



Deliverable 4.3 Decision support guideline based on LCA and cost/efficiency assessment

Larsen, Henrik Fred; Hansen, Peter Augusto; Boyer-Souchet, Florence

Publication date:
2010

Document Version
Publisher's PDF, also known as Version of record

[Link back to DTU Orbit](#)

Citation (APA):
Larsen, H. F., Hansen, P. A., & Boyer-Souchet, F. (2010). *Deliverable 4.3 Decision support guideline based on LCA and cost/efficiency assessment*. EU FP6 project, deliverable No. 4.3 http://www.eu-neptune.org/Publications%20and%20Presentations/index_EN

General rights

Copyright and moral rights for the publications made accessible in the public portal are retained by the authors and/or other copyright owners and it is a condition of accessing publications that users recognise and abide by the legal requirements associated with these rights.

- Users may download and print one copy of any publication from the public portal for the purpose of private study or research.
- You may not further distribute the material or use it for any profit-making activity or commercial gain
- You may freely distribute the URL identifying the publication in the public portal

If you believe that this document breaches copyright please contact us providing details, and we will remove access to the work immediately and investigate your claim.

NEPTUNE

New sustainable concepts and processes for optimization and upgrading municipal wastewater and sludge treatment

Contract-No. 036845

A Specific Targeted Research Project
under the Thematic Priority 'Global Change and Ecosystems'

Work Package 4 · Assessment of environmental sustainability and best practice

Deliverable 4.3 - Decision support guideline based on LCA and cost/efficiency assessment

Due date:	15.04.2010
Actual submission date:	31.08.2010
Start date of project:	01.11.2006 Duration: 41 months
Deliverable Lead contractor: (Organisation name)	DTU
Participant(s) (Partner short names)	DTU, IPU
Author(s) in alphabetic order:	Henrik Fred Larsen, Peter Augusto Hansen, Florence Boyer-Souchet
Contact for queries:	Henrik Fred Larsen Technical University of Denmark (DTU) Building 426, room 110 DK-2800 Lyngby Denmark Phone: +45 45 25 47 29 Fax: +45 45 93 34 35 e-mail: hfl@man.dtu.dk
Dissemination Level:	
<u>P</u> ublic,	
<u>PP</u> Restricted to other Programme Participants,	PU
<u>RE</u> stricted to a group specified by the consortium,	
<u>CO</u> nfidential only for members of the consortium)	

Contents

Contents	2
1 Objectives of NEPTUNE WP4	5
2 Introduction	6
3 Goal and scope	7
3.1 Goal	7
3.2 Functional unit	7
3.3 Scope	7
4 Waste water technologies investigated	9
4.1 CLUSTER 1 technologies; post treatments – short description	10
4.2 CLUSTER 2 technologies; in-line treatments – nutrient removal	11
4.3 CLUSTER 3 technologies; sludge inertization–short description	12
4.4 CLUSTER 4 technologies; sludge triage – including disintegration	14
5 Inventory	16
5.1 Physical inventory: Cluster 1	16
5.1.1 Ozonation	16
5.1.2 Sand filtration.....	19
5.1.3 PAC addition.....	21
5.2 Physical inventory: Cluster 2	23
5.2.1 Conventional waste water treatment.....	23
5.2.2 Autotrophic anaerobic ammonium oxidation (annammox).....	26
5.2.3 ICA strategies	27
5.3 Physical inventory: Cluster 3	27
5.3.1 On-site sludge incineration.....	27
5.3.2 Wet oxidation (WO) of sludge.....	29
5.3.3 High temperature pyrolysis (HTP) of sludge	31
5.3.4 Middle temperature pyrolysis (gasification) of sludge	33
5.4 Physical inventory: Cluster 4 - sludge triage incl. disintegration	34
5.5 Functional inventory	41
5.5.1 Functional inventory: Cluster 1	43
5.5.2 Functional inventory: Cluster 2.....	48
5.5.3 Functional inventory: Cluster 3.....	48
5.5.4 Functional inventory: Cluster 4.....	51

6	Life cycle impact assessment (LCIA)	52
6.1	EDIP characterization factors for pharmaceuticals and more.....	52
6.2	Environmental quantities in GaBi related to EDIP97.....	53
6.3	Note regarding normalization references.....	53
7	Running EDIP97 on GaBi 4.3/EcoInvent 2.0 integrated	54
8	Results and interpretation	56
8.1	Cluster 1: WWTTs for micropollutant removal	56
8.1.1	Ozonation.....	56
8.1.2	Sand filtration.....	59
8.1.3	PAC addition.....	60
8.1.4	Ozonation combined with sand filtration.....	62
8.1.5	PAC addition to secondary effluent combined with sand filtration.....	62
8.1.6	Including more micropollutants or conservative CFs.....	63
8.1.7	Including removal of metals and phosphorus by the sand filter.....	66
8.1.8	Including average electricity production or long-term emissions.....	68
8.1.9	Environmental sustainability assessment regarding cluster 1.....	69
8.2	Cluster 2: WWTTs for nutrient removal	71
8.2.1	Conventional WWT combined with sludge incineration.....	71
8.2.2	Autotrophic anaerobic ammonium oxidation (annammox).....	73
8.2.3	Environmental sustainability assessment regarding cluster 2.....	75
8.3	Cluster 3: Sludge inertization as disposal of WW sludge	76
8.3.1	On-site sludge incineration.....	76
8.3.2	Wet oxidation (WO) of sludge.....	77
8.3.3	High temperature pyrolysis (HTP, pyrolysis) of sludge.....	78
8.3.4	Middle temperature pyrolysis (gasification) of sludge.....	80
8.3.5	Environmental sustainability assessment regarding cluster 3.....	81
8.4	Cluster 4: Sludge triage including disintegration methods	84
8.4.1	Environmental sustainability assessment regarding cluster 4.....	85
9	Cost/efficiency assessment	89
10	Conclusions and future research needs	90
10.1	Conclusions	90
10.2	Future research needs	90
11	List of abbreviations and definitions	92
	REFERENCES	93

APPENDICES:

Appendix 1: Physical inventory data on ozonation from Hunziker (Moser 2008).....	98
Appendix 2: Physical inventory data on sand filtration from Hunziker (Moser 2008)	100
Appendix 3: Physical inventory data on PAC addition from Hunziker (Moser 2009a)	101
Appendix 4: Physical inventory data on anammox from Hunziker (Moser 2009b) ...	102
Appendix 5: Concentrations and removal rates for micropollutants and nutrients used in the LCA modeling (conv. WWT)	103
Appendix 6: Removal rates for micropollutants regarding ozonation of secondary effluent (based on Hollender (2009))	104
Appendix 7: EDIP97 fresh water ecotoxicity characterization factors (CFs) for emissions to fresh water	105
Appendix 8: Mass balance data for anammox (Miladinovic 2008a)	110
Appendix 9: Physical and functional inventory for on-site sludge incineration	112
Appendix 10: Physical and functional inventory for wet oxidation of sludge	121
Appendix 11: Physical and functional inventory for high temperature pyrolysis (HTP) of sludge.....	129
Appendix 12: Physical and functional inventory for middle temperature pyrolysis (gasification) of sludge	137
Appendix 13: Transfer coefficients for mercury to air regarding sludge inertization methods.....	146
Appendix 14: Physical and functional inventory for sludge triage	148
Appendix 15: Sensitivity analysis on PAC and electricity production.....	157

1 Objectives of NEPTUNE WP4

The main objectives of NEPTUNE WP4 include:

- Complementation of state-of-the-art life cycle assessment (LCA) to cover specific biological effects (e.g. endocrine disruption) of micropollutants and pathogens.
- Applying the LCA methodology on a variety of wastewater and sludge technologies in order to assess the environmental sustainability and best practices ranking list of
 - advanced wastewater treatment for micropollutants and pathogens removal,
 - advanced nutrient removal control methods and processes and
 - options for sludge handling and treatment
- Formulate decision support guidelines based on LCA, cost/efficiency assessment and local constraints.

These objectives are sought fulfilled by performing the following tasks:

1. Development of “new” methodology
 - a. Defining overall methodological LCA framework in agreement with INNOWATECH
 - b. Developing methodology for including potential impacts of micropollutants and pathogens in the life cycle impact assessment (LCIA) of waste water treatment technologies. Results from whole effluent testing (WET) will be included.
2. Application of the LCA methodology including the “new” LCIA methodology suited for waste water treatment technologies
 - a. Providing and generating inventory data for the included waste water treatment technologies and sludge handling techniques
 - b. Estimating characterization factors (to be used for (eco)toxic impact potentials) for included emissions (e.g. of micropollutants and pathogens) on the basis of gathered effect and fate data, and the developed extended LCIA methodology
 - c. Modeling, running and interpreting the results of the LCAs on the included waste water treatments and sludge handling methods
3. Creation of a decision supporting guideline
 - a. Describing pros and cons for the different included waste water treatment and sludge handling techniques based on the results from the LCAs and cost/efficiency analysis

The present Deliverable 4.3 reports the results of Task 3 including the supporting relevant parts of Task 2. The other tasks are all ready reported: Task 1a is included in Deliverable 4.1 (Larsen et al. 2007) and Task 1b is included in Deliverable 4.2 (Larsen et al. 2009). The part of Task 2b dealing with new methodology is included in Deliverable 4.2.

2 Introduction

The environmental sustainability assessments described in this report is based on life cycle assessments (LCA) of selected waste water treatment technology (WWTT) cases included in NEPTUNE. The basic approach in these assessments (avoided against induced impacts) is described in Deliverable 4.1 (Larsen et al. 2007) and other methodological issues (e.g. normalization and weighting) in Deliverable 4.2 (Larsen et al. 2009). The basic approach is illustrated in a simplified way for the case ozonation in Figure 2.1.

The technologies investigated include conventional WWT (reference), ozonation, sand filtration, PAC addition, anammox, sludge incineration (reference), other sludge inertization technologies (wet oxidation, high temperature pyrolysis, gasification) and sludge triage (including sludge disintegration technologies, i.e. thermal hydrolysis and ultrasound disintegration). They are divided into four clusters related to the main purpose of the treatment, e.g. nutrient removal or micropollutant removal. All are assessed within the cluster by comparing with the reference or by looking at induced versus avoided environmental impacts potentials originated from the various emissions and flows inventoried.

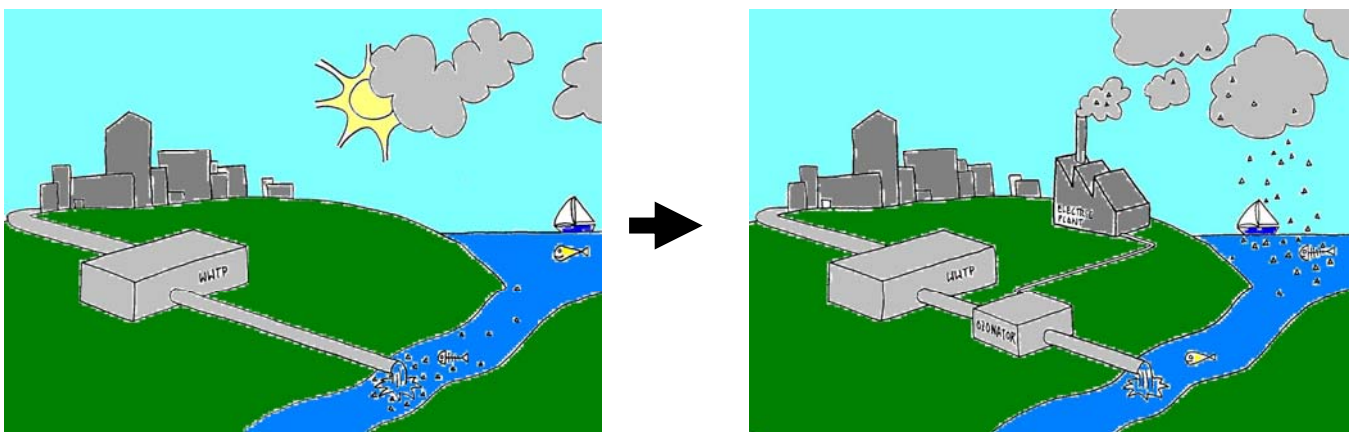


Figure 2.1 Do we induce more environmental impact than we avoid by introducing a new waste water treatment technology (sub-optimisation)?

The study is based as much as possible on data obtained directly from NEPTUNE partners for the physical and functional inventory. In particular, the functional inventory of cluster 1 on micropollutant removal, tracks 22 organic micropollutants (pharmaceuticals) that have been sampled in different WWTTs by NEPTUNE partners. The resulting inventory is then modelled according to the EDIP97 methodology on the GaBi LCA modelling software, i.e. GaBi 4 (PE 2008), with Ecolnvent 2.0 database integrated (Ecolnvent 2007).

3 Goal and scope

The present section is meant to establish the goal and scope of the study presented here. It describes the systems studied and the purpose of studying them. As such, this must be taken into consideration when assessing the results presented in subsequent sections.

3.1 Goal

The goal of the study is to address the following research questions by carrying out LCAs of selected WWTTs:

1. Is the WWTT an improvement in terms of environmental sustainability compared to conventional WWT, i.e. do we avoid more potential environmental impact than we induce by introducing the technology?
2. How does the WWT technology compare to the others in the cluster regarding environmental sustainability?

The study will address these questions regarding the WWTT included in the clusters defined in Section 4.

For those assessed technologies for which costs have been provided by NEPTUNE partners a simple cost/efficiency analysis is performed in Section 9.

3.2 Functional unit

The functional unit of all LCAs presented in this study is “*the treatment of 1 m³ of waste water containing micropollutants, nutrients and organic matter*”. For sludge the unit 1 ton dry matter (DM) will be used. The characterisation of the waste water/sludge (i.e. substances included) will be presented in Section 5 depending on the studied technology.

3.3 Scope

All LCAs presented herein this study will be evaluated according to the EDIP97 methodology (Wenzel et al. 1997) adapted to the assessment of WWTTs as described in Deliverable 4.2 (Larsen et al. 2009). The normalized results will therefore be presented with weighting factors assumed to be 1 (analogous to normalized results) and in the following impact categories:

- Acidification potential (AP)
- Global warming potential (GWP)
- Nutrient enrichment potential (NEP)
- Ozone depletion potential (ODP)
- Photochemical ozone formation potential – in low NO_x areas (POP)
- Human toxicity in soil (HTS)
- Human toxicity in water (HTW)
- Human toxicity in air (HTA)
- Ecotoxicity in water, chronic (EWC)
- Ecotoxicity in soil, chronic (ESC)

Resource consumption and waste impact categories have been omitted in this report to concentrate on the above mentioned 10 environmental impact categories. Resource consumption is a separate impact category that can't be combined with the environmental impact categories. In the case of WWT it is typically related to fossil fuel consumption and infrastructure, see Deliverable 4.1 (Larsen et al. 2007).

