% for P in case 1 and 2. For case 3, which represents smaller plant with aerobic sludge stabilization, elimination efficiencies are slightly lower: 93% for C, 75% for N and 85% for P. The non degradable C fraction remaining in effluent also contains organic AMPs, such as PFOS (see chapter 3.3).

Effluent quality is the traditional focus of urban water infrastructures, in line with the protection of water resources from eutrophication as a major function. But the systems perspective in this study requires accounting for emissions to all environmental compartments, as CNP eliminated from effluent is transferred to air or to sludge. In case of P, the majority (90% for case 1and 2 and 85% for case 3) is transferred to sludge either biologically: by assimilation in microbial biomass, sorption to flocs and or chemically: by precipitation with added iron or alum based precipitants. The iron or alumo phosphates contribute to the low plant availability of P in sludge of 61% (Bengtsson *et al.* 1997, Houillon and Jolliet 2005). Besides P, 29% of incoming C and 19% of incoming N are transferred to sludge in case 1 and 2. For case 3 with aerobic sludge stabilization, the ratios are slightly higher: 43% of C and 25% of N. Sludge is also an important sink for many AMPs (see chapter 3.3).

In contrast to P, the majority of C and N are transferred to air. In total, 95% of C is transferred to air in case 1, 66% in case 2 and 50% in case 3. The picture is similar for N: in total, 81% of C is transferred to air in case 1, 61% in case 2 and 50% in case 3.

Besides the bulk flow, the speciation of C and N released to air is important, as it has a large influence on the global warming potential (GWP). In the best case, all C is emitted as $CO_2$ and all N as $N_2$. To recap, on site emissions of $CO_2$ are considered renewable and are not accounted for in GHG reporting based on IPCC[23] convention (Sahely *et al.* 2006, Rosso and

23 Intergovernal Panel on Climate Change

Stenstrom, 2008). But other speciations, especially $CH_4$ and $N_2O$ have a much higher GWP than $CO_2$: 25 and 298, respectively (IPCC 2006).

For WWTPs with anaerobic digestion (case 1 and 2), it is estimated that in average 0.0084 g of $CH_4$ are produced for every g influent chemical oxygen demand (COD) at (Lazarova *et al.* 2012). These emissions occur especially with out gassing from sludge water after anaerobic digestion or with leakages. At 116 g/p*d COD, and a global warming potential of 25, this gives 9 kg/p*a $CO_2$ equivalents contributed by $CH_4$ emissions at WWTPs (359 g/p*a CH4). In addition, $CH_4$ may also evolve in the sewer system in case of long retention times and lack of oxygen (independent from AD).

For $N_2O$ emission, it is estimated that 0-5% of incoming N can be emitted to atmosphere as $N_2O$ at WWTP (Lazarova *et al.* 2012). According to IPCC guidelines (IPCC 2006), a factor of 0.035% is applied. This applied value is based on a single study, while other studies show higher emission factors (Kampschreur 2009). For the low factor from IPCC, the annual load of $N_2O$ is 2 g/p*a. With a higher, but still plausible factor of 0.5% the annual load is 30 g/p*a. With a GWP of 298 (IPPC 2006), $N_2O$ emissions contribute 0.6 to 9 kg/p*a $CO_2$ equivalents. In the worst case (5% as reported in Lazarova *et al.* 2012) emissions amount to 90 kg/p*a $CO_2$ equivalents.

To conclude, the analysis showed the low metabolic efficiency of the urban water chain today. The magnitude of the non recovered energy potentials underlines the importance of energetic reuse of CNP resources in wastewater. This is especially important for C, which can be exploited for bio-electricity generation. The current metabolic efficiency for C averages only 15%. Comparing the external and internal energy flows shows that C resources in wastewater can theoretically supply enough electricity to cover the demand of the full urban water chain. With the best available technology, the metabolic efficiency of the urban water chain can be increased to 36% of TEP C (case 1+). Further increase requires minimization of losses occurring during BWT (AS). Today, more than one third of C energy is lost at this treatment step. These air emissions without energy recovery contribute considerably to the high $CO_2$ intensity of current bio-electricity generation. These emissions are considered renewable and cannot be avoided, as C is a major constituent of the flow streams. Nevertheless, their magnitude – same range as the (fossil) emissions from energy consumption - underlines the importance of increased energetic reuse of C resources.

The energetic value of N and P is considerably lower than for C. Their reuse only gains indirect energy credits for substitution of fertilizer. The metabolic efficiency averages 4% for N and 20% for P. While reuse of N and P from wastewater on agricultural soils is beneficial from the energy and resource perspective, AMPs can be transferred to soils together with the nutrients (see chapter 3.3). While the metabolic efficiency for N and P is already low, it can be expected to further decrease in the future. Due to concerns about soil contamination, land application of stabilized sludge shows a decreasing trend in Germany (UBA 2012).

The current technical setup of wastewater and sludge treatment can also be considered a clean cycle process. It works reliably and fulfills its main functions for public health and protection of water resources from eutrophication. But the *quantity* and *quality* of CNP recycling from wastewater needs to be improved to contribute to clean cycles for sustainable urban metabolism.

## 3.2 Algae systems for increased metabolic efficiency

While the focus of the preceding chapter was the full urban water chain in its current state and the national averages, this chapter zooms in to the level of an individual WWTP. The integration of algae systems in the existing WWTP processes is investigated with focus on the CNP recycling in algae systems and the energy and emission balances of the WWTP with and without algae systems (Menger-Krug *et al.* 2012). As an example, a WWTP for 20 000 p.e. employing anaerobic digestion was chosen.

A process design (Figure 3-8) is proposed for integration of algae systems, which relies solely on resources from wastewater, with no external input of water, fertilizer or $CO_2$. The whole algae process chain, from cultivation to production of bio-electricity, takes place on site of the WWTP.

For growth medium, the algae systems receive primary treated wastewater (PTW, blue arrow to algae systems) and sludge water (brown arrow to algae systems). These flows are rerouted to algae systems instead of the biological wastewater treatment (BWT) step. Algae systems receive additional $CO_2$ which is delivered from sources on site the WWTP: combustion gas from biogas based co generation and gaseous emission from biological wastewater treatment (BWT, purple arrows to algae systems). The harvested biomass (green arrow) is co-digested with sludge to produce biogas. Biogas is used for co generation of electricity and heat on site (see section on energy balance below). The combustion gas, as well as sludge water, is recycled back to algae systems (back load cycles).

As this is a prospective analysis, there is no empirical data available. Therefore, the implications for the energy balance of the WWTP are calculated based on SFA results. The amount of biomass is calculated based on the nutrients provided to the algae systems combined with the uptake efficiencies and the stoichiometric requirements. From the amount of biomass generated, taking into account the harvesting efficiency and the anaerobic digestibility, the additional biogas generation is calculated. The changes in loads to different treatment steps arising from rerouting of internal flows to algae systems are used as proxies to calculate the changes in energy consumption.

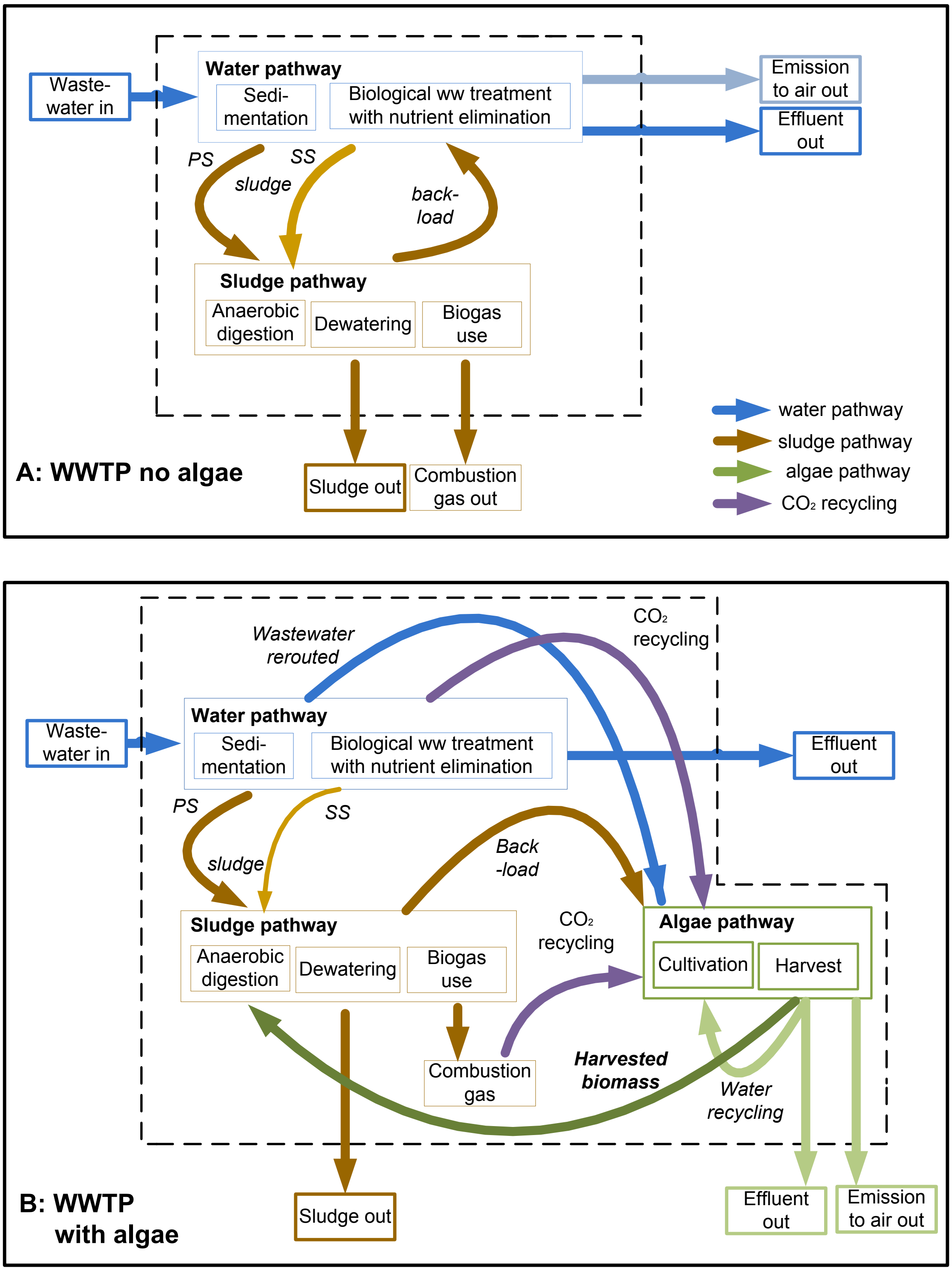


Figure 3-8: Algae systems: Proposed process design

Legend: Overview of inputs, internal flows and outputs of WWTP (A) and WWTP with integrated algae systems (B). PS: primary sludge from mechanical treatment and SS: secondary sludge from biological wastewater treatment (BWT). WWTP with algae systems employs full CO2 recycling (algae full scenario): gaseous emission from BWT and combustion gas are fully recycled to algae systems for CO2 supply (purple arrows). N and P requirements to exploit the provided CO2 are met by sludge water (fully diverted to algae systems, brown arrow) and a fraction of PTW (rerouted before BWT, blue arrow). Harvested biomass (green arrow) is co-digested with sludge to produce biogas. Biogas is used for co generation of electricity and heat on site (see section on energy balance below). The combustion gas, as well as sludge water, is recycled back to algae systems (back load cycles).

Three scenarios are presented for integration of algae systems at WWTPs (Table 3-1). The "algae light" scenario uses $CO_2$ generated at the WWTP during daytime (60% of total $CO_2$ available). To fully exploit the $CO_2$ provided, nutrient rich sludge water plus 11% of primary treated wastewater (PTW) are required to supply sufficient N and P. With the limitations imposed by nutrient requirements and uptake efficiencies of algae, and the reported rates of nutrient recycling *via* combustion gas and sludge water (back loads), 30 g of biomass are harvested daily for every person served by the WWTP. With an areal productivity of 18 g/m²*d and a harvesting efficiency of 88%, this requires 1.7 m² of cultivation area. Co digestion of harvested biomass increases biogas production by 61% compared to WWTP without algae systems.

The "algae medium" scenario uses 80% of total $CO_2$ available. It requires storage capacities for $CO_2$ which cannot be directly supplied to algae system (as it is generated during night when there is no light available). With more $CO_2$, more biomass can be produced, requiring more area, as well as more N and P. 32% of primary treated wastewater (PTW) is required for nutrient supply in addition to sludge water, producing 57 g/p*d of biomass. With constant areal productivity, required cultivation area expands to 3.2 m²/p. Co digestion more than doubles biogas production compared to WWTP without algae systems. The "algae full" scenario fully exploits $CO_2$ generated on site and therefore requires large capacities for night time storage. Here, 57% of primary treated wastewater (PTW) together with sludge water is fed to 5.7 m²/p of algae systems to fully exploit the $CO_2$ provided from on site sources. 90 g/p*d of biomass are harvested and biogas production almost triples compared to WWTP without algae systems.

Integration of algae systems changes loading rates to biological wastewater treatment (BWT) and anaerobic digestion (AD). The BWT step receives 110% of influent N due to back load from sludge water at the WWTP without algae systems. Rerouting sludge water together with a fraction of primary treated wastewater (PTW) to algae systems considerably reduces loading to BWT. The reduction in N load is larger than the reduction in volume (26-44-64% reduction in N load *vs.* 11-32-57% reduction in volume), as sludge water is low in volume but high in N. The C/N ratio in the BWT step becomes more favorable, moving from 2.4 to 2.6 with integration of algae systems (on a mass base).

With decreased load to BWT, less secondary sludge is produced. Secondary sludge has a lower anaerobic digestibility and produces less biogas and more stabilized sludge than biomass harvested from algae systems. Reduction of secondary sludge contributes to the relatively higher increase in biogas production (by 65-115-183%) compared to the increase in loading of C to anaerobic digestion (by 35-65-103%). This indicates an efficient use of digester capacities at WWTP with algae systems.

Table 3-1: Algae systems: Summary of SFA results

| | no algae | algae light | algae medium | algae full |
|---|---|---|---|---|
| Algae systems | | | | |
| Harvested biomass [g/p*d] | | 30 | 57 | 90 |
| Area needed (cultivation) [m²] | | 1,9 | 3,6 | 5,7 |
| PTW diverted to algae systems[%] | | 11 | 32 | 57 |
| **Biological wastewater treatment (BWT)** | | | | |
| Loading of C [% of incoming] | 73 | 60 | 46 | 29 |
| Loading of N [% of incoming] | 110 | 81 | 62 | 39 |
| Loading of P [% of incoming] | 100 | 80 | 61 | 39 |
| **Anaerobic digestion(AD)** | | | | |
| Loading of C [% of incoming] | 65 | 87 | 107 | 131 |
| Biogas produced- total [l/p*d] | 23 | 36 | 49 | 64 |
| Contribution of harvested biomass to biogas production [%] | | 42 | 60 | 73 |

Changes in loading rates induce changes in energy demand of the different treatment steps: lower volume and loads reduce the energy demand of the biological wastewater treatment (BWT), while additional loads increase energy demand of anaerobic digestion (AD). The reduction in N load to BWT and the increase in C load to anaerobic digestion are used as proxies for changes in electricity use of these treatment steps. Figure 3-9 shows energy balances for the WWTP without and with algae systems per person served and day of growing season. On the demand side, the electricity use of WWTP increases moderately with integration of algae systems: the savings in biological wastewater treatment (BWT) counterbalance increased consumption in sludge and algae pathway.

On the supply side, co digestion of biomass from algae systems considerably increases electricity generation from biogas. This outweighs slightly reduced amounts of biogas from secondary sludge. The contribution of primary sludge is stable in all four scenarios, as wastewater is rerouted to algae systems after primary treatment. In total, the net energy balance of WWTP improves by 41-71-102% with rising degree of $CO_2$ exploitation in algae systems.