The waste impact categories of EDIP97 are redundant here, as waste disposal/recycling is modeled by use of the mentioned 10 included impact categories. Long-term emissions from

landfills (as defined in EcoInvent 2.0, Doka (2007a)) as well as the impact category “ecotoxicity in water acute” have only been included in special cases (sensitivity analysis) because of models deemed defective (out of proportion) at the modeling phase. This is further discussed in Section 7 and in Deliverable 4.2 (Larsen et al. 2009).

Further arguments for the selecting of impact categories may be found in Deliverable 4.1 (Larsen et al. 2007) and in Deliverable 4.2 (Larsen et al. 2009).

Geographical scope: All foreground processes (e.g. ozonation) are assumed to occur in Europe or Switzerland while background processes (e.g. extraction of fossil fuel) may be assumed to occur on a global scale, of corresponding to real practice. More information on this is available in Section 5.

Temporal and technological scope: The present LCAs are considered consequential LCAs: this means they deal with changes to a given process (WWT) and the consequences of these changes. Therefore, when applicable and according to Mattson et al. (2003), marginal technologies should be used. This was actually agreed upon during the presentation of the NEPTUNE case studies at the Varna 2008 NEPTUNE meeting and will be further discussed in Appendix 15. It is here anticipated that the marginal technology for electricity production is based on natural gas (Heijungs and Ekvall 2009, Ekvall et al. 2004).

Physical inventory: As described in Section 5.1, the physical inventory is based on data inputs from NEPTUNE partners, literature data and generic EcoInvent data for background processes especially. The scope of inventory data in general is summarized in Table 3.1.

Table 3.1 Inventory scope

Physical inventory			Functional inventory*	
Infrastructure	Ancillary	Energy	Micropollutants	Nutrients
Materials	Ancillary product	Process energy	22 organics (pharmaceuticals)	Nitrogen
Transport	Transport		9 metals	Phosphorus
Disposal				
Reuse				

* Described in Section 5.2 and depending upon actual case

Table 3.1 shows the only quantities included in the inventory (foreground data). These quantities are ideally provided by NEPTUNE partners based on direct data from the investigated WWTTs. When direct data is not available, approximations are used based on literature. Each of the physical inventory quantities is then modeled using EcoInvent data (background data) whenever possible. In case a given background process is not all ready modeled in EcoInvent, literature data is used to model it but still with existing EcoInvent background processes.

Functional inventory: The functional inventory in this study focus on 22 organic micropollutants (mostly pharmaceuticals), 9 metals, nitrogen, phosphorus and other potential pollutants in the waste water, depending on the case studies. This is explained in Section 5.2.

Removal of micropollutants and other substances by the WWTTs is modeled by use of removal rates (i.e. transfer coefficients) throughout. These removal rates (or data for estimating them) are ideally provided by NEPTUNE partners and are assumed to be fixed and modeled linearly due to insufficient data to model them otherwise. In an LCA framework this assumption is expected to hold reasonably well within the usual range of the substance concentrations in waste water.

4 Waste water technologies investigated

The technologies investigated are divided into four clusters:

CLUSTER 1: *(Post-)treatment technologies for micropollutant and pathogen removal (need “standard” WWTP effluent composition)*

- Reference: Direct emission of effluent (secondary) from a conventional WWTP
- Pulverized activated carbon (PAC) addition
 - *PAC addition in the biological step*
 - *PAC addition to the effluent followed by sand filtration*
- Ozonation and sand filtration
 - *Ozonation of the effluent*
 - *Sand filtration of the effluent*
 - *Ozonation of the effluent followed by sand filtration*

CLUSTER 2: *(In-line) updating technologies for nutrients removal (need “standard” WWTP influent composition)*

- Reference: Conventional WWT; mechanical, de/nitrification, activated sludge, P removal
- Supernatant treatment, i.e. autotrophic anaerobic ammonium oxidation (anammox)
- Instrumentation, control and automation (ICA) strategies of ammonia, nitrate and combination control

CLUSTER 3: *Sludge inertization for decreasing effect of final products disposal (need digested sludge composition)*

- Reference: On-site incineration
- Wet oxidation (WO)
- Middle temperature pyrolysis (gasification, MTP)
- High temperature pyrolysis (pyrolysis, HTP)

CLUSTER 4: *Sludge triage strategies for improving sludge contents of nutrients, reducing micropollutants and pathogens for use on agricultural land (need “standard” WWTP mixed, primary and secondary sludge composition). Combined with sludge disintegration for increasing available COD in sludge for digestion*

- Reference: mixed sludge mesophilic anaerobic digested (MAD) with biogas production, dewatered and finally incinerated
- Reference: mixed sludge MAD with biogas production, dewatered and finally to agriculture
- Primary digested sludge incinerated; secondary sludge treated by short aerobic thermophilic treatment (no biogas), dewatered and finally to agriculture
- Primary sludge incinerated; secondary sludge thermal pre-treatment, MAD with biogas production, dewatered and finally to agriculture
- Primary sludge incinerated, secondary sludge treated by ultrasonic pre-treatment, MAD with biogas, dewatered and finally to agriculture

It may be noted that some technologies may qualify for more than one cluster (such as sand filtration – also relevant for nutrient removal) but they are initially classified according to the most appropriate one as agreed at the NEPTUNE Zürich meeting in 2008. The technologies are very shortly described graphically below with reference to more detailed descriptions. The basic framework for the approach used here (including the cluster framework, and the goal and scope of the study) is further described/discussed in the master thesis by Hansen (2008) on “Conceptual framework” performed as part of the NEPTUNE project

4.1 CLUSTER 1 technologies; post treatments – short description

Figure 4.1 below shows the pilot-scale experimental set-up implemented by DPU to assess PAC (pulverized activated carbon) addition to the biological step in WWT (addition in biology). The top branch represents a conventional WWT system with addition of PAC in one of the biological reactors (the membrane is instead of secondary clarifier and sandfilter). The bottom is a conventional WWT system and serves as reference to derive the effects or functional inventory data (micropollutant removal rates) due to PAC addition only. The red dots denote the sampling points. The technology is further described in NEPTUNE “2nd Periodic Activity Report” (Ante et al. 2009)

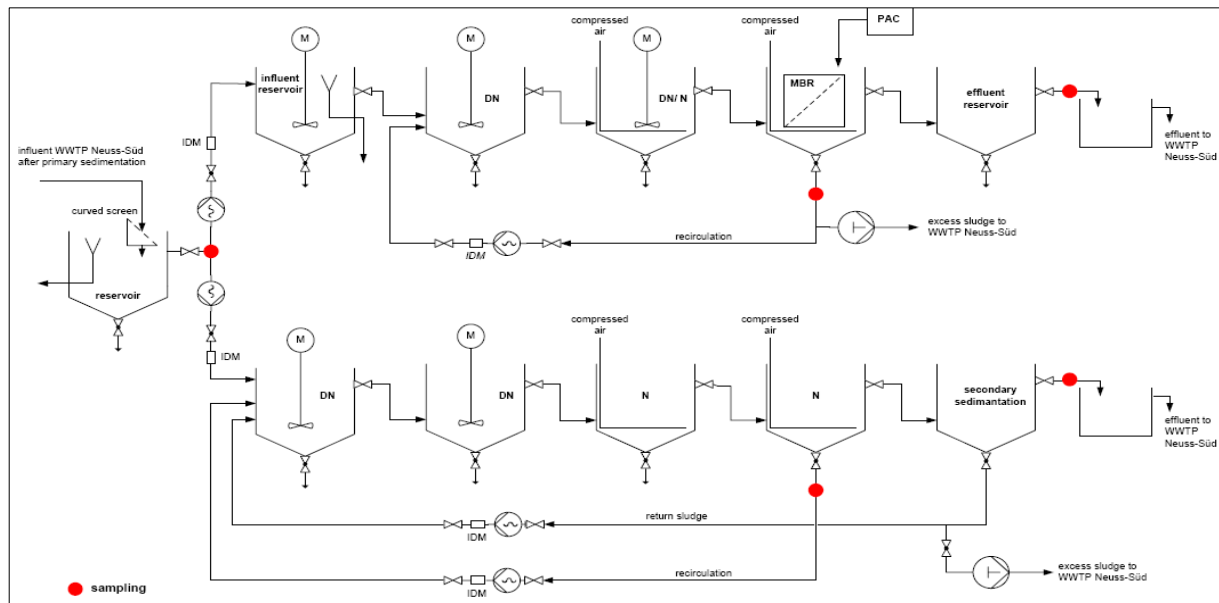


Figure 4.1 PAC addition in biology, pilot scale set-up (Ante et al. 2009)

Figure 4.2 represents the pilot-scale experimental set-up implemented by DPU to assess **PAC addition** to effluent followed by sand filtration. Results from the experiments (removal rates) are here used as functional inventory data input. This technology is also described in NEPTUNE “2nd Periodic Activity Report” (Ante et al. 2009) and further as a full scale plant in Metzger (2008).

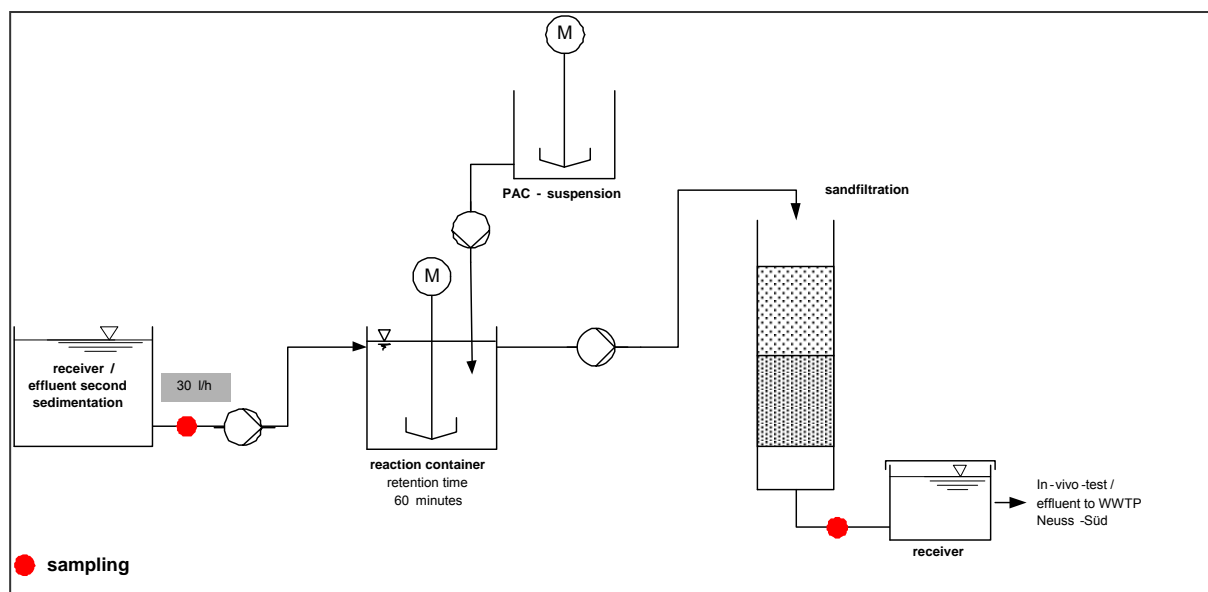


Figure 4.2 PAC addition to effluent, pilot scale set-up (Ante et al. 2009)

Figure 4.3 below shows the full-scale WWTP investigated by Hunziker and Eawag to obtain both physical (infrastructure) and functional (removal rates) inventory data on **ozonation**, **sand filtration** or a combination of both. This technology is described in details in the Eawag report by BAFU/Eawag/AWEL/BMG/Hunziker (2009) and in Hollender et al. (2009)

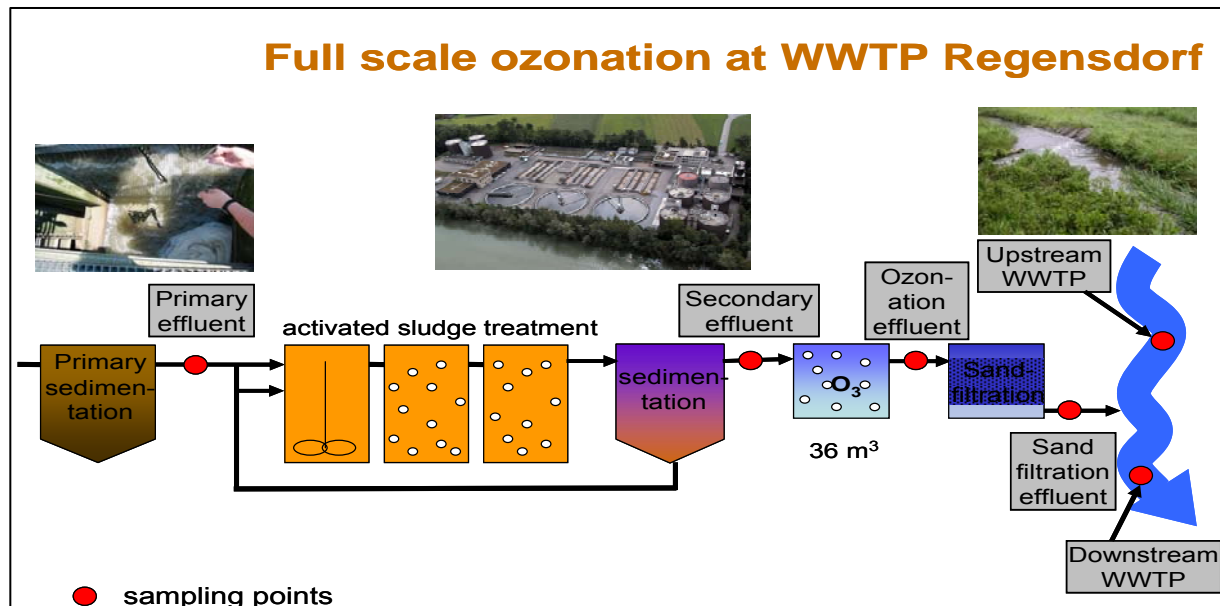


Figure 4.3 Ozonation and sandfiltration, full scale set-up (Hollender 2009)

4.2 CLUSTER 2 technologies; in-line treatments – nutrient removal

Figure 4.4 shows a **conventional WWTP** including primary mechanical separation, biological (an)aerobic (de)nitrification with an activated sludge loop, and chemical phosphorus removal. The resulting sludge is digested in an anaerobic digester with biogas production utilized a gas incinerator unit generating energy at which point the system boundary is closed. This leaves the digested sludge as untreated in the modelling. The technology is described in Doka (2007b) as a capacity-based average of Swiss WWTPs and is only used here as a reference within the cluster. Sludge disposal as off-site municipal waste incineration (Doka 2007c) is included in a separate model.

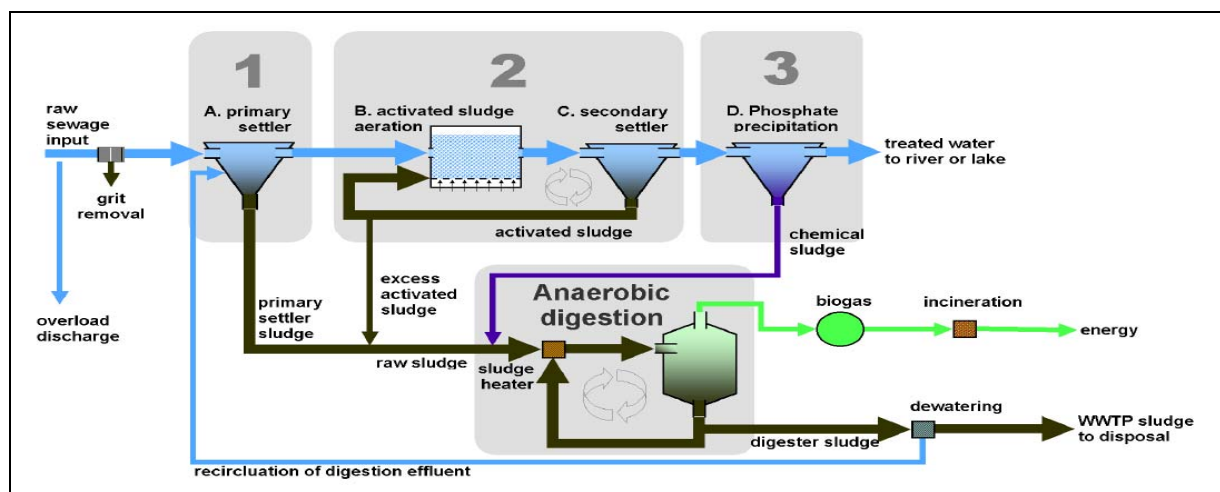


Figure 4.4 Conventional WWTP (Doka 2007b)

Figure 4.5 shows the full-scale set-up of a WWTP with **anammox** reactor (denoted in grey) on the supernatant feedback loop. The system is investigated by EAWAG to yield data on

anammox treatment of supernatant and is described in details in Siegrist et al. (2008). Anammox aim at increased biogas production and reduced aeration energy for N-removal as compared to conventional WWT

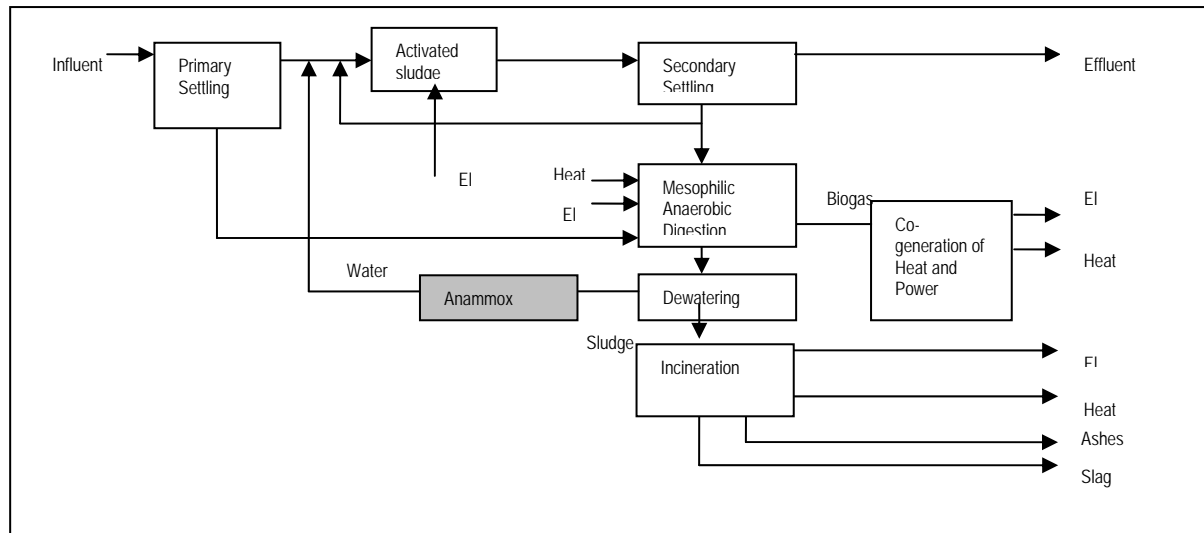


Figure 4.5 Anammox, full scale set-up

Instrumentation, Control and Automation (ICA) is aiming at upgrading conventional WWTPs (Figure 4.4) towards enhanced nutrient removal by testing existing and new control strategies. Within NEPTUNE ICA is performed by ModelEAU, Aquafin and AWMC and is described in NEPTUNE “2nd Periodic Activity Report”.

4.3 CLUSTER 3 technologies; sludge inertization–short description

The **on-site incineration** process is the reference scenario among the inertization sludge treatment alternatives. The digested sludge is mechanically dewatered (from 4%DM → ~30%DM) before incineration. Data from the process were collected Eawag, Hunziker and IRSA. Sludge incineration is shown schematically in Figure 4.6 and waste incineration is described in details in Doka (2007c).

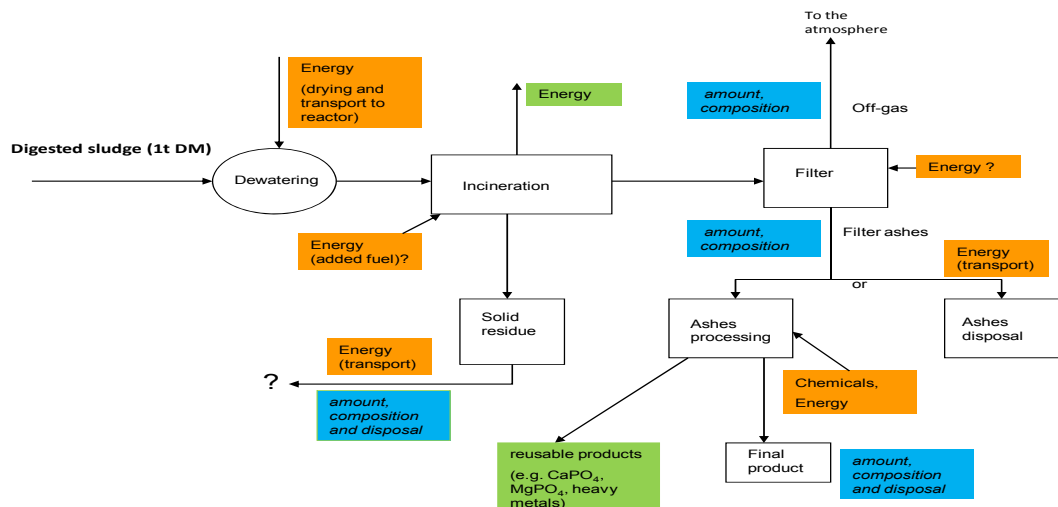


Figure 4.6 Sludge incineration

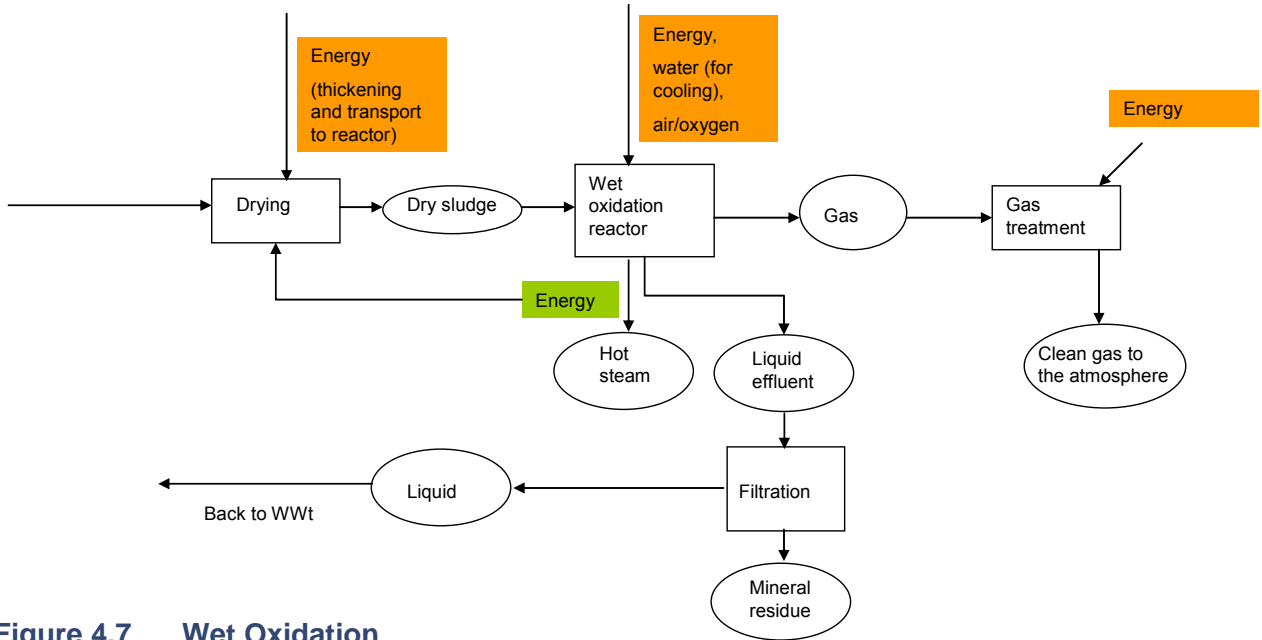


Figure 4.7 Wet Oxidation

Wet oxidation, WO (also called wet air oxidation or critical oxidation) is an oxidative process where the digested sludge is partly oxidized in the reactor, and the effluent streams are gas emissions, liquid effluent and mineral residue. The process is shown schematically in Figure 4.7. In this case the digested sludge only needs to be slightly dewatered (from 4%DM → 10%DM). Data from the process were collected by Eawag and described in details in Deliverable 1.3.

Gasification - Middle Temperature Pyrolysis, Balingen Plant

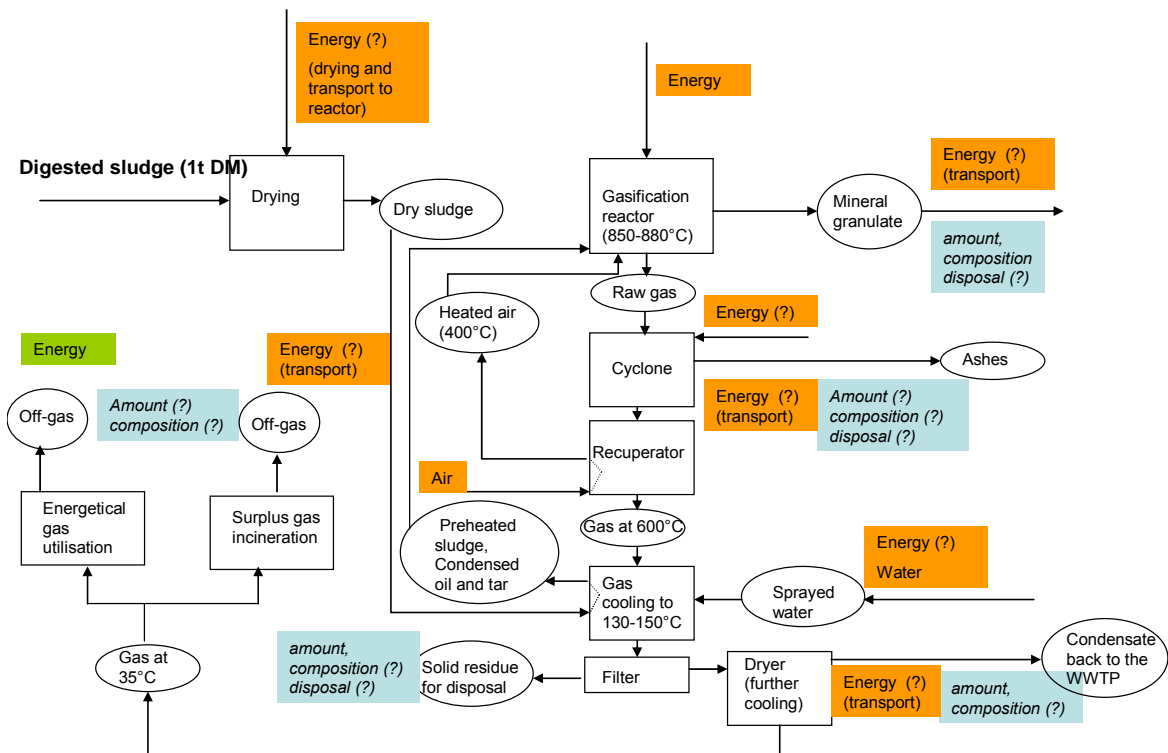


Figure 4.8 Gasification

The **gasification** process (also called middle temperature pyrolysis, MTP) transforms the sludge into gaseous compounds and creates solid and liquid residues. The digested sludge is mechanically dewatered the same way as for incineration (from 4%DM → ~30%DM) but then further dried until 70-85% DM. Gasification is shown in Figure 4.8 and data from the process were collected by Eawag and described in details in Deliverable 1.3.

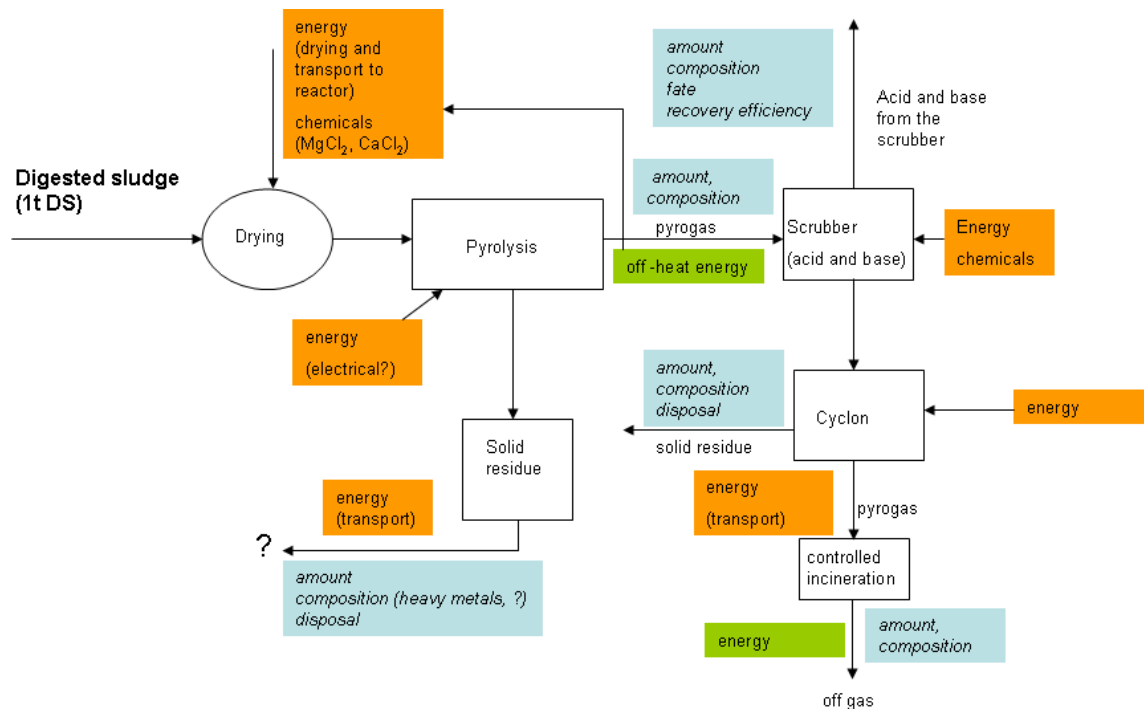


Figure 4.9 High Temperature Pyrolysis (HTP)

High temperature pyrolysis (pyrolysis, HTP) transforms the sludge into syngas and solid residues. As for gasification the digested sludge is dried (from 30%DM → 70-85% DM) after dewatering. HTP is shown in Figure 4.9 and the process was investigated by Eawag and described in details in Deliverable 1.3 and Deliverable 2.2.