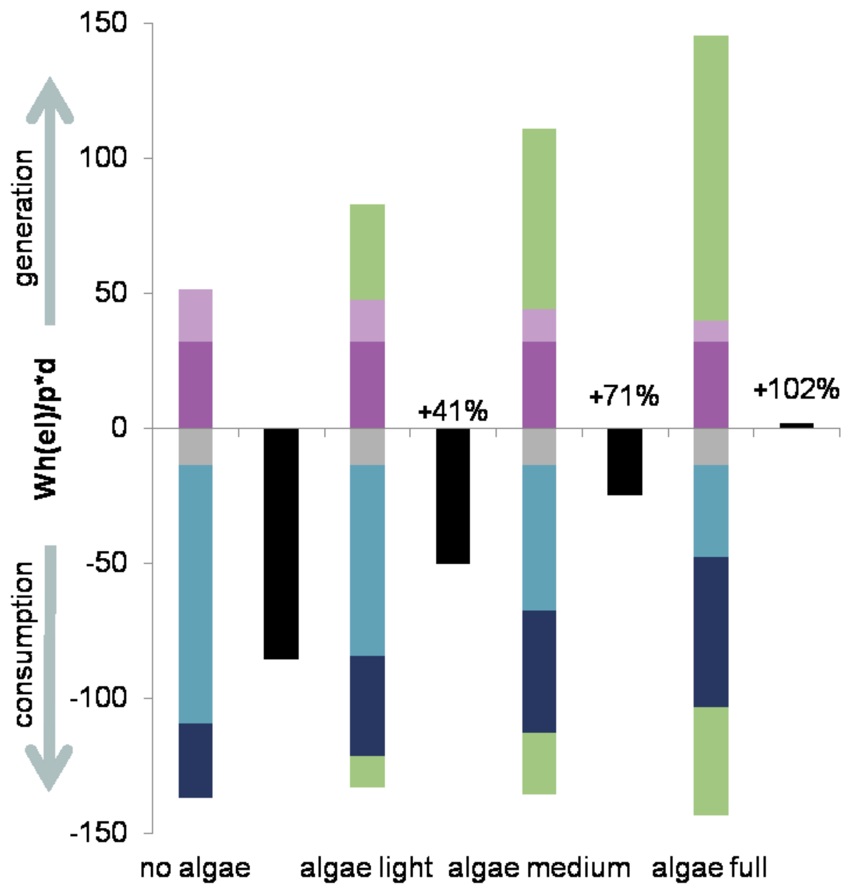


Figure 3-9: Algae systems: External energy balance

Legend: Energy balance of WWTP (no algae) and WWTP with integrated algae systems in $Wh_{el}$ per person served and day of growing season. The algae light scenario with 1.9 m²/p of cultivation area uses 60% of $CO_2$ generated at the WWTP, algae medium 80% (3.6 m²/p), algae full 100% (5.9 m²/p). Demand side: electricity use of water pathway (light blue), sludge pathway (dark blue), algae pathway (green) and other uses (grey). Supply side: electricity generation from biogas from primary sludge (dark purple), secondary sludge (light purple) and co digestion of biomass harvested from algae systems (green). Black bars represent net energy balance (demand – supply).

For comparison to other energy systems, the energy return on investment (EROI) is a useful measure. For algae systems, weighing the energy generation from co digestion and the savings in biological wastewater treatment (BWT) (energy output) against the energy consumed for biomass cultivation, harvesting and processing and the reduced energy generation from secondary sludge (energy input), gives an EROI of 2.1-2.4. For every kWh of electricity consumed due to the integration of algae systems more than 2 kWh of electricity are generated. Looking at the full WWTP, the EROI is 0.38 without algae systems. With an EROI below 1, more electricity is consumed than generated. The integration of algae systems improves the EROI of the WWTP to 0.62 in the light, 0.8 in the medium and 1.01 in the full scenario. With full $CO_2$ exploitation in algae systems, WWTP can run energy neutral during vegetation season.

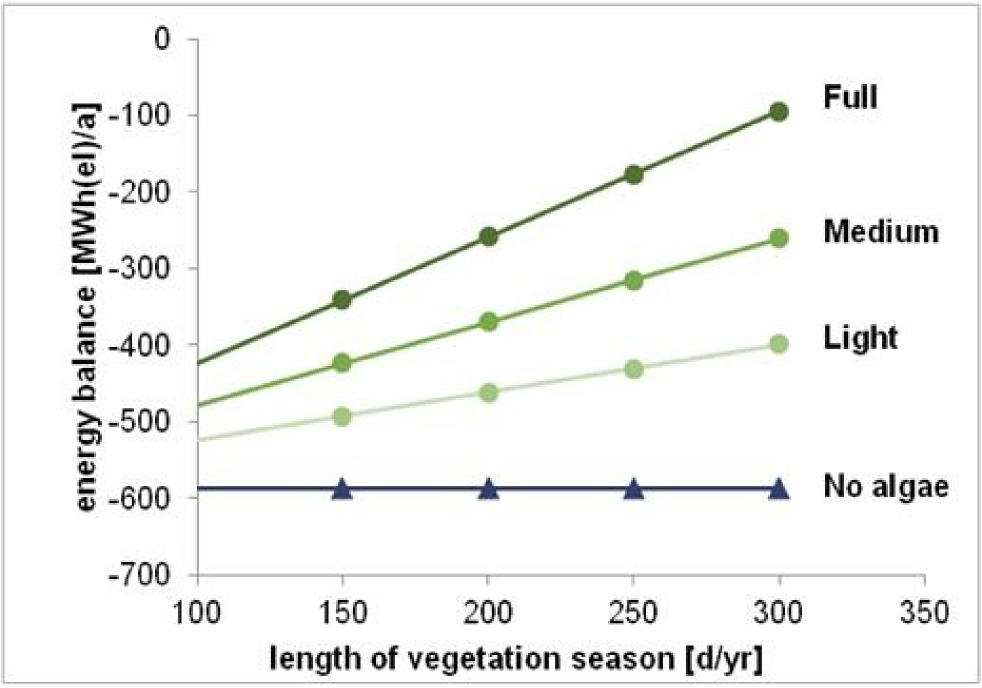


Figure 3-10: Algae systems: Annual energy balance

Legend: Annual energy balance depends on length of vegetation season. WWTP (no algae, blue) and WWTP with algae systems: algae light (light green), medium (green) and full (dark green) scenarios. Results on plant level for 20,000 persons served.

On an annual basis, the energy balance depends on the length of the vegetation season (Figure 3-10), which equals the number of days per year with a minimum temperature above 5°C and maximum temperature below 35°C (Murphy and Allen 2011). But the growth medium for algae systems at WWTPs has relatively high temperatures of approximately 15-20°C, even at low ambient temperatures, due to digester heating and hot water use in households. Warm growth medium may prolong the vegetation season for algae systems at WWTPs. Assuming 200-250 days per year algae systems improve the annual energy balance compared to WWTP without algae systems by 22-28% in the light scenario, 39-49% in the medium scenario and 56-70% in the full scenario.

So far, the positive implications for energy balance arising from integration of algae systems at WWTPs were presented. But for a full picture, the emission side has to be included (Table 3-2). The results of the SFA show, that integration of algae systems also has disadvantages for WWTPs. It reduces the elimination efficiency for CNP from effluent. To recap, C in effluent of WWTP is usually measured with the sum parameter chemical oxygen demand as proxy (COD, factor 3 for C to COD). While in the WWTP without algae systems, 8% of incoming C, 28% of incoming N and 9% of incoming P remain in effluent, loads increase with rising degree of $CO_2$ exploitation in algae systems. In the full scenario, 17% of incoming C, 35% of incoming N and 22% of incoming P remain in effluent.

Table 3-2: Algae systems: Emission balance

| | no algae | algae light | algae medium | algae full |
|---|---|---|---|---|
| **Water** | | | | |
| Elimination efficiency C/COD [%] | 95 | 91 | 87 | 83 |
| Elimination efficiency N [%] | 72 | 67 | 66 | 65 |
| Elimination efficiency P [%] | 90 | 86 | 81 | 78 |
| **Sludge** | | | | |
| Total stabilized sludge [g/p.e.*d] | 60 | 64 | 66 | 69 |
| Additional sludge [%] | | 7 | 10 | 14 |
| load of C in sludge [% of incoming] | 29 | 31 | 33 | 36 |
| **Air** | | | | |
| Load of C to air [% of incoming] | 66 | 59 | 54 | 46 |
| $CO_2$ emission on site (renewable) [g/p.e.*d] | 93 | 84 | 76 | 66 |
| $CO_2$ intensity of bio-electricity production [kg/kWh] | 1,8 | 1,0 | 0,7 | 0,5 |

The amount of stabilized sludge also increases slightly by +7-10-14% compared to WWTP without algae. Increase of stabilized sludge is modest compared to the increase in biogas production (see Table 3-1) due to the high anaerobic digestibility of biomass harvested from algae systems and reduced amounts of secondary sludge fed to anaerobic digestion (AD).

Looking at the fate of C shows that the load of C to water (17% of incoming C with algae full *vs.* 5% for WWTP) and to sludge (36% *vs.* 29%) increase with the integration of algae systems, while the load of C as $CO_2$ to air decreases (46% *vs.* 66%). While combustion gas is emitted to air at WWTPs without algae, it is recycled and partially transferred to biomass with algae systems. Despite the high anaerobic digestibility of harvested biomass, more C accumulates in stabilized sludge than without algae systems. Likewise, the amount of non-harvested biomass, which escapes *via* effluent, increases with increased C recycling and yield. The decrease of $CO_2$ emissions is noteworthy, as bio-electricity generation is considerably increased at the same time. The reduced $CO_2$ intensity of bio-electricity production indicates a more efficient energetic reuse of renewable C resources from wastewater with integration of algae systems. Due to recycling of combustion gas, one C atom can contribute to energy generation *via* $CH_4$ more than once before leaving the system in effluent, stabilized sludge or as air emission.

Zooming in on the effluent quality of WWTPs with algae systems, Figure 3-11 shows the contribution of free nutrients (blue) and nutrients incorporated in non-harvested biomass (green) to effluent concentrations of COD, total N and total P. Effluent concentrations meet limit values (dashed lines) with the assumed harvesting efficiency of 88% and in absence of post treatment in all scenarios, but they are very close in the full scenario.

Free nutrients are present in effluent in dissolved form, as organic or inorganic ions, while the non-harvested biomass incorporates nutrients mainly as organic molecules, such as lipids, carbohydrates and proteins. It represents the fraction of biomass not captured by harvesting process, likely consisting of smaller cells not forming flocs. The contribution of non-harvested biomass is visible for all nutrients, but most strongly for COD effluent concentrations (Figure 3-11-1). The effect of intense C recycling for effluent quality is obvious: the amount of non-harvested biomass in effluent increases in parallel with improved energy balances (Figure 3-11-2). The non-harvested biomass in effluent *resp.* the COD concentrations and the energy balances are tightly connected. Both are strongly influenced by total yield (at a given harvesting efficiency), which in turn is governed by the degree of C recycling.

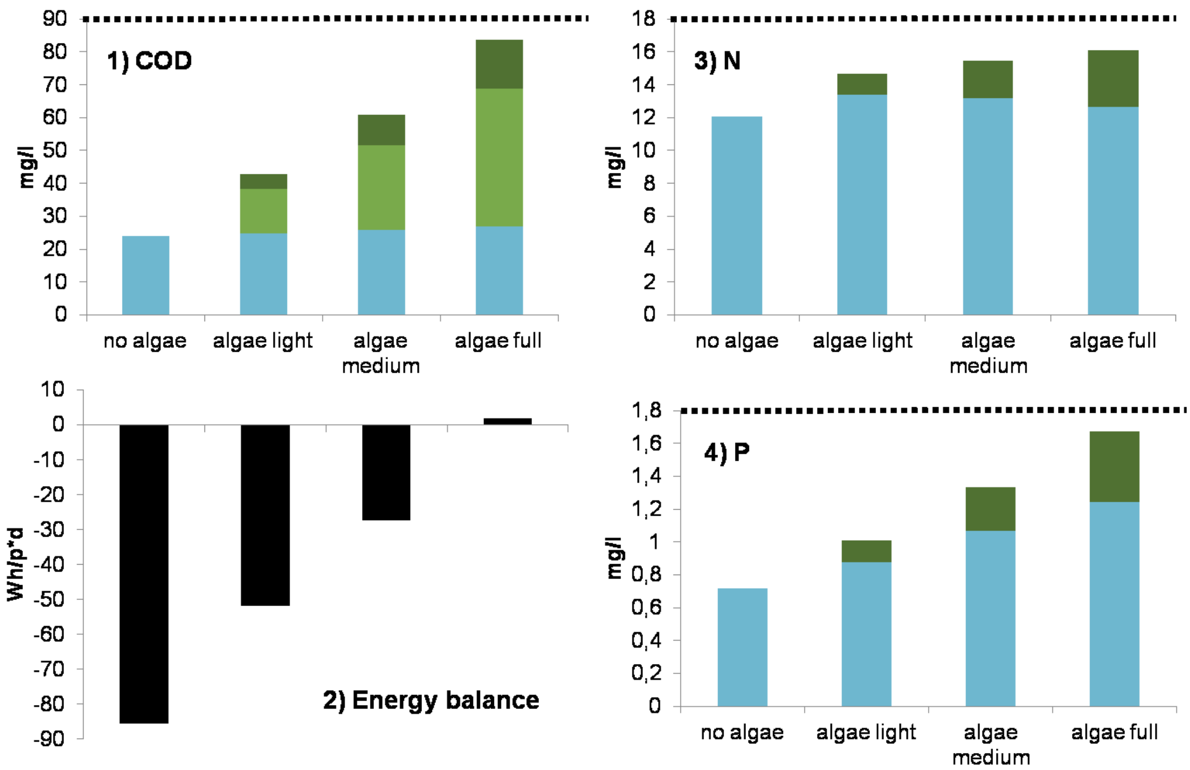


Figure 3-11: Algae systems: Effluent quality

Legend: Contribution of non-harvested biomass (green) and free nutrients (blue) to effluent concentrations of COD, N and P. Dashed lines represent respective limit value for effluent concentrations for German WWTP with 10,000-100,000 p.e.. Black bars represent energy balance for comparison. COD: Lighter shade of green represents C delivered as $CO_2$ in combustion gas and dark green represents C delivered as DOC in growth medium.

With rising degree of C recycling, COD effluent load increases by 21-72-136% compared to WWTP without algae systems. With full C recycling, more than half of the C in effluent originates from combustion gas recycled back to algae systems and incorporated in the non-harvested biomass. While contribution of non-harvested biomass pushes effluent concentrations towards limit value, the concentration of free COD is similar to WWTP without algae systems. It can be assumed that algae systems reach the same elimination efficiency for free COD as BWT (93%), albeit with a much longer hydraulic retention time.

To a lesser extent the non-harvested biomass also contributes to N and P in effluent, but there is also an increase in loads of free nutrients with integration of algae systems. Increase in load of free P in effluent is caused by the lower elimination efficiency in algae systems. With the assumed uptake efficiency of 80% for total biomass, 20% remains in the effluent as free P, while elimination efficiency in BWT is 90%. Therefore, the more PTW is rerouted from BWT to algae systems, the higher the load of free P in effluent. In algae systems, adsorption to biomass or higher uptake efficiencies would reduce effluent concentrations of free P. But in absence of these processes, P effluent concentrations move close to limit values with full C recycling in algae systems, as do COD effluent concentrations.