4.4 CLUSTER 4 technologies; sludge triage – including disintegration

The reference scenario in this cluster is treating primary and secondary sludge mixed by anaerobic digestion (biogas production) followed by dewatering and incineration. This reference system was investigated by IRSA and shown in Figure 4.10.

In **sludge triage** primary and secondary sludge is treated separately as shown in Figure 4.11. Sludge triage, together with the reference (Figure 4.10), is described in details in Deliverable 1.3. The sludge triage system was investigated by IRSA and secondary sludge is apparently richer in nutrients than primary sludge and may therefore be more suitable as fertilizer in agriculture. The system includes treatment of primary sludge by incineration, and the treatment of secondary sludge by one of the following alternatives (all secondary sludge finally disposed as fertilizer on agricultural land):

- short aerobic thermophilic treatment with intermittent feed (5 days; 45°C)
- ultrasound disintegration + anaerobic digestion
- thermal disintegration + anaerobic digestion

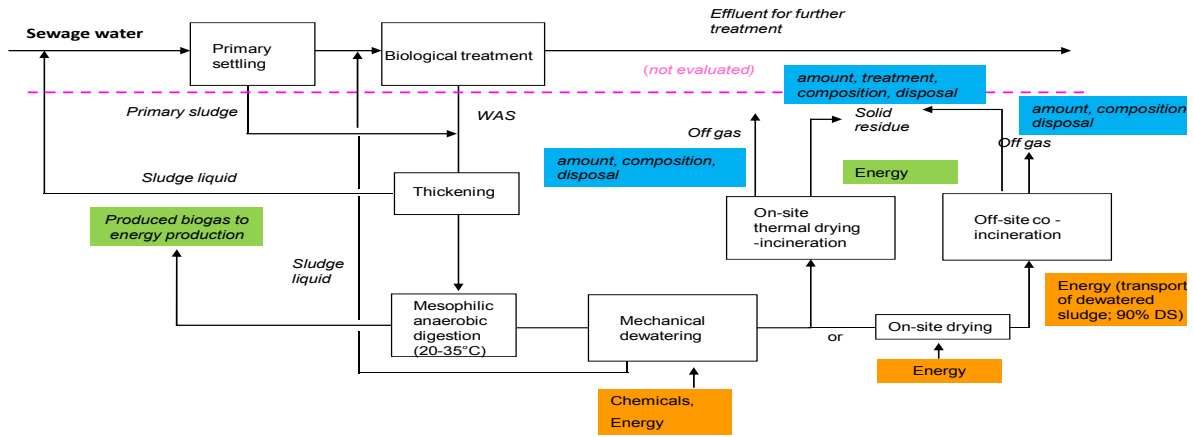


Figure 4.10 Mixed sludge digested followed by incineration (reference)

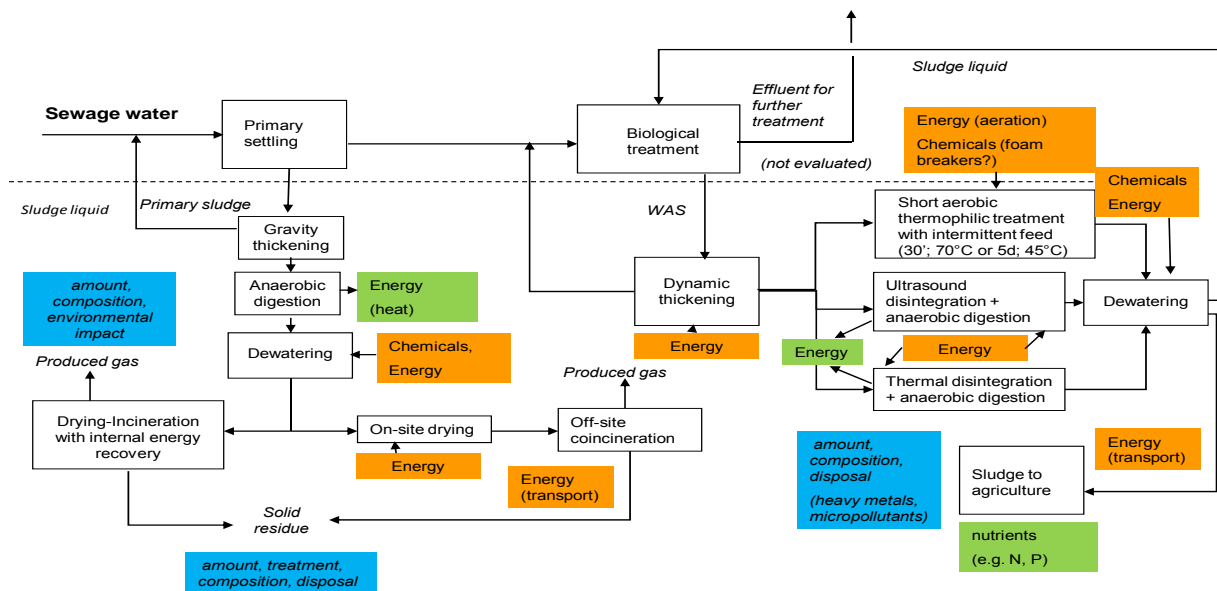


Figure 4.11 Sludge triage: Primary sludge incinerated, secondary sludge pre-treated (aerobic, ultrasound or thermal) followed by digestion (not for aerobic treatment) and finally to agriculture

5 Inventory

Inventories on running (and building) the included technologies (i.e. physical inventory related to foreground processes) are as far as possible based on measurements performed/made available by the different NEPTUNE partners investigating the technologies. These direct data are always preferred (as compared to literature data) but even if the data comes directly from the investigated set-up, there might be some assumptions involved anyway because the experimental set-ups are not always full-scale. In those cases, an approximation of full-scale conditions is assumed.

All background processes in the inventory, like electricity production (i.e. physical inventory related to background processes) are modeled by use of the Ecolnvent 2.0 database (Ecolnvent 2007). Ecolnvent is (among) the most used and best documented life cycle inventory databases worldwide and it contains international industrial life cycle inventory data on energy supply, resource extraction, material supply, chemicals, metals, agriculture, waste management services, transport services and more (Ecolnvent 2007). The database is integrated into the widely used LCA modelling software GaBi, GaBi 4 (PE 2008), and this combined system is the LCA modelling tool used in NEPTUNE.

The functional inventory, e.g. micropollutant removal rates, is as far as possible based on measurements from the relevant NEPTUNE partner.

For each included process, divided into clusters, the physical and functional inventory is described below

5.1 Physical inventory: Cluster 1

The physical inventory is, as mentioned above, ideally based on input data from NEPTUNE partners and background data from Ecolnvent. It is presented in this section as it is used to model the different scenarios in cluster 1 according to best practice – that is, taking marginal electricity into account. The use of alternative electricity scenarios is discussed in Appendix 15.

5.1.1 Ozonation

The physical inventory for ozonation is based on a data sheet from NEPTUNE partner Hunziker (Moser 2008, see Appendix 1) including the type, quantity and life span of materials used for all major infrastructure items as divided in the following categories: buildings and constructions, pipes and valves and electromechanical equipment. The data sheet also contains figures for process energy and ancillary oxygen consumption. These inventory data is based on direct data from a full-scale plant and it is therefore expected to be representative of the investigated technology. Life span of 30 years for buildings and constructions, 20 years for pipes and valves, and 15 years for electromechanical equipment is assumed here (lower end of range in Appendix 1) in order to be in accordance with the 30 years life span of a WWTP used by Doka (2007b). These life spans are used for all relevant processes throughout this study.

The data is used for modeling the physical inventory part of ozonation by use of GaBi with Ecolnvent integrated. The model is divided into “Infrastructure”, “Ancillary” and “Energy”, which are all related to the physical inventory, and shown in the example in Figure 5.1. The box named “Ozonation” in Figure 5.1 is related to the functional inventory and dealt with in Section 5.2. This way of organizing the foreground processes (i.e. according to the type of physical inventory data) is used throughout cluster 1 and partly in the other clusters, and may be summarized as follows:

- Infrastructure

- Buildings and constructions
- Pipes and valves
- Electromechanical equipment
- Ancillary
 - E.g. oxygen
- Energy

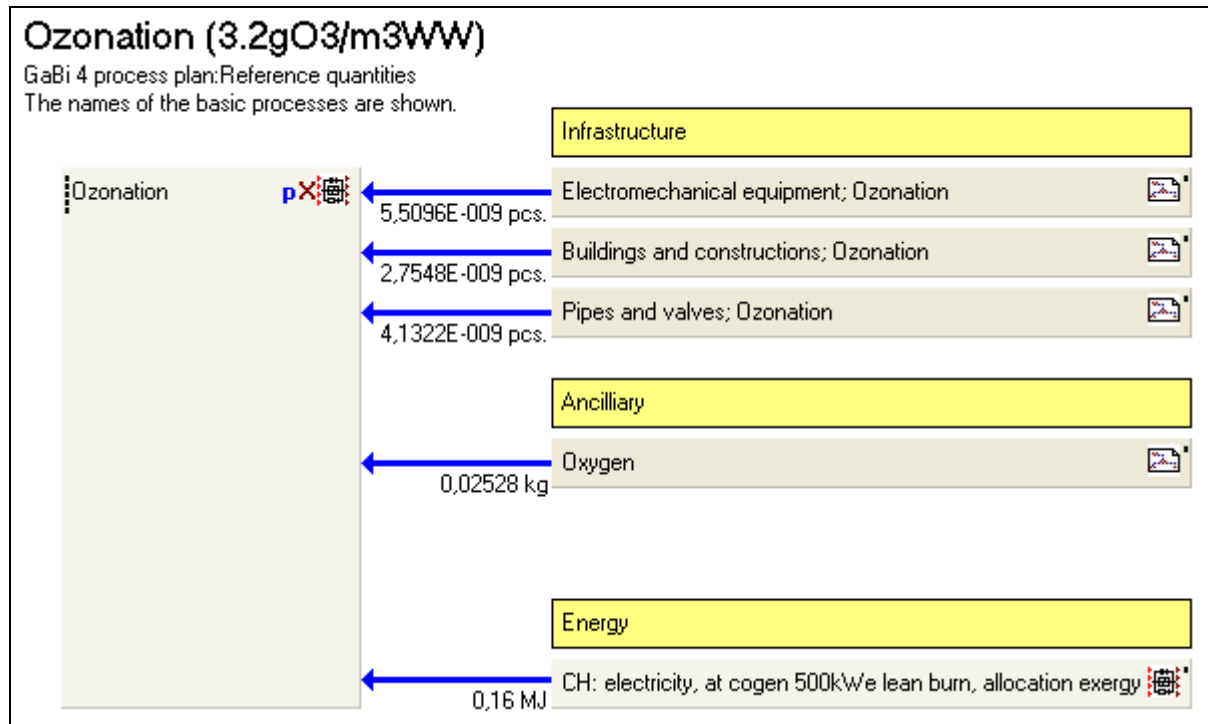


Figure 5.1 GaBi plan showing overall physical inventory modelling for ozonation

Infrastructure: Based on the data input (see Appendix 1), the physical inventory in GaBi includes a list of infrastructure materials as follows. The bullets in this section show processes from GaBi (with Ecolnvent 2.0 integrated) used to model material inputs in this case (background modeling).

- RER: aluminum, production mix, cast alloy, at plant
- RER: copper, at regional storage
- RER: reinforcing steel, at plant
- RER: steel, electric, chromium steel 18/8, at plant
- CH: concrete, normal, at plant
- RER: manganese, at regional storage
- CH: expanded perlite, at plant
- RER: polyethylene, HDPE, granulate, at plant
- RER: polyvinylchloride, at regional storage

Note that stainless steel is modeled as chromium 18/8 steel while galvanized steel and plain steel are modeled as reinforcement steel. All material processes are selected as occurring in Switzerland with materials delivered at plant or regional storage unless unavailable in Ecolnvent, in which case the closest (in terms of realism) country, region or specific process is selected.

In addition, since the materials are delivered at plant or regional storage, the following transport process is used for all, assuming a distance of 100 km within Switzerland and 300 km within Europe:

- RER: transport, lorry >16t, fleet average
- CH: transport, lorry >28t, fleet average

Finally, all materials are modeled as disposed according to the most realistic scenario including recycling or reuse of relevant materials:

- CH: disposal, building, reinforced concrete, to sorting plant
- CH: disposal, building, bulk iron (excl. reinforcement), to sorting plant
- CH: disposal, aluminum, 0% water, to sanitary landfill
- CH: disposal, copper, 0% water, to municipal incineration
- CH: disposal, inert waste, 5% water, to inert material landfill
- CH: disposal, polyethylene, 0.4% water, to municipal incineration
- CH: disposal, building, reinforcement steel, to sorting plant
- CH: disposal, polyvinylchloride, 0.2% water, to sanitary landfill
- Reused materials:
 - RER: iron scrap, at plant (inverted)
 - CH: gravel, unspecified, at mine (inverted)

As an example the modeled GaBi plan for “Buildings and constructions” is shown in Figure 5.2 with all the background processes included.

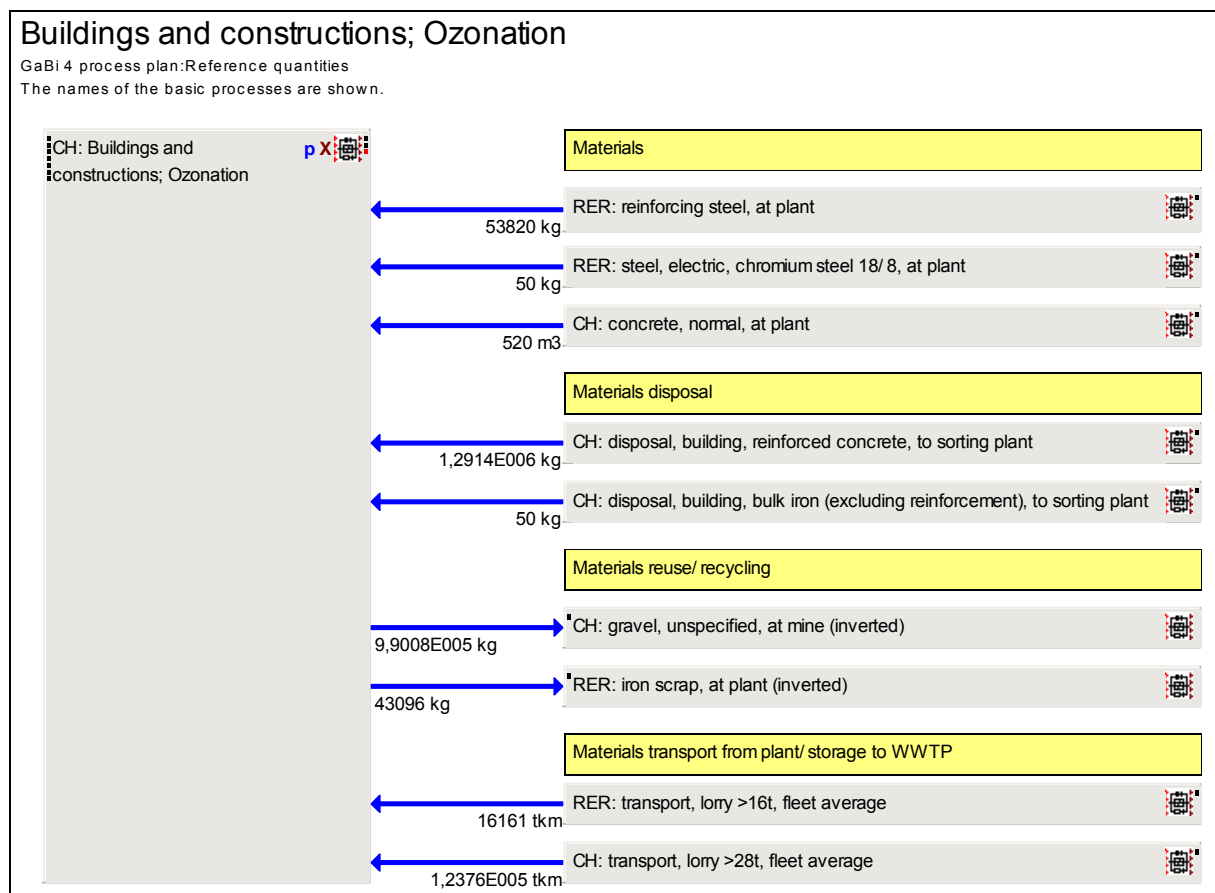


Figure 5.2 GaBi plan showing physical inventory model for “Buildings and constructions” regarding ozonation

Ancillary: From the data sheet (see Appendix 1), the oxygen requirement for ozone production has been calculated to be 7.9 gO₂/gO₃. Based on data in the Swiss report on the pilot ozonation investigations at the waste water treatment plant Regensdorf (BAFU/Eawag/AWEL/BMG/Hunziker 2009) a nominal consumption of 8.3 gO₂/gO₃ may be calculated. In the Clauson-Kaas (2006) study an oxygen consumption of 12.5gO₂/gO₃ is stated for ozonation in general, and based on product datasheets from the ozonator manufacturer Lenntech (<http://www.lenntech.com/>) a consumption of 10 gO₂/gO₃ may be calculated which is identical to the consumption stated by DPU in Ante et al. (2009). The oxygen production and its transport to the ozonation plant are modeled by modifying the Ecolnvent process “RER: oxygen, liquid, at plant [b]”, including the following processes on transport and marginal electricity production (based on natural gas, see Section 3.3) already mentioned.

- RER: transport, lorry >16t, fleet average
- CH: electricity, at cogen 500kWe lean burn, allocation exergy

Energy: The energy (electricity) requirement for ozone production is taken to be the overall energy consumption for ozonation. This is assumed to be a good approximation because it represents the most energy intensive aspect of it. The energy is required in the form of electricity and is also modeled with the Ecolnvent process based on natural gas:

- CH: electricity, at cogen 500kWe lean burn, allocation exergy

Based on the data in Appendix 1 the electricity required may be calculated to be 0.014 kWh/gO₃ (0,05 MJ/ gO₃) which is at the same level as the consumption reported from the investigation at Regensdorf (BAFU/Eawag/AWEL/BMG/Hunziker 2009). Based on product datasheets from the ozonator manufacturer Lenntech (<http://www.lenntech.com/>) a consumption of 0.01 kWh/gO₃ may be calculated. The figure used by Clauson-Kaas (2006) is 0.016 kWh/gO₃.

5.1.2 Sand filtration

The physical inventory for sand filtration is also derived from a data set from Hunziker (Moser 2008), see Appendix 2. It therefore contains the same type of information, as for ozonation, and it is handled in the same way, except that sand filtration does not have any ancillary products. The physical inventory is also in this case based on direct data from a full-scale plant and modeled completely with existing Ecolnvent data. It is therefore expected to be representative of the investigated technology. The overall modeling is shown in Figure 5.3.

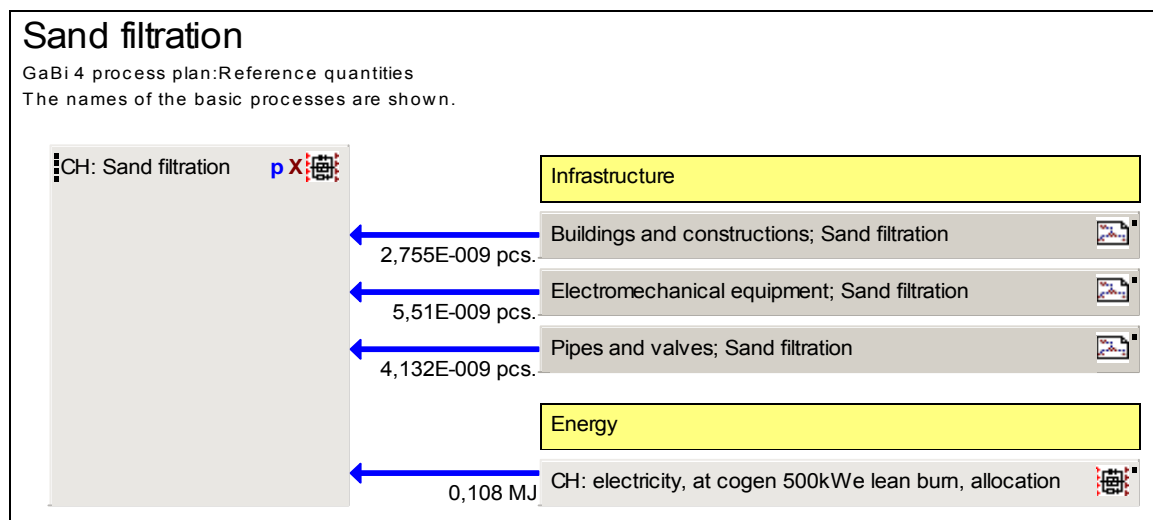


Figure 5.3 GaBi plan showing overall physical inventory modelling for sand filtration

Infrastructure: The materials used in this case are the following:

- CH: concrete, normal, at plant
- RER: polyethylene, HDPE, granulate, at plant
- RER: reinforcing steel, at plant
- RER: steel, electric, chromium steel 18/8, at plant
- CH: sand, at mine
- WEU: hard coal, at regional storage

They are transported using the same assumption as for ozonation. Their disposal is modeled in the following ways:

- CH: disposal, building, reinforced concrete, to sorting plant
- CH: disposal, polyethylene, 0.4% water, to municipal incineration
- CH: disposal, building, reinforcement steel, to sorting plant
- Reused materials:
 - CH: gravel, unspecified, at mine (inverted)
 - RER: iron scrap, at plant (inverted)
 - CH: sand, at mine (inverted)

As an example the modeled GaBi plan for “Pipes and valves” is shown in Figure 5.4 with all the background processes included.

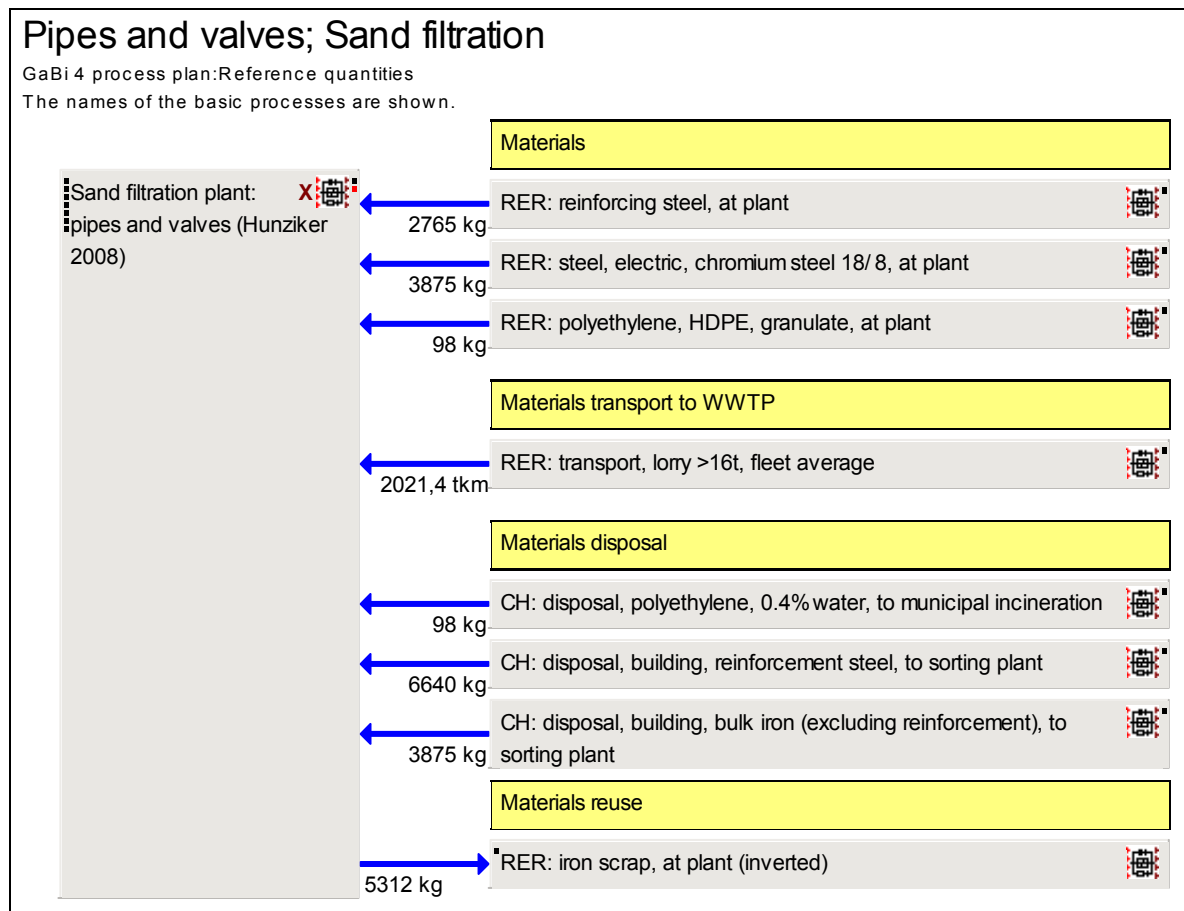


Figure 5.4 GaBi plan showing physical inventory model for “Pipes and valves” regarding sand filtration

Energy: The energy to sand filtrate 1m3 WW is taken to be 0.03 kWh from Hunziker (Appendix 2). It is modeled with the same Ecolnvent process as in ozonation. This figure is at the same level as the one (0.02 kWh/m3WW) stated by Clauson-Kaas (2006).

Overall, it may be said that the physical inventories for both ozonation and sand filtration have an adequate correlation (same order of magnitude) to the models used by Clauson-Kaas (2006).

5.1.3 PAC addition

The physical inventory for PAC addition from Hunziker in Appendix 3 (Moser 2009a) is used to model both PAC additions in biology (Figure 4.1) and PAC addition to the effluent (Figure 4.2). However, the flocculent consumption stated in Appendix 3 is not included as it has not been part of the experiments performed by NEPTUNE partner DPU on which the functional inventory is based. The PAC doses used (20 g/m3, 40g/m3 and 80 g/m3) are based on data inputs from DPU, see NEPTUNE “2nd Periodic Activity Report” (Ante et al. 2009). The production of PAC (actually granulated activated carbon, GAC) is modeled by use of data from Muñoz (2006). All other background processes are based on Ecolnvent data.

As a first iteration, the dosed PAC was assumed to be recuperated 100% and regenerated with a loss of 90% as done in Muñoz (2006). However, it has been clarified during the Varna NEPTUNE meeting (October 2008) that the PAC is not to be recuperated at all. This implies that that production of virgin PAC instead of regenerated PAC is allocated to the functional unit.

In this case, the physical inventory data is based on direct data from a full-scale plant (Appendix 3) and it is therefore expected to be representative of the investigated technology. The PAC dosing quantities are measured (by DPU), but the modeling of the ancillary product PAC is based on literature data for GAC and therefore expected to be the most uncertain part of the PAC addition modeling. This issue is dealt with in Appendix 15.

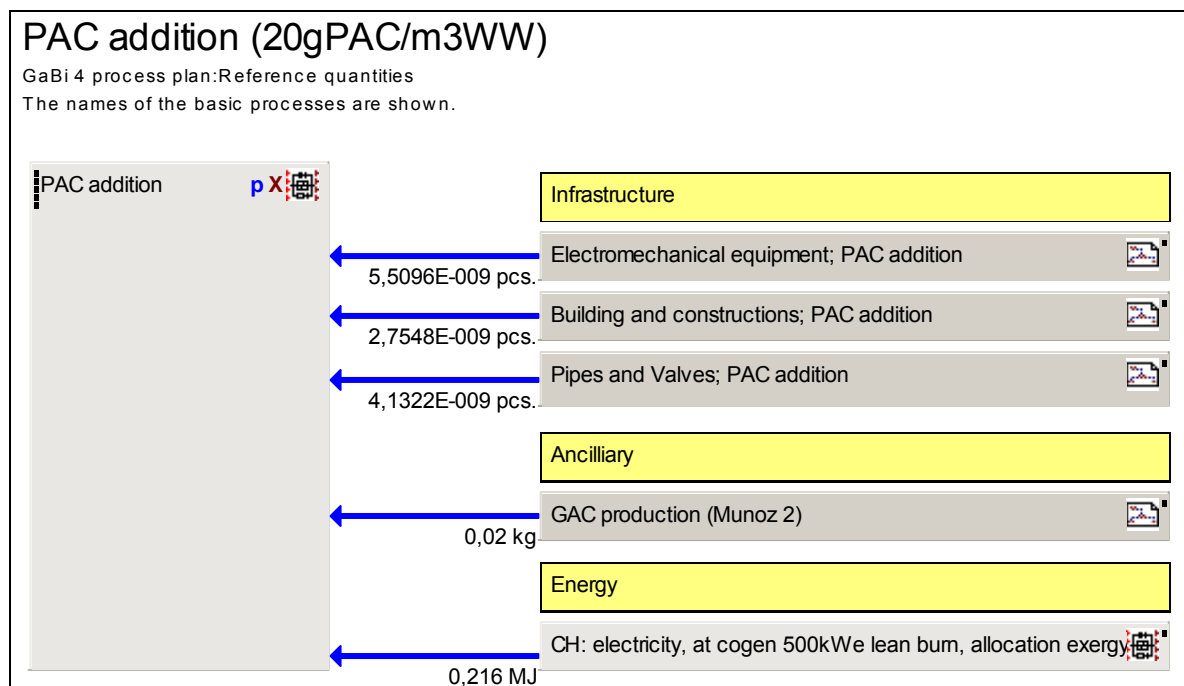


Figure 5.5 GaBi plan showing overall physical inventory modelling for PAC addition

The overall modeling for PAC addition is shown in Figure 5.5.