For free N, the increase in effluent load is not caused by differences in elimination efficiencies: the reported N uptake efficiency for total biomass in algae systems is the same as the elimination efficiency in standard biological wastewater treatment (BWT) (75%). To recap, nutrient rich sludge water is fully diverted to algae system in all three scenarios. The amount of primary treated wastewater (PTW) varies between 11% and 57%, according to the N and P required to fully exploit the $CO_2$ available. In all scenarios, this approach leads to a slightly higher than required N supply *i.e.* the amount of primary wastewater required to supply sufficient P supplies surplus N. The N/P ratio of growth medium (determined by the mixing ratio of sludge water and primary treated wastewater) is above optimum: 10 in the light scenario, 9 in the medium and 8.6 in the full scenario. The ideal growth medium has an N/P ratio of 8, based on the reported biomass composition of and uptake efficiencies. The N/P ratio is closest to optimum in the full scenario, as reflected by slightly decreasing free N concentrations from light to full scenario. For the full scenario the concentration of free N is similar to WWTP without algae systems.

As effluent concentrations are close to limit values with full exploitation of $CO_2$, it is prudent to add a post treatment step to maintain barrier function of WWTPs and protect aquatic ecosystems from eutrophication. Post treatment with activated carbon seems a viable option: absorption processes reduce the non-harvested biomass and the incorporated CNP, as well as free COD and free P. The energy demand is approximately 2.5 to 5 $kWh_{el}/p*yr$ or 7-14 $Wh_{el}/p*d$ (Haberkern *et al.* 2008, Hansen *et al.* 2012). Between 8% and 16% of energy savings in algae full scenario would be scavenged by post treatment. But as an additional benefit, post treatment also reduces the loads of many other pollutants in wastewater, *e.g.* heavy metals or AMPs (see chapter 3.3).

For the scenarios presented so far, average values for wastewater composition were used. With full C recycling, N/P ratio is close to required optimum and effluent concentrations are just below limit values. Changes in influent loads can push effluent concentrations above limit values. For a wastewater composition deviating by 20% from average, limit values are exceeded and post treatment becomes mandatory for WWTPs with full C recycling in algae systems (except for lower C influent load, see Figure 3-13). The same applies for reduced uptake efficiencies for N and P. With the same influent variations, effluent concentrations of WWTP without algae systems stay safely below limit values despite moderate increases. Higher influent loads of C and especially N increase the energy demand of WWTP. P load in contrast has a much lower impact on energy demand (unless post treatment is required).

First, changes related to N and P flows are considered: variations of wastewater composition, uptake efficiencies and biomass composition (Figure 3-12). As C is the limiting nutrient, changes related to N and P flows have little effect on total yields and consequently COD concentrations and energy balances. Deficits of N or P in relation to available $CO_2$ can be compensated by increasing the amount of primary treated wastewater (PTW) rerouted to algae systems. Related energy savings in biological wastewater treatment (BWT) moderately improve the energy balance in most scenarios but at cost of effluent quality. Changing the amount of PTW rerouted to algae systems impacts on the N/P ratio of the growth medium. The N/P ratio of the actual supply moves away from the respective optimum and causes exceedance of limit values for the respective surplus nutrient. As discussed above, the model does not allow luxury uptake of N or P above stoichiometric requirements or adsorption of free P to biomass. These processes could reduce effluent concentrations below limit values despite variations. It is concluded that in absence of these processes, flexibility for changes in N and P flows requires addition of a post treatment step to meet effluent limit values with full C recycling.

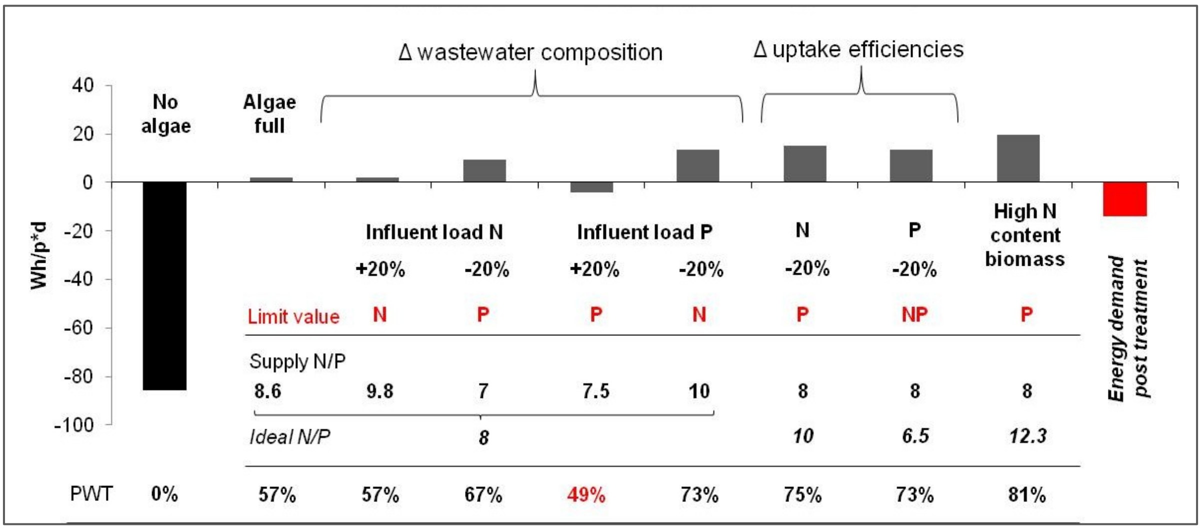

Figure 3-12: Algae systems: Scenario Analysis (1)

Legend: Energy and emission balances for variations of algae full scenario. Energy balances of no algae, algae full and parameter variations of algae full scenario (bars). Red bar to the right shows energy demand of post treatment for comparison. Below bars: (1) Limit value: Exceedance of effluent limit value(s) is indicated by red letters: N and/or P. Effluent concentrations of C resp. COD stays below limit values with all analyzed variations, as yield is not affected. (2) Supply N/P: the N/P ratio of the growth medium in the scenario, determined by the mixture ratio of PWT and sludge water (3) Ideal N/P: optimum N/P ratio required by biomass based on the respective stoichiometric composition of biomass and the respective nutrient uptake efficiencies. If supply is above optimum, surplus N stays in effluent. If below optimum, surplus P stays in effluent. (4) PWT: amount of primary treated wastewater rerouted to algae systems. In most scenarios, a higher amount of PWT is required to compensate deficits of N or P in relation to available $CO_2$. Related energy savings in BWT moderately improve the energy balance. Except for scenario with high P influent load: here only 49% PWT are required to meet N and P demand, downgrading the energy balance by 7% compared to algae full scenario. In scenario with low P uptake efficiency, N and P limit values are exceeded. The former due to N/P balance (surplus N) the latter due to the reduced uptake efficiency. In contrast, with reduced uptake efficiencies for N, effluent concentrations increase but stay below limit values. Parameter variations: (1st set) changes in wastewater composition with +/-20% influent load for N and P; (2nd set) changes in nutrient uptake efficiencies for N (60% instead of 75%, -20%) and P (65% instead of 80%, -20%), and higher N content of biomass (9% N content instead of 6%, +33%).

On the upside, energy demand of post treatment (red bar in Figure 3-12) can easily be afforded by the energy savings compared to WWTP without algae and has additional benefits for effluent quality. Actually, with a higher N influent load energy demand for WWTP can be expected to increase by 15% due to higher energy consumption for aeration and less favorable C/N ratio in biological wastewater treatment (BWT), increasing the relative savings in this scenario.

Including the energy demand of post treatment, reduces energy savings compared to full scenario (*without* post treatment): by 2% or less for lower uptake efficiencies for N or P, for lower influent load of P and for a higher N content of biomass; by 7% for a lower N influent load; by 15% *resp.* 20% for a higher N *resp.* P influent load. To recap, while the algae full scenario just meets limit value, adding post treatment is prudent and scavenges 16% of savings compared to WWTP without algae systems. Compared to algae full scenario *with* post treatment, all analyzed variations have an equal or better energy balance except the scenario with higher influent load of P.

In contrast to changes related in N and P flows, changes related to C flows affect total yield, as C is the limiting nutrient. Total yield is tightly connected to the energy balance, as well as to effluent concentration of COD by the contribution of non-harvested biomass. In Figure 3-13, the interrelationship between energy balance (y-axis) and COD concentration in effluent (x-axis) is further investigated. Increasing the influent load of C by 20% (black triangle) improves the energy balance by 34%, compared to the algae full scenario. With more C available, total yield increases: more area is needed for algae cultivation (6.8 m²/p instead of 5.7 m²/p) and more primary wastewater is required to supply demand of N and P (69% instead of 57%). But the increase in non-harvested biomass in effluent accompanying the higher yield pushes COD concentration above limit value. Scenario moves to upper right in Figure 3-13. Effluent concentrations of N and P show only small effects due to increased contribution of non-harvested biomass, and stay below limit values (compare Figure 3-11 for contribution of non-harvested biomass to effluent concentrations of N and P). But meeting COD limit values requires a post treatment step, which can easily be afforded with the energy savings compared to algae full scenario. Savings compared to WWTP are 120% or more. With a higher C influent load energy demand for WWTP can be expected to increase by 5-10%, thus increasing the relative savings in this scenario.

A lower C influent load reduces the total biomass yield together with the area required and the volume of PTW rerouted to algae systems. While energy balance is downgraded by 37%, effluent concentrations of COD (and to a lesser extent N and P), decrease and scenario moves to lower left towards algae medium scenario. Similar to lower C influent loads, lower uptake efficiencies for C considerably downgrade energy balances, while non-harvested biomass and therefore effluent concentrations decrease. A lower uptake efficiency for C as $CO_2$ delivered by combustion gas shows a stronger effect, than for C delivered as DOC by growth medium. Reduced C influent load and reduced uptake efficiencies move the scenarios to lower left, close to or even below algae medium scenario. While the area demand is also reduced in these scenarios, the scenarios with reduced uptake efficiencies require the same $CO_2$ storage infrastructure as the full scenario.

Given the importance of non-harvested biomass for effluent quality, especially for COD, harvesting efficiency is a key factor at WWTP with algae systems. To recap, the algae full scenario requires a harvesting efficiency of 88% to stay below COD limit values, which is in the higher range of values reported in the literature (65%, Park and Craggs 2010; to 95%, Lundquist 2010). Increasing the harvesting efficiency (+10%) moves the algae full scenario to upper left, improving energy balance and effluent quality simultaneously. By removing the formerly non-harvested biomass, load of COD in effluent is reduced. Energetic reuse of the formerly non-harvested biomass improves energy balance compared to full scenario by 19%. Adding a post treatment step to algae full scenario (blue square), which removes the non-harvested biomass but without energetic reuse, can reach the same effluent quality. But due to the additional energy demand, energy balance is moderately downgraded compared to full scenario (16%).

With decreased harvesting efficiency (80%), increasing contribution of non-harvested biomass pushes COD concentration, as well as P concentration above limit values. As consequence of lower harvesting efficiency, the amount of harvested biomass decreases downgrading the energy balance by 10%.The medium scenario with 80% $CO_2$ recycling still meets limit values with 76% harvesting efficiency and the light scenario (60% $CO_2$ recycling) even with only 55%, but with a similar downgrading of energy balance.

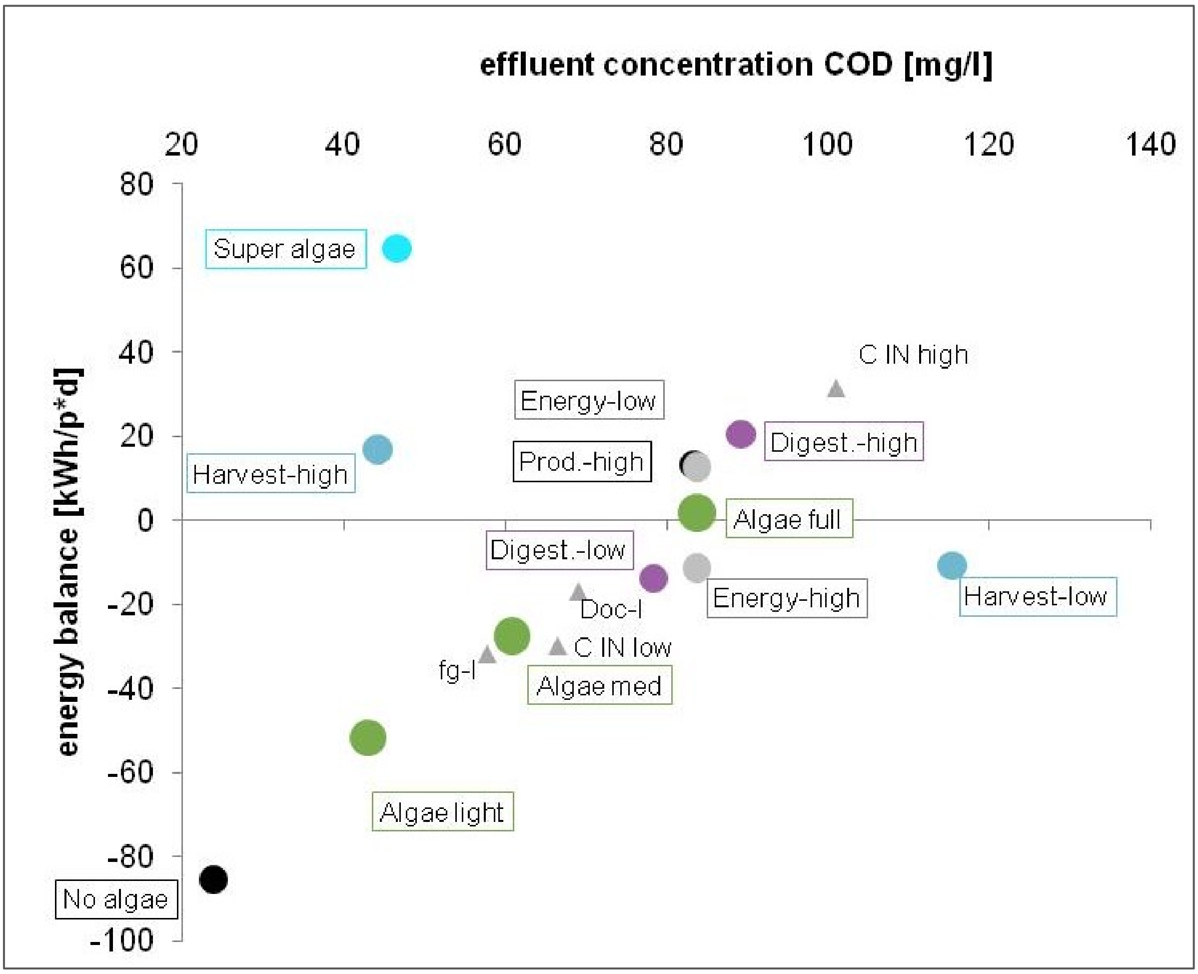


Figure 3-13: Algae systems: Scenario Analysis (2)

Legend: Energy balance vs. COD effluent concentrations for WWTP without algae (no algae, black circle), algae light, algae medium and algae full (green circles) and parameter variations of full scenario: Influent load of C +/- 20% (influent C high, back triangle; influent C low, grey triangle); lower uptake efficiencies for C delivered as DOC by growth medium (25% instead of 50%, DOC l) and for C as $CO_2$ delivered by combustion gas (50% instead of 75%, $CO_2$-l, grey triangles); Harvesting efficiency 88+/- 10% (blue circles); Post treatment (blue square): algae full scenario with post treatment step, which removes 66% of the non-harvested biomass but without energetic reuse (energy demand 14 $Wh_{el}$/p*d); Anaerobic digestibility 70+/-10% (purple circles); Energy demand of algae systems(Energy-high and Energy –low, 70 kWh/m²*d +/-25%, grey circles); Areal productivity (Prod.-high, 18 to 25 g/m²*d, +38%, black circle behind "Energy –low"); Super algae: combining optimistic values for harvesting efficiency (97%), anaerobic digestibility (77%), energy demand (53 kWh/ha*d) and biomass productivity (25 g/m²*d).