Infrastructure: The materials used in this case are the following:

- CH: concrete, normal, at plant
- RER: polyethylene, HDPE, granulate, at plant
- RER: reinforcing steel, at plant
- RER: steel, electric, chromium steel 18/8, at plant
- RER: copper, at regional storage
- RER: polyvinylchloride, at regional storage

They are transported using the same assumption as for ozonation. Their disposal is modeled in the following ways:

- CH: disposal, building, reinforced concrete, to sorting plant
- CH: disposal, building, concrete, not reinforced, to sorting plant
- CH: disposal, building, bulk iron (excluding reinforcement), to sorting plant
- CH: disposal, polyethylene, 0.4% water, to municipal incineration
- CH: disposal, building, reinforcement steel, to sorting plant
- CH: disposal, copper, 0% water, to municipal incineration
- CH: disposal, polyvinylchloride, 0.2% water, to sanitary landfill
- Reused materials:
 - CH: gravel, unspecified, at mine (inverted)
 - RER: iron scrap, at plant (inverted)

As an example the modeled GaBi plan for “Electromechanical equipment” is shown in Figure 5.6 with all the background processes included.

Energy: The energy consumption (dosing PAC, stirring and more) for a full scale plant is 0.06 kWh/m³ waste water according to NEPTUNE partner Hunziker (Appendix 3). It is here assumed that the energy consumption is constant, i.e. independent of the dosed amount within the range investigated here (20 – 80 g PAC/m³).

Ancillary: The ancillary product in this case is PAC. It is modeled according to data from Muñoz (2006), model Muñoz 2, for the production of mineral (coke-based) granulated activated carbon (GAC). This was done due to a dearth of specific and reliable data on PAC production and in this respect does not exactly represent the actual situation where Norit SAE Super PAC is used. Actually, Norit was contacted but refused to provide any valuable information about their product. Therefore, the production of 1 kg PAC is modeled with the following inputs:

- 12 kg - CH: water, deionised, at plant
- 5,76 MJ - UCTE: electricity, natural gas, at power plant
- 13,2 MJ - RER: natural gas, burned in industrial furnace >100kW
- 60,8 MJ - RER: hard coal, burned in industrial furnace 1-10MW
- 1 kg - UCTE: hard coal mix, at regional storage

As mentioned above it was the assumption in the first case that used PAC could be regenerated and therefore the production of 1 kg of regenerated PAC, taking into account a 10% loss of material during regeneration has also been modeled. This model is however only included in the sensitivity analysis in Appendix 15.

Along with the data presented above, other sources have been available to model the life cycle inventory (LCI) of activated carbon production (and regeneration) during the NEPTUNE project period and the elaboration of this report. These included another foreground model Muñoz 1 with lower hard coal consumption (also included in Muñoz 2006) as well as one from CGTF (Siegrist 2008a) and ProBas (2008). In addition, also fully aggregated models from ProBas (2008) and CGTF (Siegrist 2008a) modeled in Simapro are available.

The foreground ProBas (2008) model, along with all aggregated ones, is considered unreliable or not suited for use in this case. The Probas model originates from a German database with a different modeling methodology, than Ecolnvent, and transparency is not at all optimal. In a similar way, the aggregated models originated from either that database or other unknown databases with different modeling methodologies and potentially fewer flows than Ecolnvent.

The Muñoz 2 model is selected as being the most representative and reliable one. All models are further dealt with in Appendix 15.

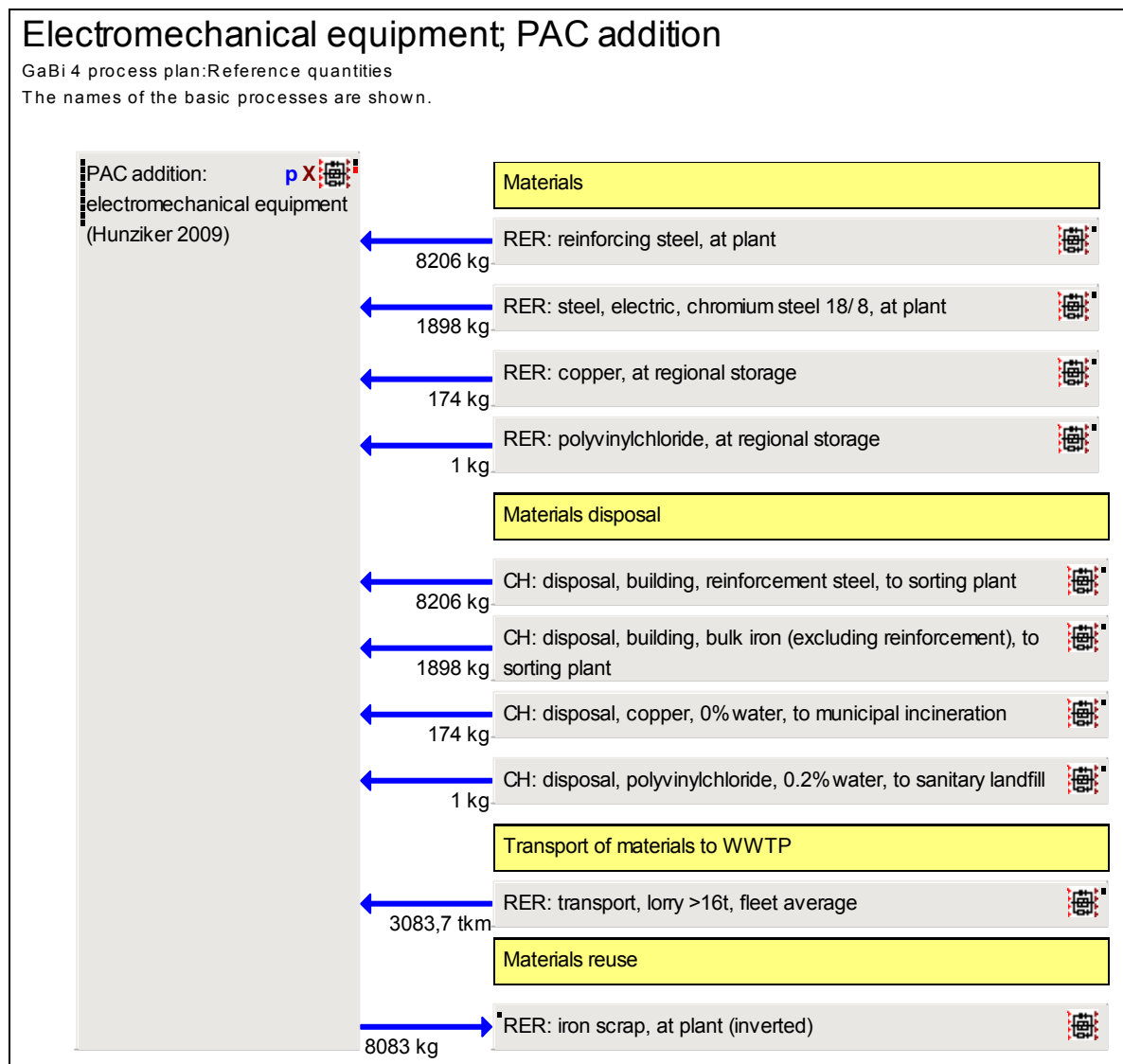


Figure 5.6 GaBi plan showing physical inventory model for “Electromechanical equipment” regarding PAC addition

5.2 Physical inventory: Cluster 2

The physical inventory included here only includes conventional waste water treatment and anammox. Regarding ICA results are reported in Deliverable 1.2.

5.2.1 Conventional waste water treatment

The inventory for the conventional WWT technology is based on the Ecolnvent report by Doka (2007b) and simplified versions of it developed by Muñoz et al. (2007) and Hansen (2008). For details on this inventory, please refer to the report by Muñoz et al. (2007). The

differences applied to this inventory in the present NEPTUNE version are described in this section.

Infrastructure: The infrastructure is taken as the EcoInvent inventory for Swiss wastewater treatment plants (capacity class 3; 25,000 PCE/year, see Doka 2007b) and, as opposed to the model in Muñoz et al. (2007), does not include the sewer system. Furthermore, the EcoInvent WWTP inventory includes the same type of materials as included in the ozonation, PAC and sand filtration inventories and required for the construction of the plant in the Swiss context. However, the transport requirements in particular are different because different assumptions have been used (see Doka 2007b). In addition, a few additional materials as well as some production/construction processes are included.

As a result of these changes to the inventory and the fact that overhead requirements are allocated to the plant (only direct requirements were considered in cluster 1), the infrastructure inventory is in this case more complete and therefore larger relative to those of ozonation, PAC and sand filtration.

Energy: The energy requirements in the inventory are calculated according to Muñoz et al. (2007) except that electricity is modeled – as in cluster 1 – with the EcoInvent process for marginal electricity production by natural gas in Switzerland. Note that in this case again, overhead requirements are included, especially for electricity. In addition, natural gas and fuel oil are considered within the energy requirements:

- CH: electricity, at cogen 500kWe lean burn, allocation exergy
- RER: natural gas, burned in boiler modulating >100kW
- CH: light fuel oil, burned in boiler 100kW, non-modulating

However, in case the energy balance is positive (more energy produced than consumed), the GaBi model sets the energy input required at zero instead of allocating the excess energy as an output in order to model it as replacing energy from other sources. The effect on the NEPTUNE model version used here is that there is no demand on natural gas and light fuel oil (energy demand covered by internal production, i.e. cogeneration at digester) and only net electricity demand (external supply) is included.

Ancillary: The ancillary products required to implement this technology are chemicals used to precipitate phosphorus during tertiary treatment. They are calculated as shown in Muñoz et al. (2007) as a function of the amount of phosphorus removed and modeled with the following EcoInvent processes:

- RER: iron sulphate, at plant
- CH: iron (III) chloride, 40% in H₂O, at plant ecoinvent
- RER: aluminum sulphate, powder, at plant ecoinvent

In addition, the transport to the plant is included based once again on the assumptions described in Doka (2007b).

The GaBi plan for the NEPTUNE conventional waste water treatment model is shown in Figure 5.7.

Disposal of sludge: It is here assumed that the remaining sludge amount after digestion and dewatering, i.e. excess sludge after anaerobic digestion with 63% moisture, is incinerated at a municipal waste incineration plant. The modeling is based on Doka (2007c) and the GaBi plan is shown in Figure 5.8.

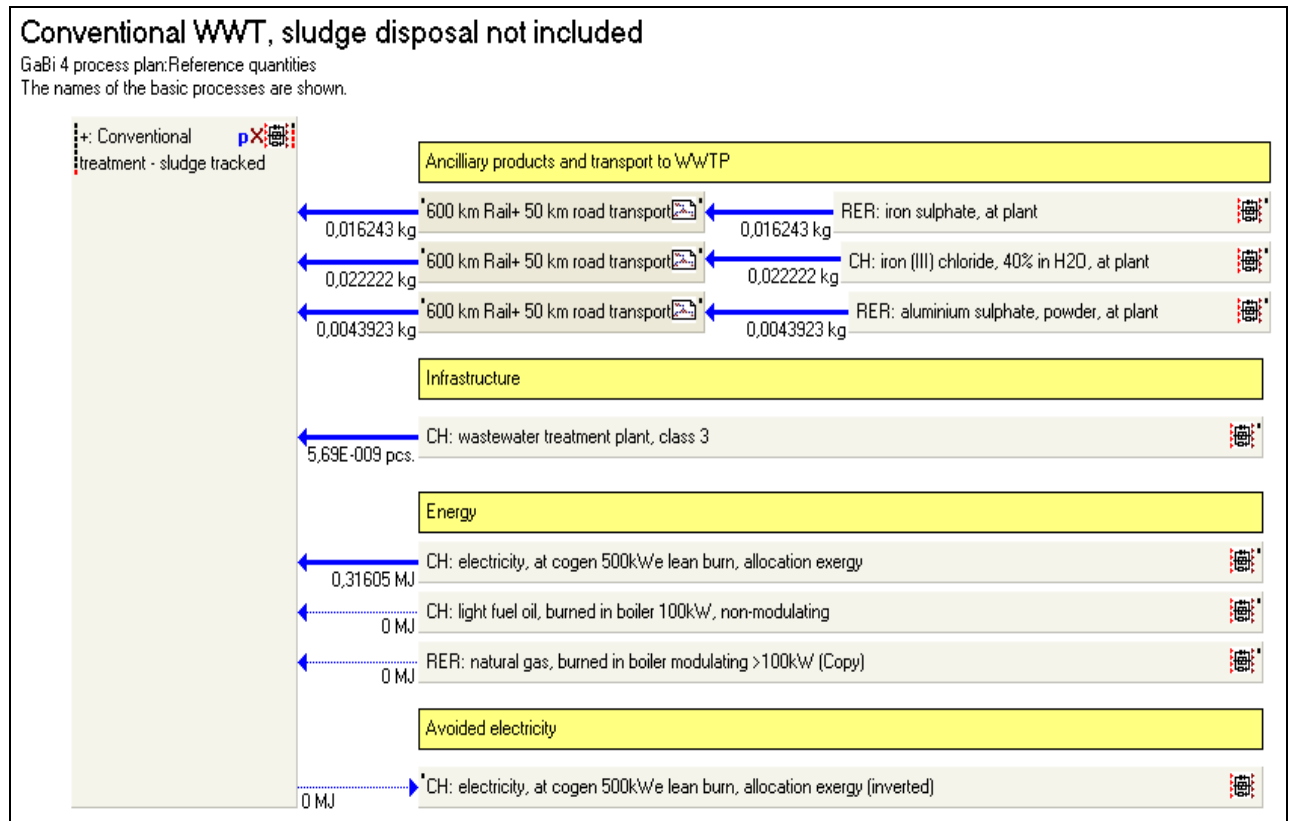


Figure 5.7 GaBi plan showing physical inventory model for conventional waste water treatment – sludge incineration not included

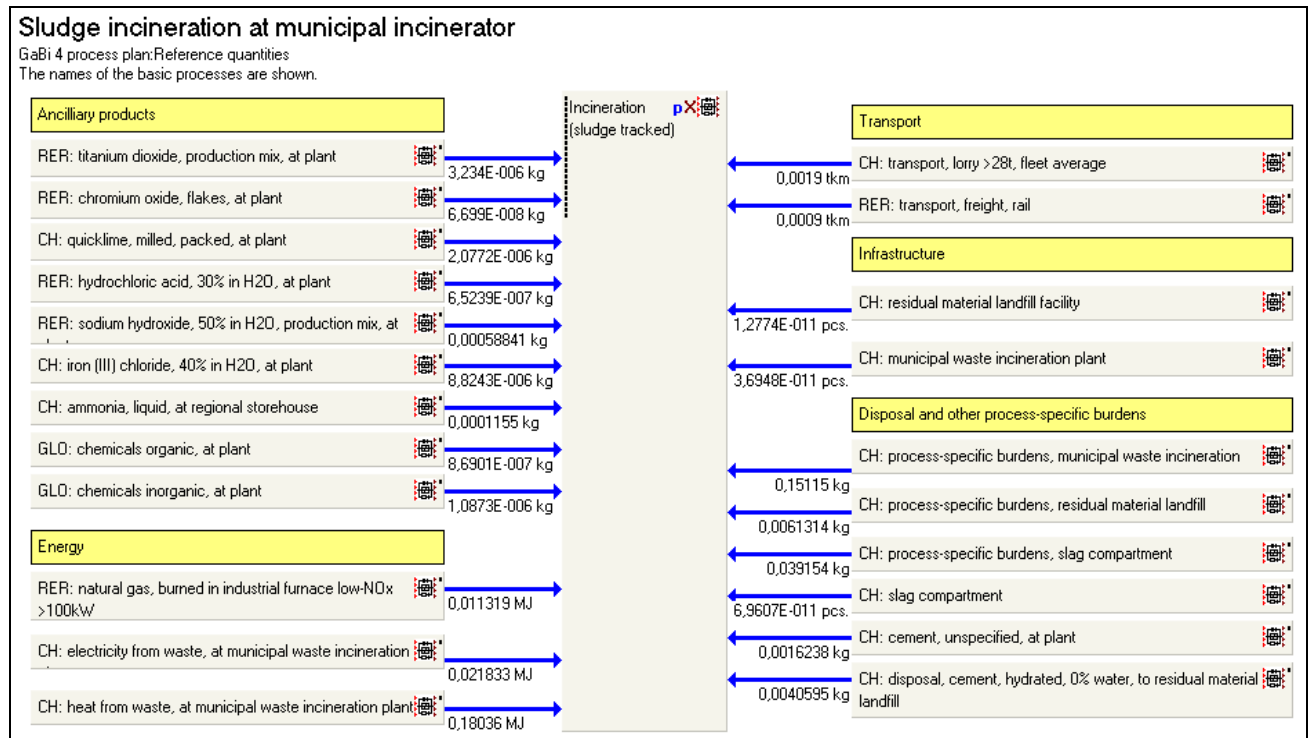


Figure 5.8 GaBi plan showing physical inventory model for sludge incineration at a municipal sludge incineration plant

5.2.2 Autotrophic anaerobic ammonium oxidation (anammox)

The physical inventory for anammox is based on input data from the NEPTUNE partners Hunziker (Moser 2009b).

The overall modeling for anammox is shown in Figure 5.9.

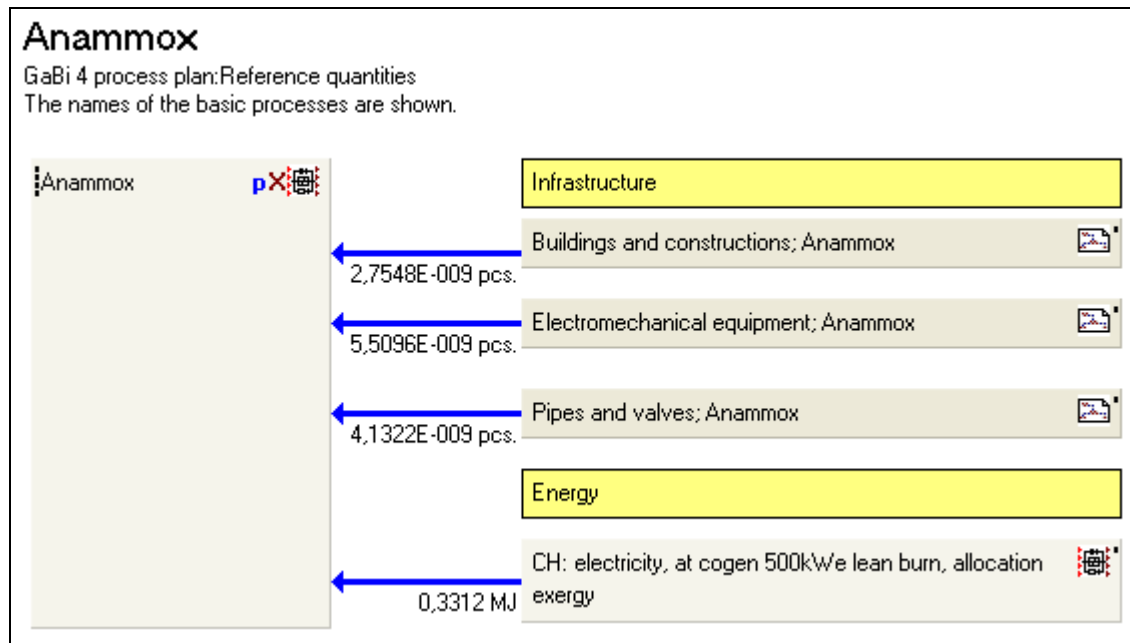


Figure 5.9 GaBi plan showing physical inventory model for anammox

Infrastructure: Based on the data input from Moser (2009b), see Appendix 4, the physical inventory in GaBi includes a list of infrastructure materials. Most of these have all ready been introduced for ozonation and PAC addition infrastructure and the same EcoInvent background processes are used here. However glass is used for building and construction, and rubber and polypropylene are used for electromechanical equipment. These “new” materials are modeled with the following EcoInvent background processes.

- CH: foam glass, at regional storage
- RER: synthetic rubber, at plant
- RER: polypropylene, granulate, at plant

The following “new” disposal processes are therefore also added:

- CH: disposal, building, mineral wool, to sorting plant
- CH: disposal, mineral wool, 0% water, to inert material landfill
- CH: disposal, building, polyethylene/polypropylene products, to final disposal
- CH: disposal, rubber, unspecified, 0% water, to municipal incineration

Energy: The gross energy consumption for running anammox is taken to be 1.1 kWh/m³ from Moser (2009b), see Appendix 4. It is modeled with the same default EcoInvent electricity production process (CH: electricity, at cogen 500kWe lean burn, allocation exergy) as all the other processes. Anyway, as may be observed from Figure 5.9 the value 1.1 kWh/m³ is not used directly but instead the net difference in energy consumption with or without anammox (0.092 kWh/m³ ~ 0,3312 MJ/m³), see further details in Section 5.5.2.

Ancillary: No ancillaries are used

5.2.3 ICA strategies

Reported in Deliverable 1.2.

5.3 Physical inventory: Cluster 3

The physical inventory included here comprises all investigated methods for sludge inertization.

5.3.1 On-site sludge incineration

Incineration is here used as the reference process, as it is the only existing well-developed technology and considered as the “default” sludge disposal method by NEPTUNE, while alternative techniques like pyrolysis and gasification are relatively new, and exist as newly developed full scale plants.

The physical inventory for on-site incineration is described shortly below and in more details in Appendix 9.

Infrastructure: It has not been possible to get a construction inventory list (like the ones for ozonation, PAC addition etc., Appendix 1 – 4) for the Winterthur sludge incineration plant – only functional inventory data as described in Section 5.5.3. Instead, the infrastructure of a Swiss municipal incinerator (Doka 2007c) is used here (CH: municipal waste incineration plant) It has a capacity of 100,000 tons/year and an expected lifetime of 40 years. It includes the production of all the materials in the inventory list and their corresponding disposal.

In order to aim at modeling the infrastructure in a similar way as other processes in NEPTUNE (ozonation etc.) i.e. including also the disposal and recycling of infrastructure material as for ozonation etc., the process “CH: municipal waste incineration plant “ has been adapted to:

“CH: municipal waste incineration plant (NEPTUNE, with recycling)”.

The infrastructure modeling used is shown in Figure 5.10.

Infrastructure (incineration), Winterthur

GaBi 4 process plan: Reference quantities
The names of the basic processes are shown.

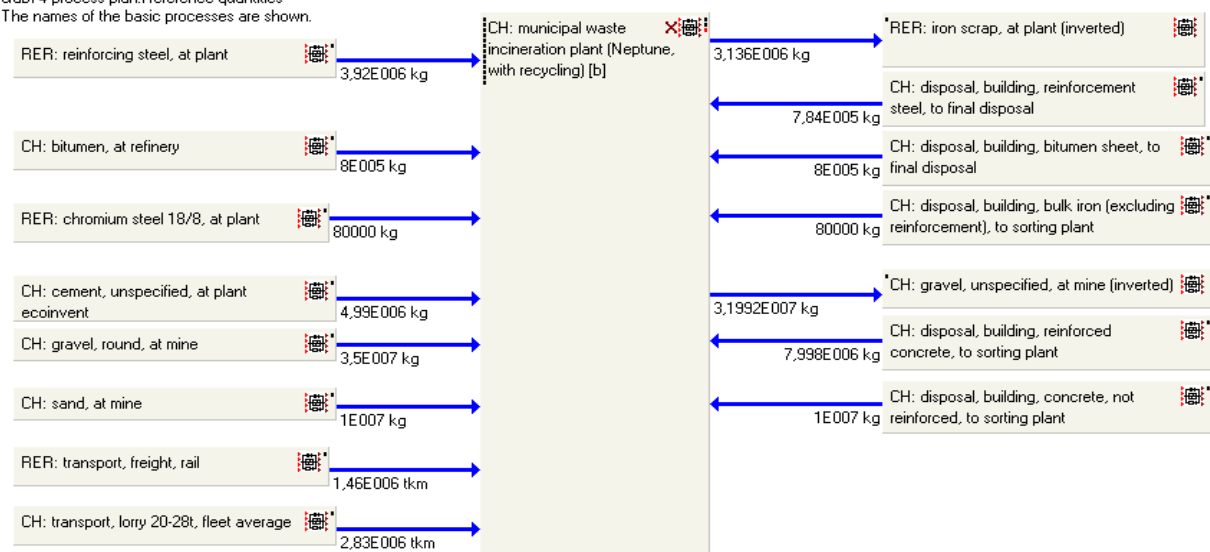


Figure 5.10 GaBi plan showing infrastructure model for on-site incineration of sludge

Energy: The plant needs externally supplied fuel and electricity to run the incineration process and produces heat. Part of the electricity comes from the digesters of the wastewater treatment plant, and another part from the power grid. Fuel oil and biogas (from the wastewater treatment plant) are used as fuels. Besides the NEPTUNE default marginal electricity process (i.e. “CH: electricity, at cogen 500kWe lean burn, allocation exergy”) the following processes are included:

- CH: electricity, at cogen with ignition biogas engine, allocation exergy
- CH: biogas, from sewage sludge, at storage
- CH: light fuel oil, at regional storage
- CH: heat, at cogen 500kWe lean burn, allocation exergy (inverted, neptune)

The electricity consumption is calculated in Appendix 9 and results in the following figures, see Table 5.1.

Table 5.1 Electricity consumption for on-site sludge incineration (kilo Watt hour per ton dry matter, kWh/tDM)

Electricity consumption	Dewatering	20 kWh/tDM	External electricity supply
	Incineration	110 kWh/tDM	Digester gas use
	Incineration	220 kWh/tDM	External electricity supply
Total electricity consumption		350 kWh/tDM	

The total fuel consumption (biogas and fuel oil for heating) is estimated to 820 kWh/tonDM and the heat production 620 kWh/ton DM.

Ancillary: A lot of ancillary products are used and dominated by chemicals used for the treatment of the off-gas. A list is shown in Table 5.2 with the corresponding EcoInvent process used in the GaBi modeling. The ancillaries are described in more details in Appendix 9.

Table 5.2 Ancillaries used for on-site sludge incineration

Ancillary	EcoInvent process used in modelling
Flocculant	GLO: chemicals inorganic, at plant
Quarry sand for filtration	CH: sand, at mine
Calcium chloride (CaCl ₂)	RER: calcium chloride, CaCl ₂ , at plant
TMT15/Na 3T	GLO: chemicals organic, at plant
Sodium persulfate (Na ₂ S ₂ O ₈)	GLO: sodium persulfate, at plant
Sodium hydroxide (NaOH)	RER: sodium hydroxide, 50% in H ₂ O, production mix, at plant
NH ₃	CH: ammonia, liquid, at regional storehouse
Hydrochloric acid (HCl)	RER: hydrochloric acid, 30% in H ₂ O, at plant
Sulphuric acid (H ₂ SO ₄)	RER: sulphuric acid, liquid, at plant
Salt for softening (NaCl)	RER: sodium chloride, powder, at plant

The full GaBi plan for the NEPTUNE onsite incineration plant is shown in Figure 5.11.