Compared to variations of the harvesting efficiency, variation of the anaerobic digestibility (purple circles) has a considerably lower impact on COD effluent concentrations, but a slightly higher impact on energy balance. Variation of anaerobic digestibility by 10% give +22% *resp.* -14% change in energy balance compared to full scenario (harvesting efficiency +19% *resp.* -10%). A higher anaerobic digestibility increases the energy output per unit of harvested biomass. It also increases C recycling via combustion gas and reduces the amount of stabilized sludge generated. Values for mono-digestion of algae biomass in the literature range from 40-60% (Clarens *et al.* 2010) to 70-90% (Hernandez and Cordoba 1993). But it has to be taken into account that co-digestion with sludge potentially enhances digestibility of algae biomass compared to mono digestion, due to reduced ammonia inhibition (Sialve 2009, Samson and Leduy1983).

Two more parameter variations are included in Figure 3-13: the energy demand of algae systems and the areal productivity. In contrast to the parameter variations discussed so far, they have no effect on effluent quality, only on energy balance. For the base scenarios 70 $kWh_{el}$/ha*d for cultivation and harvest, and a *total* biomass productivity of 18 g/m²*d during the vegetation season (and 88% harvesting efficiency) is assumed. Values in the literature for energy demand of algae systems range from 50 (Campbell 2011) to 127 kWh/m²*d (Collet 2011), with biomass productivities around 25 g/m²*d. Increasing the biomass productivity to 25 g/m²*d improves energy balance by 17% compared to full scenario and as positive side effect considerably reduces area demand. Reduced energy demand for algae cultivation and harvest, has a similar effect on energy balance, but not on area demand.

A "super algae" scenario is created to show the potential of the approach with optimized technologies. It combines optimistic values for harvesting efficiency, anaerobic digestibility, energy demand, and biomass productivity (as reported in the legend of Figure 3-13). With a vegetation season of 250 days, super algae can fully supply annual electricity demand of WWTP, and produce a surplus of more than 100 $MWh_{el}$/yr, while WWTP without algae systems has an electricity demand of 600 $MWh_{el}$/yr. Effluent concentrations of COD are similar to the algae light scenario *i.e.* slightly higher than for the WWTP without algae.

To conclude, the results show that it is feasible from a flow stream perspective to produce enough bio-electricity from algae systems, to run WWTP energy-neutral during the vegetation season or even turn them into net energy producers. This can be achieved with nutrients and $CO_2$ from wastewater, without any external resource input. C *resp.* $CO_2$ availability is the limiting factor for yield with the proposed process design *i.e.* in absence of external $CO_2$ sources. Bio-electricity produced at WWTPs with algae systems has a low $CO_2$ intensity, indicating efficient re-use of renewable C resources from wastewater.

While intensive C recycling in algae systems considerably improves the energy balance, it also impacts on effluent quality, mainly *via* the contribution of non-harvested biomass. This effect is most visible for C *resp.* COD: effluent concentrations increase due to the contribution of non-harvested biomass in parallel to improved energy balances, as both depend strongly on total yield. Non-harvested biomass also contributes to effluent concentrations of P and N, albeit to a lesser extent.

The algae full scenario marks the upper limit for C recycling in absence of post treatment: limit values are met while an energy neutral operation of the WWTP during vegetation season is achieved. The results highlight the tight connection between C flows, total yield and effluent quality for algae systems at WWTPs. The harvesting efficiency is identified as a technical key parameter at the crossroads of energy balance and effluent quality.

As effluent concentrations are close to limit values with full C recycling, post treatment is highlighted as an opportunity to reliably meet effluent limit values for COD, N and P. The energy costs for post treatment are determined at 8-16% of total savings.

Post treatment becomes mandatory with a wastewater composition deviating from average. Besides reliably meeting effluent limit values for COD, N and P, adding post treatment also improves the effluent quality in terms of AMPs.

It is noteworthy that besides the post treatment also processes in algae systems themselves can contribute to elimination of anthropogenic pollutants: bio-oxidation, bio-sorption or bio-assimilation, supported by a long hydraulic retention time of 3-6 days in an aerated environment. Elimination of heavy metals and persistent organic pollutants is described for laboratory studies (see chapter 1.3.3, but remains to be proven in pilot projects. Besides the fate of AMPs in algae systems, there are other important research needs. This includes the possible formation of algae toxins during biomass growth under certain growth conditions, as known from eutrophic water bodies, and the $N_2O$ emission from N in wastewater.

## 3.3 Connecting the urban water chain to urban metabolism

The previous chapters described the status quo of energy and material flows along the urban water chain (chapter 3.1) and assessed the integration of algae systems for increased metabolic efficiency (chapter 3.2). In this final chapter, the perspective of the analysis is extended to put the results generated so far into context of the urban metabolism.

The aim of this chapter is to assess the importance of the urban water chain – with and without algae systems - in context of the urban metabolism. This includes the role of the urban water chain in context of:

- the urban energy and C flows
- the urban nutrient flows (N and P)
- the urban flows of anthropogenic micropollutants (AMPs)

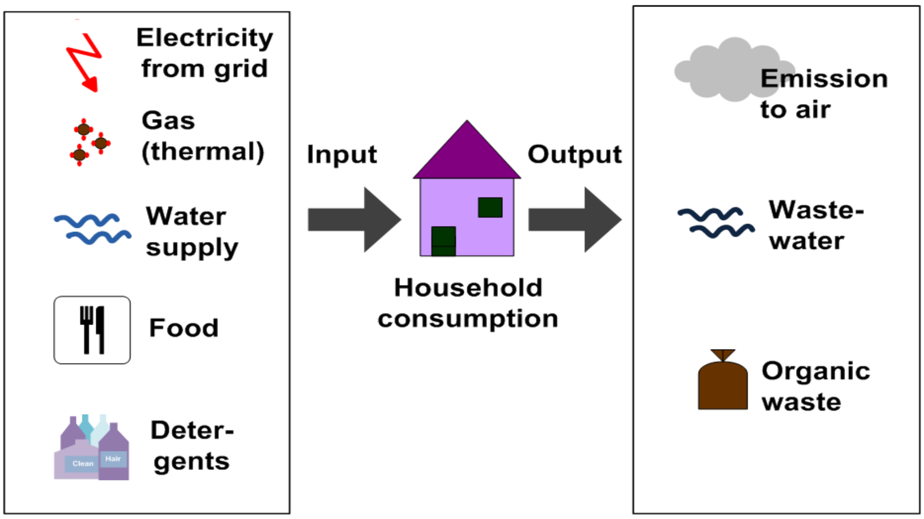


Figure 3-14: Model city: Overview of material and energy flows

Legend: Households consumption in model city. Extended perspective of the analysis includes the connection points of the urban water chain to full urban metabolism: the flows of energy, water, food and cleansing product related to daily household consumption and the associated flows of CNP. On the post use side, air emissions, wastewater and organic waste are included.

For the extended perspective, the flows of energy, water, food and cleansing products related to daily household consumption are quantified for a semi hypothetical model city (Figure 3-14). The bulk flows (energy and material) and the associated CNP flows are traced from the point where they enter the household until they leave the astysphere as emission to air, water and land/soil. This includes the transformations of CNP in the consumed food during human metabolism; on the post use side it includes organic waste treatment, in addition to the wastewater and sludge treatment analyzed in the previous chapters. For wastewater and sludge management, four different cases – with and without algae systems - are compared. Besides CNP, also the flows of PFOS are assessed to discuss the problem of AMPs.

The model city analyzed in this study is pictured in a rural setting - with land resources for algae cultivation available around the WWTP - with 20 000 inhabitants. But gardening and urban agricultural activities (own production of food), as well as industrial or commercial activities are excluded from the present analysis. The focus lies on the daily household consumption. The daily household consumption of water, food and cleansing products represent the input side of the urban water chain and thus the connection between the urban water chain and the full urban metabolism. On the post use side, the organic waste management is included as "sister infrastructure" which receives parts of the food flows. The energy consumption of households is included to put the energy balance and the C flows of the urban water chain into the urban perspective. While far from a complete analysis of the urban metabolism, the analyzed flows cover large parts of the daily household consumption and allow a first assessment of the relevance of the urban water chain for different aspects of the urban metabolism.

*Input flows: Energy*

Energy represents a large part of daily household consumption. Household related energy consumption includes electricity and thermal energy. Energy consumption varies between individuals and depends on many factors such as household size, user behavior, floor space, insulation, appliances used, mode of hot water preparation (electricity or gas) (Schmidt *et al.* 2011, ). For model city, it is assumed that thermal energy for hot water and space heating is provided by natural gas with an average consumption of 8000 $kWh_{therm}$. As natural gas is used for thermal applications, the electricity consumption in model city is slightly lower than the German average: 1300 $kWh_{el}$ (Schmidt *et al.* 2011). With the C intensity of natural gas (50 g/kWh) and electricity from German grid (110 g/kWh)[24], the flows of C total ~550 kg/p*a. The largest contribution comes from the thermal energy (400 kg/p*a). The low C intensity of electricity from German grid reflects the large share of renewable (20%) and nuclear (18%) energy sources in German electricity mix (AGEB 2012). With electricity from

24 C and N emissions from electricity consumption occur upstream of the household and only the energy enters the household - in contrast to natural gas for thermal energy which is physically transferred to the household. For simplicity, air emission from electricity consumption are also termed "household emissions"

brown coal with a high C intensity (up to 600 g/kWh, UBA 2012), the contribution of electricity would be ~800 kg/p*a.

Besides C, most energy carriers such as coal and natural gas contain a certain amount of N that is released during combustion mostly as $N_2$. These flows add up to 2.9 kg/p*a N released to air. As the released $N_2$ adds to the non reactive N pool in the atmosphere, its ecological relevance is low. But the combustion processes may also transform a small fraction of this internal N or $N_2$ from air to $N_2O$. Air emissions of $N_2O$ are highly relevant as it is a very potent GHG. The emission of $N_2O$ totals 10 g/p*a for electricity consumption and 3.2 g/p*a for natural gas consumption.

It is noteworthy, that the wastewater treatment may contribute more to the $N_2O$ emissions than the full energy consumption albeit with large uncertainties. As calculated in chapter 3.1, wastewater treatment may release 2 g/p*a (IPCC 2006) to 30 g/p*a (Lazarova *et al.* 2012, Kampschreur *et al.* 2009) of $N_2O$ from N in the flow streams. This underlines the importance of further research in the topic of $N_2O$ emission from N in wastewater.

*Input flows: Water*

Looking at the mass flows related to daily household consumption, it is obvious that water represents the largest mass flow: 41 t/p*a or 820 000 t for model city. On the outgoing side, the mass flow is even larger as it includes also urban run off and sewer infiltration, which more than double the volume of flow streams. For comparison, material consumption of a typical city in an average industrialized country is estimated with 15–25 t/p*a (Krausmann *et al.* 2008). Thus, the urban water chain is a very important mass transport system in urban areas. Other studies confirm, that water is the largest component of the urban metabolism in terms of mass (Kennedy *et al.* 2007, Decker *et al.* 2000).

*Input flows: Food*

But the mass flows related to food consumption should not be underestimated. Even though conventionally not regarded as infrastructure system, the food supply system in cities also manages large mass flows. For model city, the food consumption accounts for ~680 kg/p*a or 13 600 t/a. To recap, while the focus of this study lies on the household consumption, it quantifies the full food consumption of the inhabitants of model city (including out of house consumption). The contribution of the major food groups to total food provision in model city, based on data from BMVEL (2012), is shown in Figure 3-15 (grey bars).

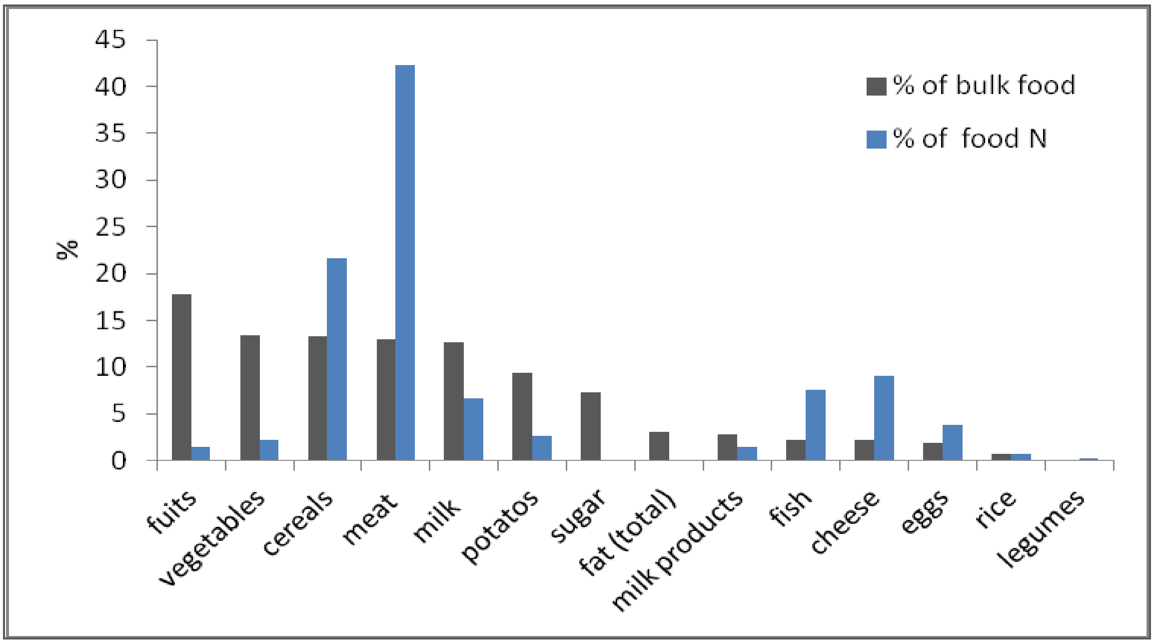


Figure 3-15: Model city: Food consumption (bulk food and N content)

Legend: Contribution of several food groups to total food provision (grey) and to total N content (blue) [%].Values for total food provision (grey) based on BMELV 2012, values for N content calculated from protein content reported in González et al. 2011 applied to BMELV 2012 data.

Not all food entering the household is consumed. In a recent study, it was found that food waste in Germany amounts to ~130 kg/p*a (Kranert *et al.* 2012). In the households, 65% of the food wastes are avoidable, for example food refused due to mismanagement, transgression of best-before-date or personal preferences (SRU 2012). The avoidable losses are considerably higher than the unavoidable losses during preparation, such as peels and inedible parts. Actual human consumption of food is ~550 kg/p*a (80% of food entering the households). Thus, the transfer rate to organic waste is 20% for model city.

Having established the bulk flows of food in model city, the associated CNP flows can be assessed. The assessment of the content of CNP in the food flows is not straightforward: from the provision of food, to food losses to actual human consumption to content of CNP in food, uncertainty becomes higher. For C content of food, an average of 20% of bulk food or ~136 kg/p*a is calculated. 80% of that amount is consumed with the food (~109 kg C) and 20% is transferred to the organic waste (27 kg C).

In absence of net biomass generation: growth in young people, pregnancy or lactation, the human body can be considered as steady-state equilibrium. All CNP taken up is released to the environment (Villarroel Walker 2010, Baker *et al.* 2007, Fissore *et al.* 2011). C is the energy source for human metabolism. As in all heterotrophic organisms, C from food is oxidized to $CO_2$, fueling the energy metabolism (catabolism). For model city, human respiration is estimated with 100 kg C/p*a in average, based on ( Villarroel Walker 2010, Baker *et al.* 2007, Fissore *et al.* 2011, Prairie and Duarte 2007). This leaves ~9 kg/p*a C from food transferred to wastewater. It is noteworthy, that the energy consumption of the human body itself causes considerable emissions of (renewable) C to air: ~100 kg *vs.* 140 kg for electricity and 400 kg for natural gas consumption (fossil C).

The N content of food can be quantified via the protein content, as N is almost exclusively incorporated in proteins (Fissore *et al.* 2011, González *et al.* 2011). Based the share of several food groups (BMELV 2012) and their protein content (González *et al.* 2011), the amount of N in food entering the household in model city is calculated with 6.7 kg. Of that amount, 5.4 kg is consumed by the human body and 1.3 kg is transferred to organic waste.

Some food groups contribute over proportionally to N content of food (Figure 3-15, blue bars). Food groups high in protein include animal products: meat and milk products, fish and eggs, and some plant products, such as cereals. Interestingly, potatoes which are an important part of the German diet are particularly low in N, while leguminoses with high protein content are sparsely consumed in Germany. The contribution of animal based products to total protein and N consumption is 70%.

The majority of N consumed with food leaves the body via urine and feces. The transfer rate to wastewater used in other studies is 90-100%. This means losses to air during human metabolism of less than 10% (0%: Baker *et al.* 2007, Fissore *et al.* 2011; 10%: Villarroel Walker 2010, Antikainen 2007). With a 90% transfer rate, the expected N load in wastewater from food consumption is 4.8 kg/p*a or 13.2 g/p*d, based on the food data (“top down” calculation).

But calculating the amount of N in food “bottom up” with values reported from the wastewater side gives a different picture. The relevant technical standards report 3.7 kg of N transferred to wastewater from food consumption (ATV 2000, DWA 2008, DWA 2013). These values refer to average loads in wastewater, including commercial operation and indirect dischargers, fitted to “person equivalents (p.e.)” via the C content. The results of this study indicate that these values may be underestimating the N load when applied to mainly residential areas such as model city. For the purpose of this study, a transfer rate to wastewater of 70% (losses to air during human metabolism of 30%) are assumed to resolve the discrepancy between bottom up and top down calculation, acknowledging the uncertainty.

The P content of food is calculated with 0.75 kg/p*a for model city. Of this amount, 0.14 kg/p*a (20%) are transferred to organic waste. The P in consumed food (80%) is fully transferred to wastewater via urine and feces (Baker *et al.* 2007, Fissore *et al.* 2011, Villarroel Walker 2010). The N/P ratio of food with the reported values is 9, which is plausible albeit on the higher end of the spectrum. In Figure 3-16, the values for the load of N and P in food calculated in this study are compared to published values (Baker *et al.* 2007, Fissore *et al.* 2011, Villarroel Walker 2010, Beck and Villarroel Walker 2012, Neset *et al.* 2007, 2008, Antikainen 2007). N and P load of food entering the household is in the medium to lower range of published values.

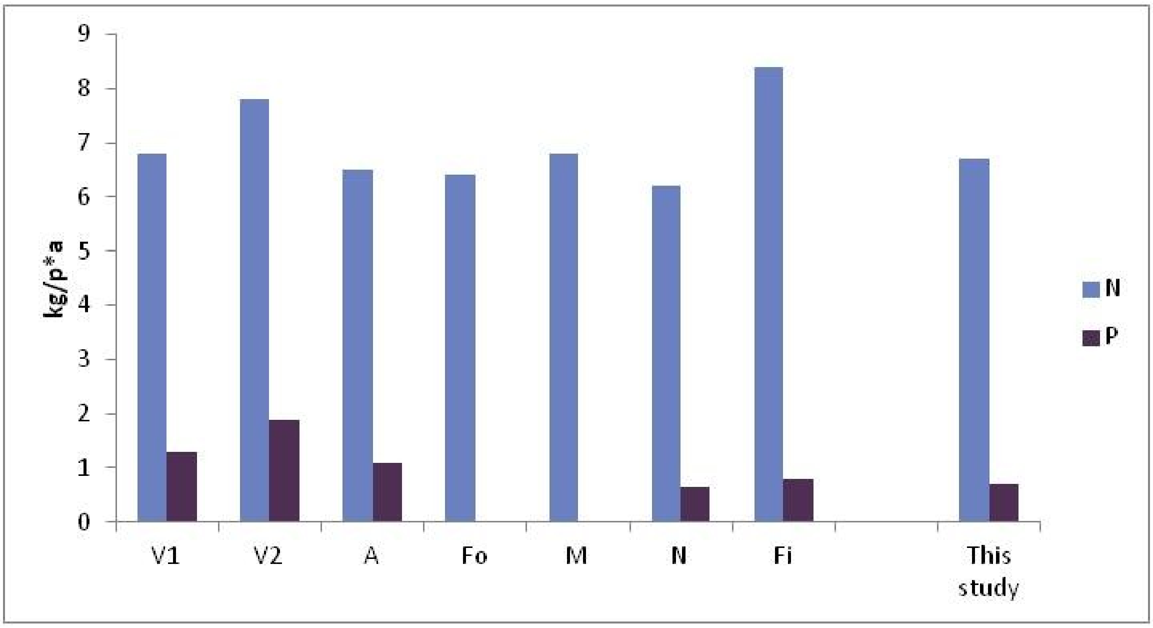


Figure 3-16: Model city: Food NP in comparison to other studies

Legend: Comparison of published values for the load of N and P in food entering the household with the values used in this study. Published values for the load of N (blue) and P (purple) in food. V1: Villarroel Walker 2010, low value; V2: high value; A: Antikainen 2007; Fo: Forkes et al. 2007; M: Metson et al. 2012; N: Neset et al. 2007 (N flows), 2008 (P flows); Fi: Fissore et al. 2011.

Despite the different geographical focus and regional dietary habits, the standard deviation for N load in food is only 10%. For P it is considerably higher (40%)[25]. The N/P ratio of food lies between 4.1-5.9 for the studies of the Upper Chattahoochee Watershed (Villarroel Walker 2010, Beck and Villarroel Walker 2012) and Finland (Antikainen 2007) and 9.5-10.5 for

[25] When the present study is excluded, the standard deviation is not considerably changed for N, but decreases to 37% for P

Linkoeping (Sweden, Neset *et al.* 2008) and Minneapolis (Baker *et al.* 2007, Fissore *et al.* 2011) (this study 9).

*Input flows: Detergents*

Like energy, water and food, also detergents are used in household on a daily base. Data from UBA (2012) show that private end users consume 1.3 million tons of detergents, including: 630 000 t of cloth washing products, 220 000 t of fabric softeners, 260 000 t of dish washing products and 480 000 t cleansing products for body and hair.

For model city, the associated load of CNP is calculated to 5.7 kg C, 0.3 kg N and 0.14 kg P per person and year. CNP from detergents is transferred to wastewater, contributing considerably to the P load and the C load in wastewater, while the contribution is negligible for N.

Figure 3-17 sums up the flows of CNP entering the households: energy, water, food, and detergents (top); and leaving the household as air emissions (energy and human metabolism), in wastewater or in organic waste (bottom).

*Flows to and from households exhibit different patterns for CNP*

Looking at the flows of CNP entering and leaving the household reveals different patterns for the elements. Looking at the C flows, the most dominant flows are associated with energy consumption. They account for approximately 80% of analyzed C flows to households. But the C flows related to food should not be underestimated (~20%).

On the outgoing side, the majority of C flows are transferred to air as $CO_2$. This includes the energy related C flows as well as the majority of C in food. All these emissions of C to air occur with energy recovery. This is obvious for the C flows related to consumption of electricity and thermal energy in households, but also the C from food is used energetically to fuel human metabolism. Only a fraction of the analyzed C flows is collected in the infrastructure systems for organic waste and wastewater.

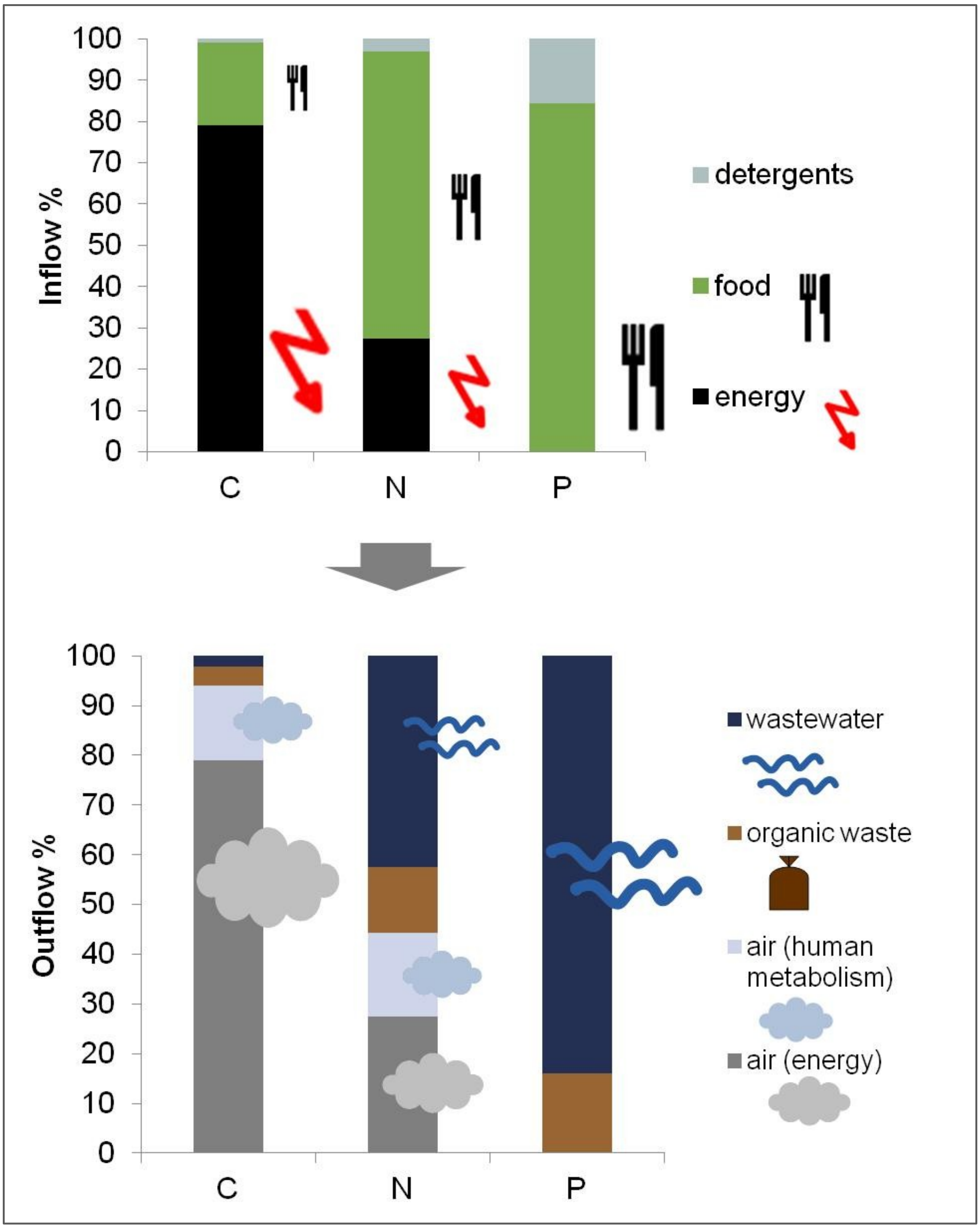


Figure 3-17: Model city: Summary of CNP flows to and from households

Legend: Overview of flows of CNP entering the household on a daily base related to energy, water, food, and detergents (top); and leaving the households as air emissions (energy and human metabolism), in wastewater or in organic waste (bottom) [% of total analyzed flows].

For N and P, the picture looks different than for C. Here, food represents the dominant inflow and wastewater the dominant outflow. For N, food accounts for 6.7 kg/p*a, while energy related flows contribute 2.9 kg/p*a. The contribution of detergents is small (0.4 kg/p*a). A large share of N is transferred to wastewater, with a load of ~4 kg. In context of total analyzed N flows (~9.5 kg) the load in wastewater represents 42%, in context of the non energy related N flows (~7 kg) it represents 56%. The organic waste receives a considerably smaller share of N.

For P, the food also represents the dominant flow in the analyzed system, accounting for 0.75 kg/p*a P and 84% of analyzed P flows. In case of P, detergents contribute considerably to the analyzed P flows (0.13 kg/p*a, 24%) despite strict regulations in Germany. The majority of P (90% of analyzed P flows) is transferred to wastewater, while organic waste contains only a small share (<10%).

The urban water chain receives the majority of N and P from food, albeit only a fraction of the C (10%). During human metabolism, the majority of C in consumed food is emitted to air, explaining the C depletion and the unfavorable[26] C/N ratio in wastewater (see chapter 1.3.2). While wastewater contains only a fraction of food C, it can in theory fully supply the energy consumption of the infrastructures (see chapter 3.1). The organic waste contains considerably less N and P on a per person base than wastewater, despite the high collection rate. To recap, a high collection rate for organic waste is assumed for model city, while in reality only half of food waste is collected in the organic waste. The C load is higher than in wastewater and represents a considerable energy resource in analogy to wastewater.

*Urban flows of anthropogenic micropollutants (AMPs)*

While wastewater represents a large pool of N and P for recycling, the presence of anthropogenic micropollutants (AMPs) is a challenge. AMPs enter the household with products and goods such as impregnated carpets and clothes, electronics, cleaning products for house and body, biocides for house and garden, paints and plastics. Together with the built environment, these products and goods represent the urban stock of AMPs.

The exact pathways of AMP transfer to the environment are largely unknown. They may be released by abrasion, by cleaning with water or by out gassing. AMPs may accumulate in household dust which is transferred to wastewater during cleaning. Run-off from urban surfaces collected in mixed sewer systems also contribute to the AMP load of the urban water chain.

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[26] Unfavourable C/N ratio for heterotrophic organisms: not enough C to assimilate the available N. C is required as fuel and as „building blocks“ for biomass. In contrast, autotrophs can use $CO_2$ from air to assimilate N in biomass, fueled by solar enery

The load of AMPs in wastewater thus mirrors chemical use in modern society. Other pathways from household to the environment may include the household waste[27], litter[28] or diffuse emissions to land, water and air. The current technical setup of the urban water chain is not effective against AMPs due to their low biodegradability. Based on their biochemical characteristics, AMPs are transferred to sludge or remain in effluent. It is estimated, that the effluent contains 100 - >1000 different AMPs in concentrations in the ng to µg range (Schluep *et al.* 2006). Assuming that 10-50% of the effluent load of WWTP can be attributed to AMPs, the load to water via this pathway adds up to 50 g to 500 g per person and year. For model city, this means a load of 1 t to 10 t of AMPs to water via the effluent. For the sludge, a similar or even higher load can be expected. Given the large pool of potential AMPs with 100 000 different chemicals used in the EU, 30 000 of them in amounts larger 1 t, this estimate seems plausible (Schluep *et al.* 2006).

For this study, PFOS was chosen as a model substance to discuss the problem of AMPs. PFOS can be regarded as a prime example to illustrate the chemical pollution problem. It is exclusively of anthropogenic origin and not formed in nature (UBA 2007). It is a perfluorinated substance and there are no known degradation mechanisms under environmentally relevant conditions (Buser and Morf 2009). Since start of production in the 1960´s, it has reached a worldwide distribution. Since 2008, PFOS is banned in the EU. Despite the ban, PFOS is still emitted from long lived products in the so called urban stock, for example carpets. In an EU wide survey of rivers, the JRC detected PFOS in more than 95% of samples, underlining the ubiquitous distribution of this AMP. A related study found a load of 27 µg/p*day (10 mg/year) in EU rivers. The contribution of wastewater to the load in rivers is unknown, but studies in Switzerland and Germany found WWTPs the major source of river pollution with per-and polyfluorinated substances (Huset *et al.* 2008 and Becker *et al.* 2008).

For model city, the load of PFOS to the environment is estimated with 20 mg/p*a. This includes the 10 mg/p*a emitted to water, as found in a monitoring study of rivers in the EU (Pistocchi and Loos (2009). The load of PFOS[29] in wastewater is estimated with 14 mg/p*a, representing 70% of the total load. With the reported partitioning factors for the "standard

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[27] Household waste is transfered to landfills or incinerated. Possible pathways from landfills: leachate (usually treated on site or transfered to WWTP) and air emissions.

[28] Waste accidentilly or illegally disposed of in the environment, including „wild" landfills

[29] Including precursor substances that are degraded to PFOS during wastewater treatment (Buser and Morf 2009)

treatment" (Buser and Morf (2009), Huset *et al.* 2008 and Becker *et al.* 2008), effluent from WWTP contributes 50% to the load of PFOS to water (25% of the total load).

During "standard treatment", PFOS accumulates in sludge (65% of the incoming PFOS) and is thus removed from effluent. The load of PFOS in sludge is 9 mg/p*a, representing 45% of the total load. If sludge containing PFOS is applied to land, quality of soil deteriorates and it becomes a secondary source of emission to water. If sludge is incinerated, the Carbon-Fluorine bond is broken and PFOS is degraded[30].

As the effluent (and sludge) contains 100 - >1000 different AMPs besides PFOS, cocktail of AMPs is a growing concern for WWTPs in Germany: a 4$^{th}$ treatment stage for effluent, as recently introduced in Switzerland, is discussed; sludge use on land has shown a decreasing trend in the last years due to concerns about soil contamination (UBA 2012). While the concentrations of individual AMPs in effluent is in the ng to µg range, the total load adds up to 50 g to 500 g per person and year.

The relative importance of wastewater as a pathway – compared to others such as industrial sources or atmospheric deposition – is uncertain for many AMPs, as it requires detailed data on use pattern, regulatory status and enforcement and environmental fate. But for the totality of AMPs in urban areas, the urban water chain is recognized as an important pathway (Schluep *et al.* 2006, Diamond and Hodge 2007, Zimmerman *et al.* 2008, Fatta-Kassinos *et al.* 2011, Ferrari *et al.* 2004a+b, Muñoz *et al.* 2009a+b, Bolong *et al.* 2009).

In contrast to wastewater or sludge from wastewater treatment, organic waste contains less AMPs, but also considerably less N and P. AMPs may also be present in organic waste, for example remainders from agrochemicals, conservation agents, or wrongly disposed (household) chemicals. But as the food was destined for human consumption, low loads can be expected.

30 Other AMPs, especially heavy metals remain in the ashes or the air filter material.

*Flows of CNP and AMPs in post use infrastructure systems*

Having established the inputs of CNP and AMPs to the post use infrastructure systems, their further pathways can be assessed. In model city, organic waste is treated in a composting facility. Most of the organic waste in Germany is treated by composting (70%); the remaining is treated by anaerobic digestion or other processes. The final product of organic waste treatment, compost, can be used as fertilizer and soil conditioner. During the treatment processes, most of the N and P are transferred to the final product and thus recycled (70% of incoming N and 100% of incoming P), while some C is lost to air (40% of incoming C). In relation to total input flows of N and P to households, the plant available N and P in compost represents 6% (N) and 8% (P). There for, reuse of compost from organic waste can be classified as a clean cycle, but with a low magnitude. While the quality of recycling is good, the quantity of N and P for recycling is low.

The C resources in organic waste represent a considerable energy potential in analogy to wastewater. The organic waste contains more than double the amount of C compared to wastewater. While the exploitation via biogas is no technical challenge, most of the organic waste in Germany is treated aerobically. This is mainly due to economies of scale, as the investment in digester and generator requires large mass flows to amortize in due time, as already discussed in relation to anaerobic digestion of sludge at small WWTPs. Sometimes, an integrated anaerobic treatment of organic waste and sludge is proposed. But if AMPs are present in the sludge, this contradicts the clean cycle approach. The AMPs would be diluted in the mixed substrate but nevertheless transferred to land with the agricultural application. On the other hand side, if the sludge is incinerated the nutrients from the "clean" substrate organic waste are lost for reuse.

The algae + case represent a synergy that follows the clean cycle approach. The $CO_2$, which would otherwise be lost to air during treatment in the composting facility, can be used as additional supply to algae systems at WWTPs.

For the wastewater and sludge management, 4 different cases are analyzed for model city. Looking at the flows of CNP and the fate of AMPs, there are considerable differences between the 4 cases.

The first case "basic urban water chain" corresponds to the basic case as analyzed in chapter 3.1, without anaerobic digestion and with agricultural reuse of sludge. In the basic case, N and P are recycled by land application of stabilized sludge. Compared to compost (6% of N and 8% of P), the recycling rate for sludge is considerably higher: 14% (N) and 47% (P). While beneficial from an energy and resource (P) perspective, recycling of NP in sludge to agricultural lands holds the risk of chemical pollution of soil resources, as sludge may also contain many AMPs. For PFOS, 65% of the load in wastewater accumulate in sludge, resulting in 9 mg/p*a transferred to agricultural soils. Thus, every kg of P recycled also introduces more than 10 mg of PFOS to agricultural soils. While the application on soils represents a large dilution, this cannot be classified as a clean cycle. While the quantity of recycling is good - nutrients in sludge from wastewater treatment represent a large pool for recycling - the quality of recycling is low due to the emission of AMPs to soil. Compared to nutrient recycling via compost from organic waste, the quantity of nutrients is higher for sludge but the quality of recycling is lower.

The second case "best available technology" refers to an optimistic scenario with anaerobic digestion and sludge incineration. The energy balance for the best available technology case is well above the German average due to minimized energy losses and maximized electricity production (see chapter 3.1). Sludge is incinerated and the nutrients are lost for reuse. But incineration also degrades the PFOS contained in sludge (65%, right side of

Figure 3-18). As there is no emission via sludge, the emission of PFOS to the environment is considerably lower than in the basic case. But the effluent load of PFOS is the same as in the basic case (35%).

The third case "algae" is based on the "best available technology" case and employs algae systems with full $CO_2$ recycling as described in chapter 3.2 (without post treatment). The required area is 6 m²/p, totaling 12 ha around the WWTP. To recap from chapter 3.2, a large fraction of N and P is recycled between the algae systems and the anaerobic digestion before leaving the system boundary of the WWTP via air emission, effluent or sludge (left side of

Figure 3-18).

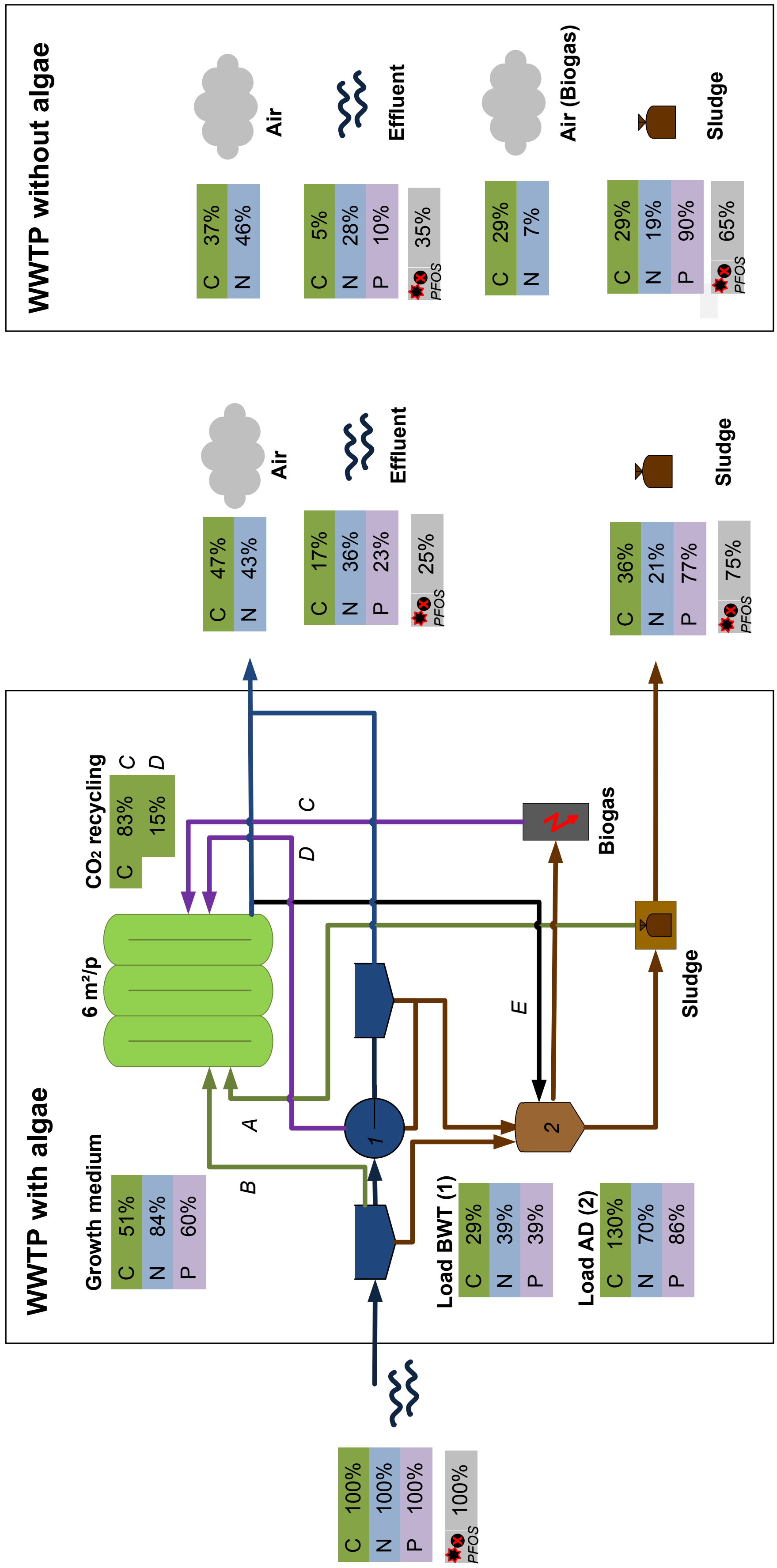


Figure 3-18: Model city: Flows of CNP and PFOS for WWTP with algae

Legend: WWTP with algae systems (left): Internal cycling via growth medium (flow A+B) and CO2 reuse (flow C+D). Harvested biomass (flow E) is transferred to AD to contribute to biogas generation. Emission to air – water – land shown as % of incoming load. WWTP without algae systems (right): emission to air – water – land as % of incoming load for comparison.

The reuse rates for N and P with the growth medium (flows labeled “A” and “B” in

Figure 3-18) are high (84% for N, 60% for P). They are in the same range (P) or higher (N) compared to sludge application on agricultural land and thus much higher than for compost from organic waste. But in contrast to sludge application on agricultural lands, there is no emission of AMPs to soil resources during biomass cultivation with algae systems. PFOS in sludge is degraded during incineration. Thus, sludge incineration is necessary to make algae systems clean cycles. But algae systems increase the amount of PFOS that is degraded during incineration. In combination with sludge incineration, algae systems are clean cycles with a high magnitude.

While the recycling of CNP for biomass generation in algae systems increases the bio-electricity generation, it has a negative effect on the effluent quality. While limit values are met, the load of “misplaced resources” CNP in effluent increases (in absence of post treatment). But looking at the load of anthropogenic micropollutants (AMPs) gives a totally different picture: effluent quality related to AMPs can be improved by the integration of algae systems.

For the standard technical setup, 35% of PFOS in wastewater remain in effluent and 65% are transferred to sludge, representing a load of 4.9 mg/p*a and 9.2 mg/p*a (see

Figure 3-18). Processes during algae growth can increase the elimination of PFOS from effluent. Due to intense contact to cell surfaces capable of bio-sorption during a long hydraulic retention time of 3-6 days in an aerated environment, the transfer of PFOS to biomass and ultimately sludge can be increased compared to activated sludge process (Mallick 2002, Munoz and Guieysse 2006, Borde *et al.* 2003 Arranz *et al.* 2008).

For model city, it is assumed that 85% of PFOS are adsorbed to biomass during algae cultivation, compared to 65% during conventional wastewater treatment (BWT). As this value is based on laboratory studies, it is subject to large uncertainties. Combined with an HE of 88%, the algae systems reach an elimination efficiency from effluent of 75%. The resulting effluent load is 3.5 mg/p*a, compared to 4.9 mg/p*a for the standard treatment. PFOS in sludge is degraded during incineration.

Thus, algae systems have the potential to simultaneously improve the effluent quality related to AMPs as well as the energy balance (Figure 3-19). The extended energy balance shows that algae systems reduce the net electricity consumption by increasing the metabolic efficiency of the urban water chain. The metabolic efficiency of the urban water chain with algae systems is considerably higher than for the German average or for the best available technology case.

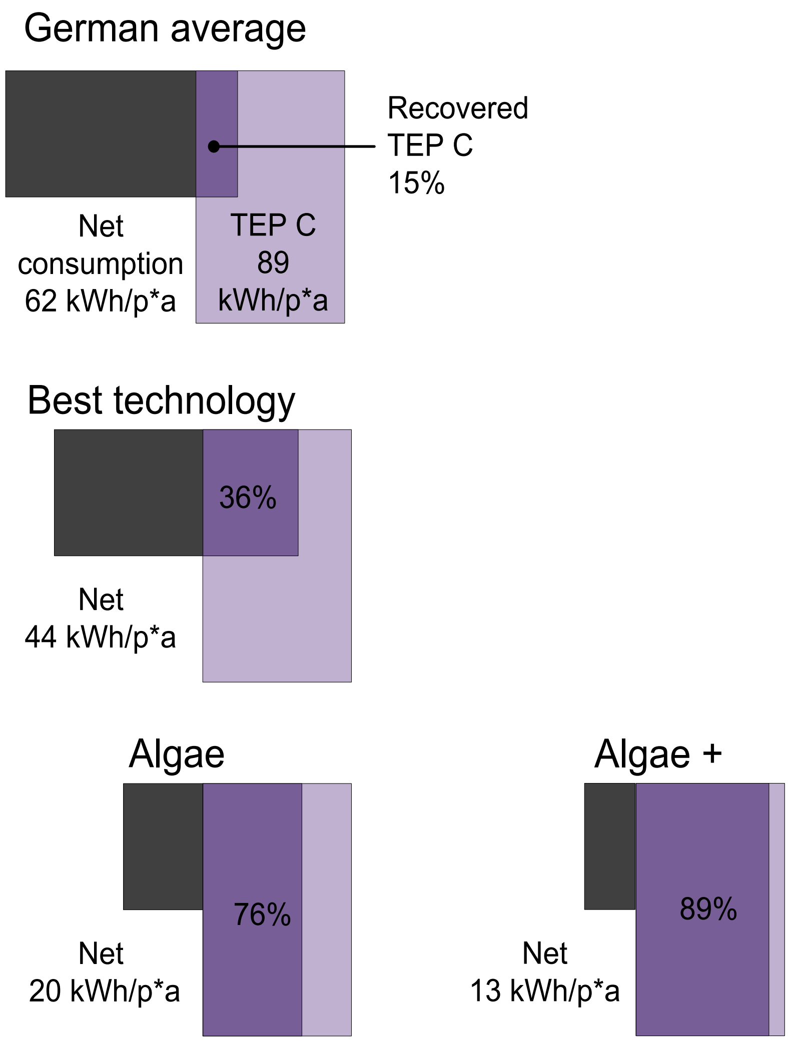


Figure 3-19: Model city: Extended energy balance of the urban water chain

Legend: The extended energy balance for the german average and the cases: best available technology, algae and algae+. A: gross consumption of electricity for handling of flow streams (grey); B: own generation of electricity from the resources in the flow streams (dark purple); C theoretical potential for electricity generation based on the resources in the flow streams (light purple); B/C: metabolic efficiency.

The metabolic efficiency of the urban water chain can be further increased with an additional $CO_{2,}$ as C is the limiting factor for biomass production. For the fourth case “algae +”, the algae systems receive additional $CO_2$ from the composting facility during growing season (ca. 4 kg C/p*a). The transfer of $CO_2$ requires the close proximity of facilities for example the organic waste treatment on site of the WWTP. With additional $CO_2$, the cultivation area can be extended to 9 $m^2$/p (18 ha) with further improvement of the energy balance (algae + case in Figure 3-19).

With increased biomass generation, post treatment is required to achieve limit values for effluent (see chapter 3.2). Post treatment reduces the load of CNP in effluent to values comparable to the WWTP without algae systems, as the majority of the non-harvested biomass is removed from effluent. Post treatment also increases the elimination of PFOS from effluent. Assuming that 95% of the non-harvested biomass is removed from effluent with post treatment, gives an elimination rate of 80%. Thus, 2.7 mg/p*a remain in the effluent and 11.2 mg/p*a is transferred to sludge and degraded during incineration. The energy required for post treatment is included in the energy balance of the algae + case (Figure 3-19).

The algae + case represent a synergy for organic waste and wastewater treatment that follows the clean cycle approach. The $CO_2$, which would otherwise be lost to air during treatment in the composting facility, can be used as additional supply to algae systems at WWTPs. The “clean” substrate organic waste does not receive AMPs from wastewater.

For model city, not only the quantity, but also the quality of CNP recycling in algae systems is important. PFOS is included as a model substance for AMPs in this study. Despite the large uncertainties related to the flows and fate of PFOS, the results highlight two important aspects. Firstly, there is no emission of AMPs during cultivation, in contrast to „open” application of sludge from agriculture. Thus, algae systems represent clean cycles. Secondly, processes during algae growth can increase the elimination of AMPs from effluent. This effect is reinforced if post treatment is applied (energy requirements included in the energy balance of the algae + case). If the increased elimination works reliably under operating conditions, this provides a strong additional incentive for integration of algae systems. While other technologies for advanced effluent treatment reduce the load of AMPs in effluent at cost of increased net energy consumption, algae systems can decrease AMP load in effluent while considerably increasing the production of bio-electricity (Figure 3-20).

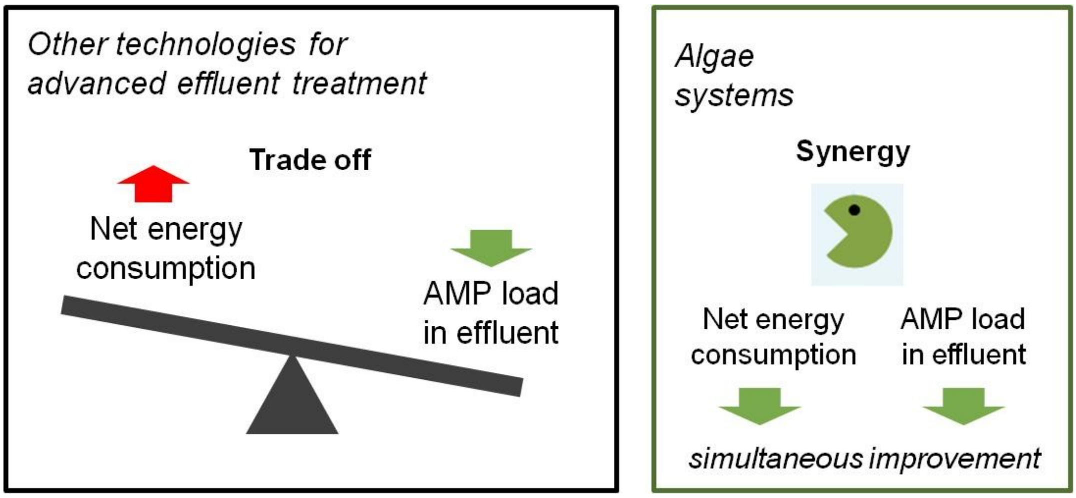


Figure 3-20: Model city: Advantages of algae systems

Legend: Advantages of algae systems compared to other technologies: Other technologies for advanced effluent treatment reduce the load of AMPs in effluent at cost of increased net consumption (left), while algae systems can simultaneously decrease AMP load in effluent and net consumption (right).

*The urban water chain in context of the urban energy balance*

With the extended scope in this chapter, the energy balance of the urban water chain – with and without algae systems – can be put in context of the urban energy balance. Household electricity consumption in model city averages 1300 $kWh_{el}$/p*a (Schmidt *et al.* 2011). Setting this electricity consumption for household in model city (26 GWh/a) as 100%, the electricity demand of the urban water chain adds 5.5% to 3% without algae systems (basic and best available technology) and 1.7% to 1.2% with algae systems (Figure 3-12). For the German average of 62 $kWh_{el}$/p*a, as calculated in chapter 3.1, the additional contribution of urban water infrastructures is 5% of household electricity consumption.

While the net electricity consumption of the urban water chain represents only a fraction of household electricity consumption on a per person base, its importance becomes obvious when seen from the city perspective (Figure 3-21). Seen from the city perspective, facilities are large single consumers. As illustrated in Figure 3-21, energy consumption is concentrated there, while households are distributed with different densities over the city area. For model city with 20 000 inhabitants, the electricity consumption for the water supply facilities equals the household electricity consumption of 400 persons (all cases). For wastewater and sludge management facilities, net consumption equals the household electricity consumption of 550 persons (German average). In fact, for many cities the WWTP is the largest single electricity consumer.

For the best available technology case, urban net consumption can be reduced by the equivalent of 230 persons; with algae systems (algae case) by the equivalent of 600 persons When additional $CO_2$ from the composting facility is diverted to algae systems (algae + case), by the equivalent of 704 persons.

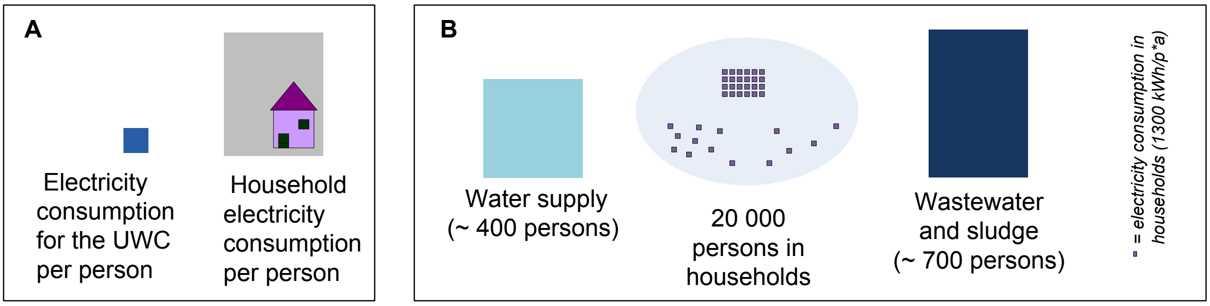


Figure 3-21: Urban energy balance: Electricity demand for the urban water chain compared to household demand and spatial distribution

Legend: Electricity demand for the urban water chain vs. household electricity demand on a per person base (left A) and seen from city perspective (spatial distribution; right B)

The improved energy balance of the urban water chain is due to the increased production of bio-electricity (Figure 3-19). The integration of algae systems considerably increases energy recovery from biogas and to a lesser extent from sludge incineration. This increased bio-electricity generation can contribute considerably to the electricity production from renewable sources (currently 20% of total electricity production in Germany). Setting the per person share of current renewable electricity production in model city to 100%, the urban water chain with integrated algae systems can contribute 23% to 29% additional bio-electricity. To recap, this is achieved by recycling CNP from wastewater in algae systems, without external input of water or fertilizer as required by other energy crops. In contrast, the best available technology case employs optimized anaerobic digestion and sludge incineration, but only contributes 12% additional bio-electricity. In the basic case, there is no generation of bio-electricity.

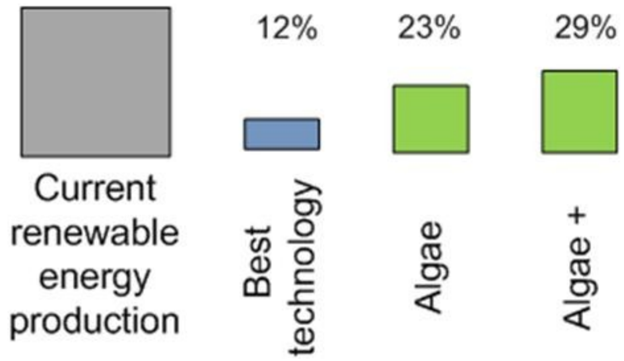


Figure 3-22: Urban energy balance: Potential contribution of the urban water chain to renewable energy production

Bio-electricity even holds a special role within the renewable energy sources. The profile of electricity generation over time is steady (basic load). For many other renewable energy sources, such as sun and wind, the electricity generation depends on external factors which cannot be controlled. Furthermore, biogas and dried sludge can be stored, thus the electricity can be produced when demand is high and production can be reduced when the demand is low. In contrast to electricity from sun, wind and water, bio-electricity represents a tunable energy. Thus, an increased share of bio-electricity is beneficial for the (urban) energy system beyond the sheer number of kWh produced, as it can contribute to cover peak demand and balance out electricity generation and demand (Schmidt *et al.* 2011).

*Excursion: Comparison of algae systems integrated in the urban water chain with alternative systems for bio-electricity production.*

Given the importance of bio-electricity for sustainable energy systems, the question arises how clean cycles in algae systems compare with alternative systems for bio-electricity production. While a full comparison of algae systems integrated in the urban water chain with alternative systems is outside the scope of this study, some important aspects are highlighted here. Compared to other energy crops, the algae systems integrated in the urban water chain require a smaller area to provide the equivalent amount of bio-electricity (Wijffels and Barbosa 2010, Colosi and Clarens 2010, Miller 2011). Furthermore, they do not require fertile soils. Looking at the water use, there is no external water demand from algae systems, while other energy crops may require irrigation.

The higher areal productivity of algae systems compared to other energy crops comes at the cost of higher N requirements (Miller 2011). While algae systems require more N per unit biomass, they allow the reuse of nutrients N and P from wastewater in closed systems. Thus they require no external input of N or P. In contrast, for other energy crops, the majority of N and P may come from mineral fertilizer. On the emission side, fertilizer application on agricultural soils causes diffuse emission of N and P to water by leaching or erosion (particle bound transport). Groundwater pollution (N) is also an issue accompanying energy crops in intensive farming systems. These emissions have to be added to the emissions from the WWTP (without algae). In contrast, for the integrated algae systems the CNP emissions to water are accounted for in the WWTP emissions. While the load in effluent is slightly higher than without algae systems (in absence of post treatment), it meets the limit values.

Looking at the AMPs, there are no emissions of AMPs during cultivation and potentially a reduced load from WWTP to the environment for the integrated algae systems. In contrast, for the other energy crops, the higher emissions of the WWTP without algae have to be accounted for plus potentially additional emission of AMPs due to application of agro chemicals such as herbicides, pesticides or fungicides. To conclude, clean cycles in algae systems are an interesting alternative to other energy crops as they offer synergies on the supply side and the emission side including AMP emission.

*Excursion: Urban nutrient flows in a wider context.*

While the urban water chain foremost represents a water infrastructure, transporting large mass flows of water through urban areas, it can also be regarded as a part of the urban “nutrient infrastructure”. The food supply system is rarely perceived as an infrastructure system, despite its managing of large volume of flow streams. Therefore also the tight connection to the urban water chain is often overlooked. The majority of N and P in the consumed food are transferred to the urban water chain, underlining the importance of nutrient reuse from wastewater for a sustainable urban metabolism.

Within the flows analyzed in this study, food clearly contributes the majority of N and P. But also in context of the full urban metabolism, food arguably represents a major flow of N and P. Other large urban sources may include industrial sources, for example the chemical industry: basic chemicals, fertilizer and detergent production and the food processing industry such as slaughterhouses, cheese making and breweries; or urban agriculture and gardening (Leach et al. 2012, Fissore 2011). While a full assessment is outside the scope of this study, some considerations about upstream burdens of food production are included here to assess the magnitude of N and P flows associated with the urban food and consequently with the urban water chain.

For N, upstream burdens for various food groups are listed by Xue and Landis (2010). For P, upstream burdens of factor 2 for plant products and factor 2.6 for animal products were reported (Cordell *et al.* 2009). Applying these factors to the N and P content of food for model city gives an upstream burden of 28 kg/p*a for N and 1.8 kg/p*a for P (Figure 3-23). Thus, even if the full food supply for model city is produced within the city boundaries, the N and P in food would still represent a considerable portion of the flows: 19% and 29% of the total flows for N and P. As the area demand for the full food supply is estimated with 2500 m²/p (SRU 2012), it is unlikely that food is fully produced within the city boundaries.

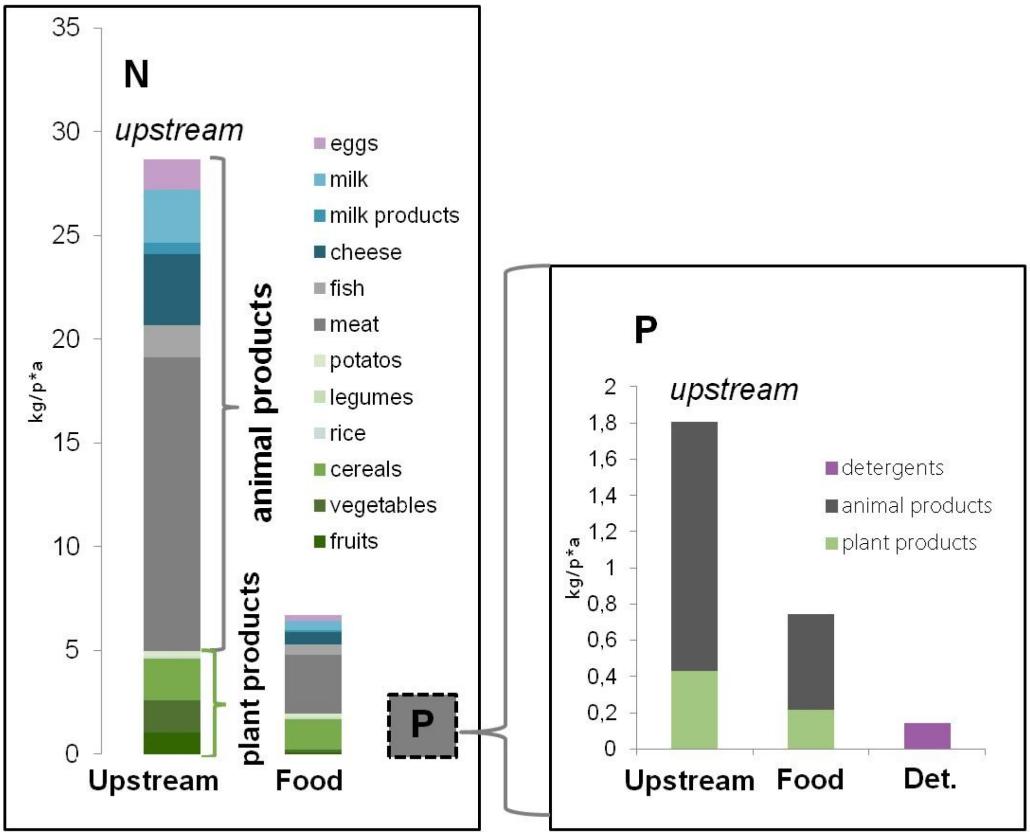


Figure 3-23: Urban nutrient flows: Upstream burdens for food supply

Legend: Upstream burdens for N and P for full food supply for model city and N and P content of food [kg/p*a]. P content of detergents shown for comparison.

Looking at the full nutrient pathway through the agrosphere and astysphere shows that nutrient reuse is important for both: for the supply chain including agriculture, animal husbandry[31] and food processing and distribution; as well as for the disposal chain including wastewater and to a lesser extent organic waste.

For another comparison, one can consider that the average fertilizer consumption in Germany is 19 kg N and 1.3 kg P on a per capita base (on an elemental base, destatis 2012) for an available agricultural area of 2100 m²/p – slightly lower than the area required for the full food supply (SRU 2012). Thus, the amount of N and P in food represents 35% and 54% of the average fertilizer consumption in Germany. These comparisons underline the magnitude of N and P flows associated with the urban food and consequently with the urban water chain.

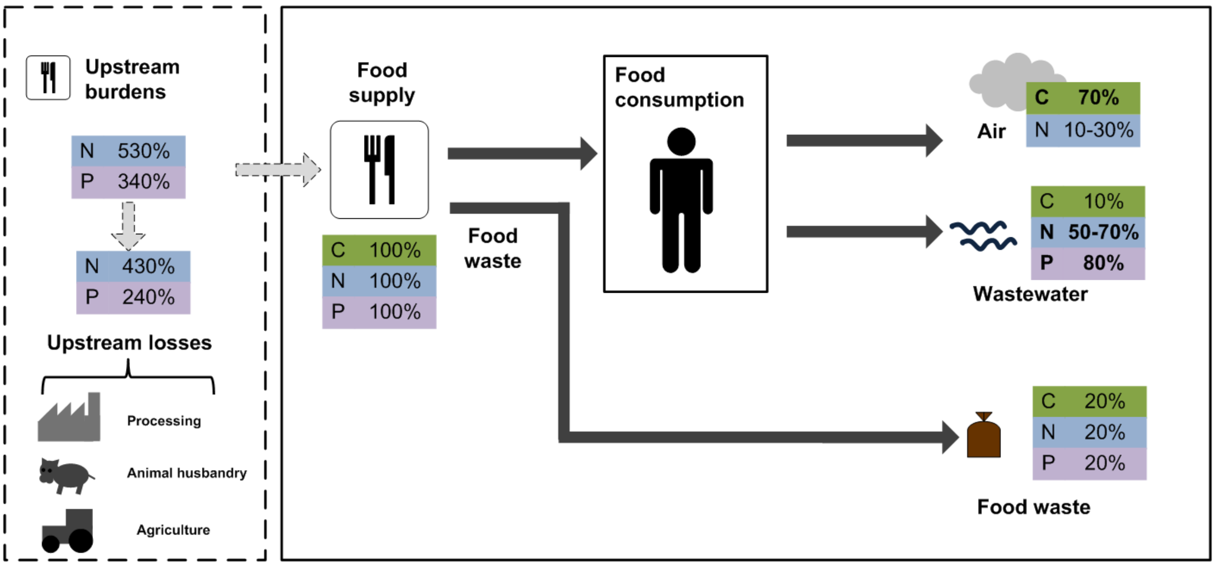


Figure 3-24: Urban nutrient flows: Upstream burdens and downstream emissions

As illustrated in Figure 3-24, the urban water chain receives the majority of N and P from food: 50-70% of N and 80% of P from food are transferred to ww. While the pool of N and P for recycling in wastewater is smaller than the upstream burdens associated to the food sup-

[31] Concerning animal husbandry, it is noteworthy that the animal metabolism is no different than the human metabolism concerning the flows of CNP. But the animals are kept in an active growth / biomass producing state so that a much higher fraction of CNP is diverted to the biomass (meat, eggs, milk) than in the average human. But also for animals, the majority of N and P from feed is transferred to manure and can be recycled. Whether and which AMPs are present depends on the agricultural system.

ply, it still represents a considerable fraction of the total flows and is thus an important starting point for N and especially P reuse.

*Relevance of the urban water chain for the urban metabolism*

The analysis showed that clean cycles in algae systems can contribute to a sustainable urban metabolism in several aspects. Figure 3-25 gives a summary of the topics discussed above and grades the performance of the urban water chain for different indicators, with and without algae, as well as the relevance of the indicator for the urban metabolism.

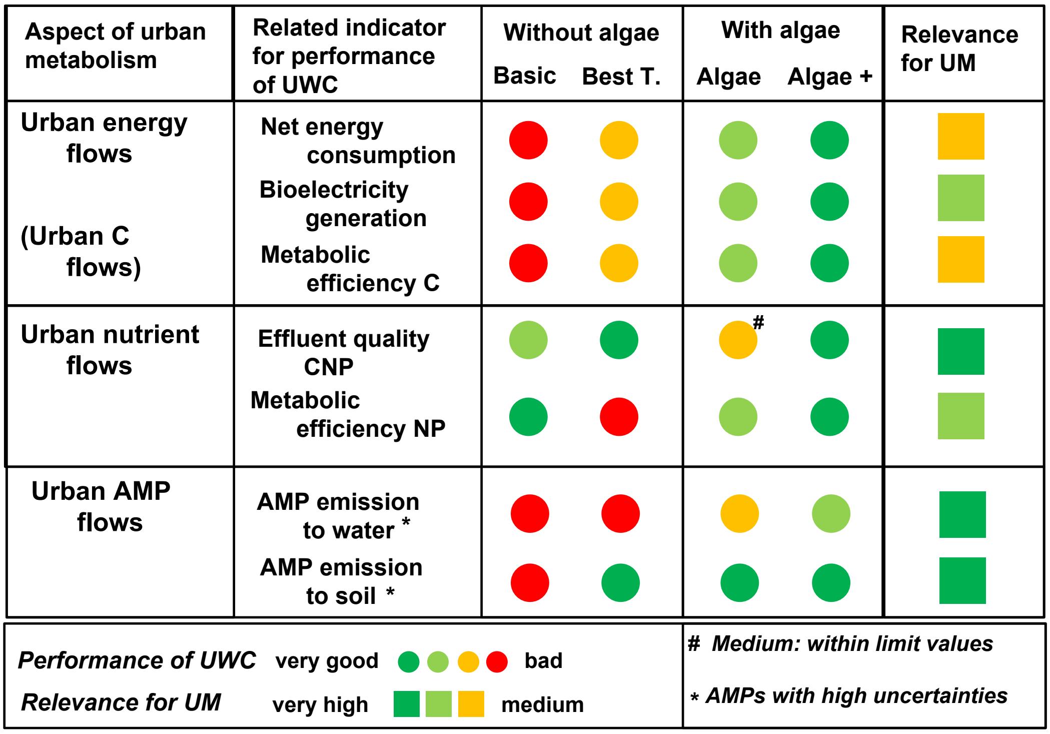


Figure 3-25: The relevance of the urban water chain in context of urban metabolism

Firstly, in context of the urban energy balance, the consequent energetic reuse of resources from wastewater in algae systems can considerably contribute to electricity production from renewable sources on an urban scale. Bio-electricity is an important pillar for sustainable energy systems as it covers base loads. The net consumption of the urban water chain on a per person base is rather low compared to total electricity consumption in households. Despite this, the spatial concentration and potential contribution to bio-electricity production make the urban water chain with algae systems an important player for the transition towards a sustainable urban energy system.

Secondly, in context of the urban nutrient flows, the recycling of the resources of N and P in wastewater is highly relevant for a sustainable urban metabolism. These flows represent a large share of the urban nutrient flows and are also important with a wider perspective on the food supply. But as the urban water chain is also a major pathway of anthropogenic micropollutants (AMPs), clean cycles are required. Urban flows of AMPs are the third important aspect to consider. In algae systems, N and P can be recycled despite the presence of AMPs. There is no emission of AMPs to the environment during cultivation in closed systems, in contrast to”open” agricultural applications. Their closed nature makes algae systems clean cycles. Furthermore, algae systems can reduce the load of AMPs to the environment. In that sense, the area designated to algae systems has a double function: to produce biomass for electricity generation and to allow for a long hydraulic retention time for the wastewater treatment. Taken together, the synergies offered by the integration of algae systems in the urban water chain can contribute considerably to a sustainable urban metabolism.

# 4 Conclusion

The analysis showed that algae systems integrated in the urban water chain can contribute to a sustainable urban metabolism in several aspects. They use resources for bioenergy production which would otherwise be wasted. Thereby, they increase the metabolic efficiency compared to the current situation. At the same time, they improve the emission balance regarding AMPs. Thus, algae systems are clean cycles and potentially even “cleaning cycles”, reducing the emissions of AMPs during recycling. The methodology applied in this study allowed for assessing these synergies and their relevance in context of urban metabolism.

In the first part of the study, the status quo of the urban water chain in Germany was assessed (chapter 3.1). The results show the low metabolic efficiency of the urban water chain today. For C, the metabolic efficiency for C is below 25%, even for facilities employing biogas combustion and sludge incineration. Even with the best available technology, it is below 40%. The non reused energetic potential is large compared to brut consumption. In theory, bio-electricity from C resources can fully supply the energy demand of the urban water chain.

For N and P, the metabolic efficiency of the urban water chain is also low: 20% for P and 4% for N. The concept for reuse of N and P currently employed is the application of sludge generated during wastewater treatment on agricultural land. While the sludge contains considerable amounts of nutrients, especially P, agricultural reuse is decreasing in Germany due to concerns about chemical pollution of soils (UBA 2012).

While the focus lies on the post use side of the urban water chain, the scope of the study includes the full pathway of water through settlements: from sourcing of water to treatment and provision of tap water for water use in households, to transport and treatment of wastewater and sludge. Due to the extent and complexity of the system, there are important limitations. Data for the individual stages of the urban water chain were compiled from statistics and various sources in literature. The analysis also has to rely on assumptions as for some stages of the urban water chain, especially for sludge handling, data availability is low. A detailed account is found in the methods section.

In this study, three cases were used to represent wastewater and sludge treatment in Germany. In reality, every WWTP is different and there are many particularities in process design and associated energy consumption and generation. An SFA was used to assess the current energetic reuse of N and P, to analyze flows and fate of C and to quantify on site $CO_2$ emissions. The SFA method is inherently subject to uncertainties. Influent loads and partitioning factors are average empirical values, which are subject to large variations in reality. Partitioning factors can only reflect tendencies of elemental behavior within a complex system.

The theoretical energy potentials estimated in this study mark the upper limit of energy harvesting, constrained only by resource characteristics. They provide no information about the technical feasibility of increased energy harvesting from flow stream resources (technical potential), and the related costs (economic potential), which are reserved for future studies. Despite the limitations, the applied methodology provided a holistic picture of the status quo of the urban water chain in Germany.

Based on the status quo as baseline for comparison, a concept for increased reuse of CNP is assessed: the integration of algae systems at WWTPs (chapter 3.2). A technical setup is proposed in this study. It relies solely on the resources available on site, with no external input of fertilizer, water or $CO_2$ required. This study provides the first detailed description of integration of algae systems; including a SFA of CNP and the implications for energy and emission balance (see also Menger-Krug *et al.* 2012).

The results show that it is feasible from a flow stream perspective to produce enough bioelectricity from algae systems, to run WWTP energy-neutral during the vegetation season or even turn them into net energy producers. This can be achieved with nutrients from wastewater, without any external resource input. C *resp.* $CO_2$ availability is the limiting factor for yield with the proposed process design *i.e.* in absence of external $CO_2$ sources.

While intensive C recycling in algae systems considerably improves the energy balance, it also impacts on effluent quality, mainly *via* the contribution of non-harvested biomass. The harvesting efficiency is identified as a technical key parameter at the crossroads of energy balance and effluent quality. Post treatment is highlighted as an opportunity to reliably meet effluent limit values for COD, N and P. Besides reliably meeting effluent limit values for COD, N and P, adding post treatment also improves the effluent quality in terms of AMPs.

Due to the prospective nature of the system under analysis, there is no empirical data for many key parameters, such as nutrient uptake efficiencies, areal productivity, harvesting efficiency and anaerobic digestibility. Instead, the analysis had to rely on data from pilot applications and laboratory studies, which remain to be confirmed or rejected in practice. Ranges of values from literature were used in a scenario analysis highlighting the key factors for the performance of algae systems at WWTPs. The influence of algae systems on the energy demand of other processes at the WWTP was assessed based on SFA results and the validity of the applied proxies remains to be proven in practice.

While this study has shown the feasibility of the concept from a flow stream perspective, many other aspects require analysis on the way to implementation. This includes acceptance and social aspects, as well as political and economical aspects. For the latter, the future developments of energy costs - for fossil and renewable energy which again depend on the political framework - are important aspects to consider.

To put the potential improvement with algae systems in context of the urban metabolism, the scope of the study is extended (chapter 3.3). The extended perspective includes the flows that represent the connection points between urban water chain and the full urban metabolism. The system boundaries include the daily household consumption of energy, water, food and cleansing products. The study traces the pathways of CNP: the input, the transformations during human metabolism, the transfer to wastewater and organic waste infrastructures and the fate during the treatment processes and the emissions to air. While information on the bulk flows are available from official sources (BMELV, UBA, AGEB), this is the first study to quantify CNP flows associated to household consumption in Germany, albeit only for a semi hypothetical model city with high uncertainties.

Besides CNP, also the flows of PFOS are included in this study. Including this notorious AMP into the analysis of CNP recycling, serves as a starting point to discuss the AMP problem and the quality of recycling. But for a full picture many more AMPs (>1000) with different use patterns, biochemical characteristics and toxicological end points – as well as the effect of mixtures - need to be included. While the AMP problem, especially for wastewater and the necessity for clean cycles is used as the base of the argument, the results of this study highlighted two important aspects. Firstly, algae systems can provide bio-electricity without emission of AMPs during biomass cultivation. Thus, they can be characterized as a clean cycles. “Closed” algae systems fulfill the requirements for nutrient recycling from the urban water chain –given the presence of AMPs - in contrast to „open” application of sludge in agriculture. Secondly, processes during algae growth can increase the elimination of AMPs from effluent. If the increased elimination works reliably under operating conditions, this provides a strong additional incentive for integration of algae systems. Besides the fate of AMPs in algae systems, there are other important research needs. This includes $N_2O$ emission from N in wastewater and the possible formation of algae toxins during biomass growth.

For algae-to-energy systems integrated in the urban water chain, the results of this analysis warrant further research on the scale of pilot applications. The developed model of substance and energy flows of the integrated system provided information on energy flows, on nutrient recycling within the system, on loads to the individual treatment steps and on loads to the environment. This information is useful to design pilot projects. Data gathered from pilot projects can in turn refine the model. In that sense, the model presented in this study can be used as a tool for system design and optimization.

For this study, a framework for analysis of water infrastructures was developed: the extended energy balance. It proofed a useful tool to analyze the integration of algae systems, as it assesses the metabolic efficiency in addition to the external energy flows. As required by the clean cycle approach, it includes an emission balance covering all environmental compartments. This framework for analysis of water infrastructures can also be useful for evaluation of other reuse oriented concepts.

On the way towards a sustainable future, humanity faces many challenges. Given the high share of humanity living in cities, the urban metabolism needs to be reorganized from its present linear form towards a higher metabolic efficiency and clean cycles. Cities need to restructure their resource consumption and energy systems to negotiate the human impact on their hinterlands and ultimately on the planetary boundaries. Human activity in its present form highly alters the global cycles of CNP. Given the risks associated with a transgression of the planetary boundaries, a more sustainable management of these flows also in urban areas is required. This includes a sustainable, C efficient and renewable energy supply, as well as recycling of resources from food and other consumption related CNP flows. At the same time, cities need to find ways to minimize pollution of the environment. This includes CNP as misplaced resources, but also AMPs, as the chemical pollution problem is another planetary boundary. While algae systems cannot provide the single solution for any of these challenges, the synergies can contribute to solving all of them.

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