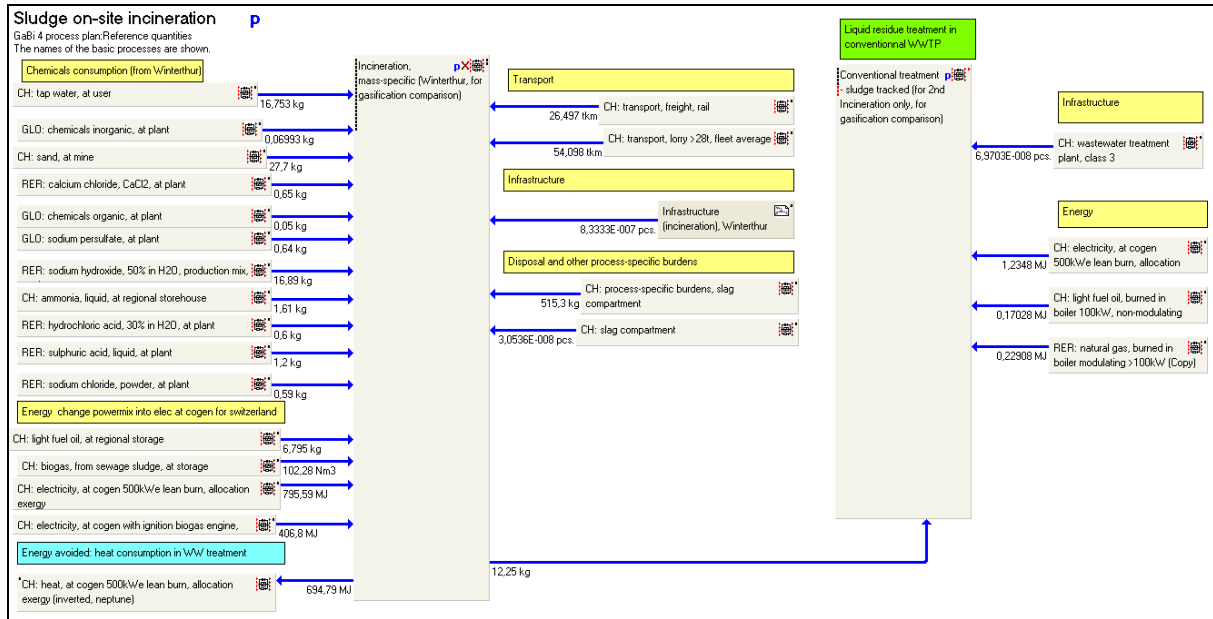


Figure 5.11 GaBi plan showing full physical inventory model for on-site incineration of sludge

5.3.2 Wet oxidation (WO) of sludge

Wet oxidation (also called wet air oxidation or critical oxidation) is an oxidative process where the digested sludge is partly oxidized in the reactor, and the effluent streams are gas emissions, liquid effluent and mineral residue. The physical inventory of WO is described shortly below and in more details in Appendix 10.

Infrastructure: As a proxy for the infrastructure materials and disposals, data from the gasification plant is used (see Appendix 10). The infrastructure is modeled in a similar way as other processes in NEPTUNE and shown in Figure 5.12.

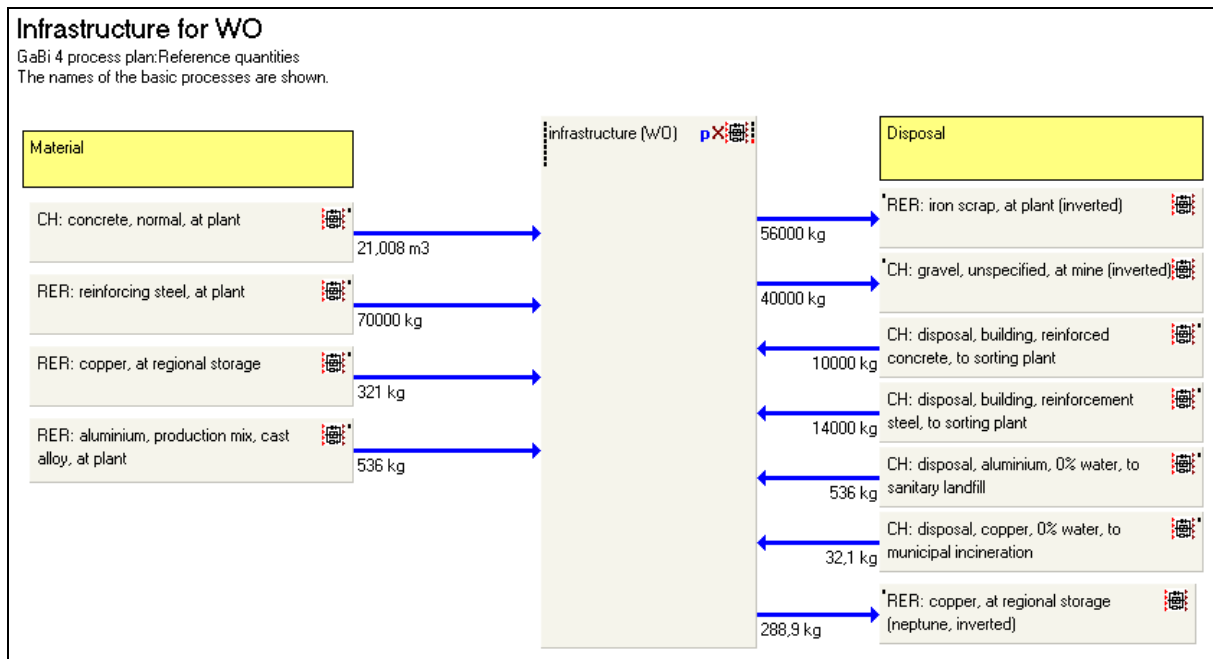


Figure 5.12 GaBi plan showing infrastructure model for WO of sludge

Energy: The plant needs externally supplied electricity for dewatering and running the wet oxidation but also produces electricity and heat via the biogas produced from the liquid returned to the digester (liquid rich in digestible organics). Of the energy gain a split of 1/3 electricity production and 2/3 heat production is assumed. The energy exchanges are modeled in a similar way as described for other NEPTUNE processes and the balance including the electricity consumption is calculated in Appendix 10 and results in the following figures (see Table 5.3)

Table 5.3 Electricity consumption for WO of sludge (kilo Watt hour per ton dry matter, kWh/tDM)

Electricity consumption	190 kWh/tDM	External electricity supply
Electricity production	- 60 kWh/tDM	Based on digester gas
Total electricity consumption	130 kWh/tDM	

No external fuel is used and the heat production amounts to 120 kWh/tonDM.

Ancillary: Nitric acid (1.2 kg/ton DM) and oxygen (870 kg/ton DM) is used. Oxygen is modeled as shown in Figure 5.13.

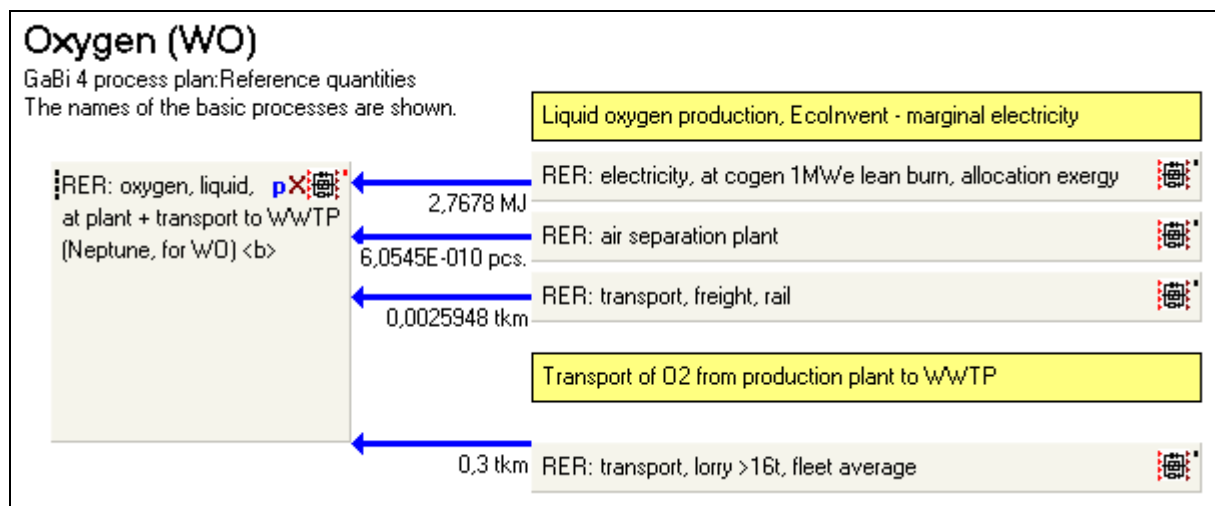


Figure 5.13 GaBi plan showing model for oxygen

The full GaBi plan for the NEPTUNE WO plant is shown in Figure 5.14. The potential impact (i.e. due to metal content) of the rich liquid residue going back to the WWTP is not included directly in the GaBi modeling. It is however commented upon in the results section (Section 8.3.2).

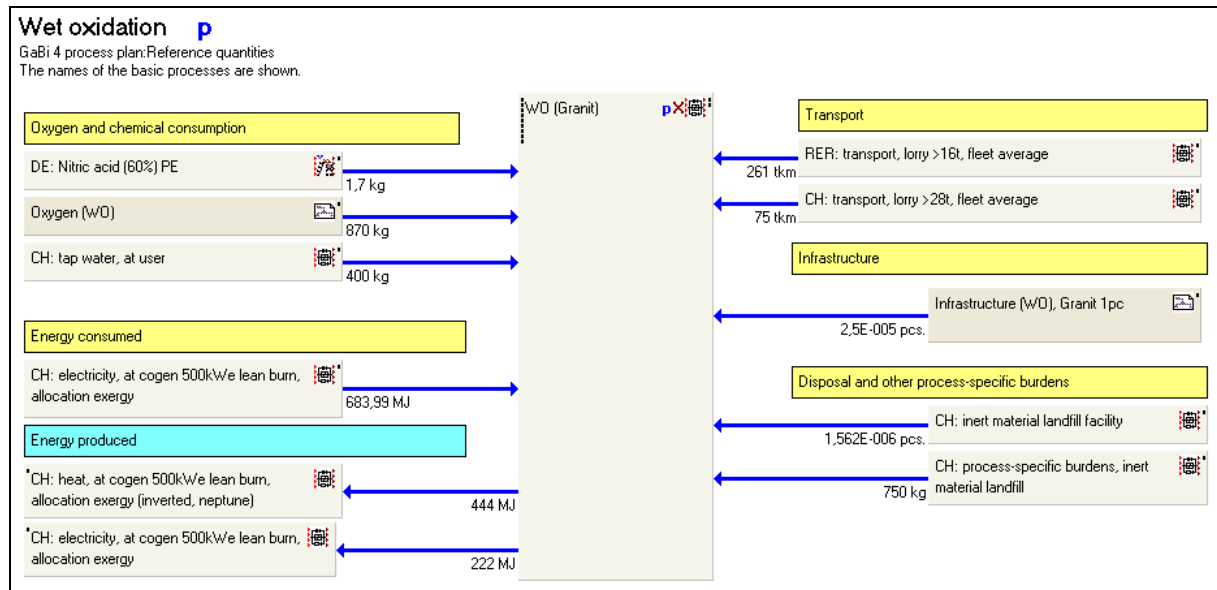


Figure 5.14 GaBi plan showing full physical inventory model for wet oxidation of sludge

5.3.3 High temperature pyrolysis (HTP) of sludge

At very high temperature and no oxygen present the sludge is transformed into syngas and solid residues during high temperature pyrolysis (HTP, pyrolysis). The process is kind of similar to the process gasification (also called middle temperature pyrolysis, described in Section 5.3.4) but operates at temperatures above 1000 °C. The sludge needs drying to a level of 70% - 85% DM before treatment in the HTP process. The physical inventory of HTP is described shortly below and in more details in Appendix 11.

Infrastructure: The infrastructure materials are based on an inventory list from the pyrolysis pilot plant in Switzerland (see Appendix 11). The infrastructure is model in a similar way as other processes in NEPTUNE and shown in Figure 5.15.

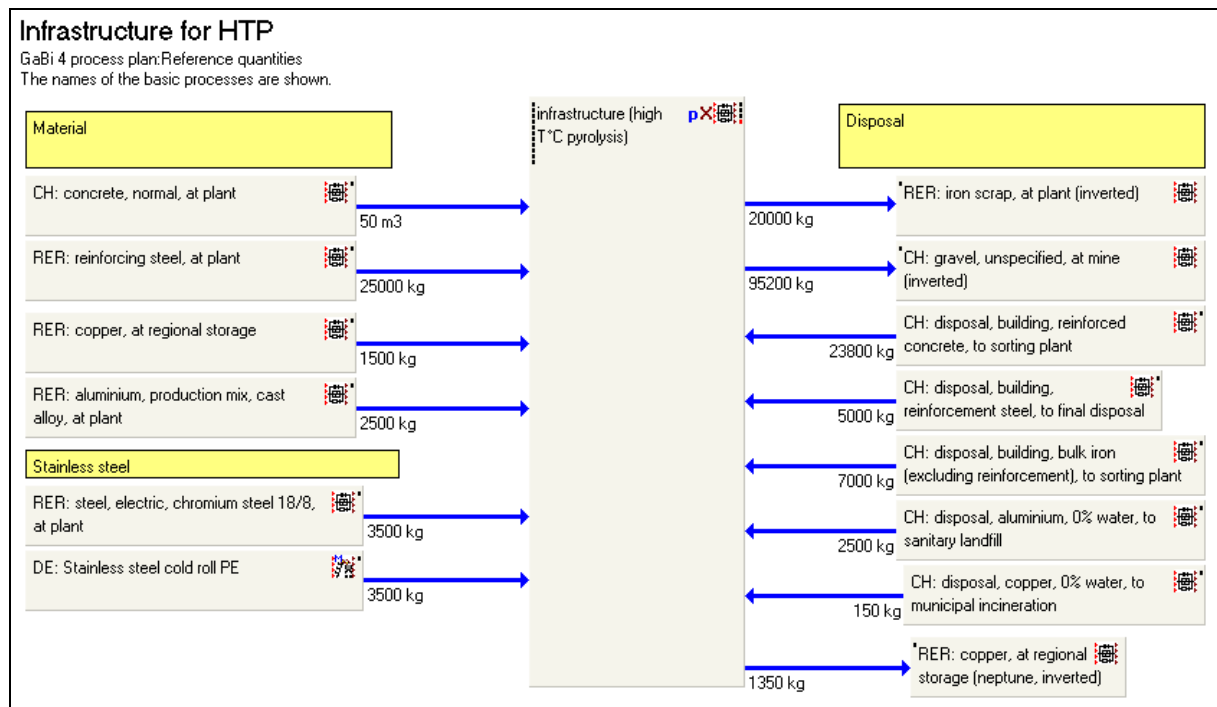


Figure 5.15 GaBi plan showing infrastructure model for HTP of sludge

Energy: The plant needs externally supplied electricity for dewatering and running the pyrolysis but also produces electricity and heat from burning the syngas. The energy exchanges are modeled in a similar way as described for other NEPTUNE processes but theoretical electricity and heat gain is used as no measurements were available. This certainly leads to an advantage in the energy balance for HTP as compared to the other methods in the cluster and should be kept in mind when comparing within the cluster (see Section 8.3.5. Both a balance including heat drying and one including solar drying is calculated in Appendix 11 and results in the following figures, see Table 5.4A.

Table 5.4A Electricity consumption for HTP of sludge (kilo Watt hour per ton dry matter, kWh/tDM)

HTP	Heat drying	Solar drying
Electricity consumption	340 kWh/tDM	420 kWh/tDM
Electricity production	- 960 kWh/tDM	- 960 kWh/tDM
Total electricity consumption	- 620 kWh/tDM	- 540 kWh/tDM

Regarding the heat balance HTP with heat drying has a surplus of 690 kWh/ton DM whereas HTP with solar drying has a surplus of 2200 kWh/ ton DM.

Ancillary: No chemicals are included in the inventory (see Appendix 11) but as gas cleaning (scrubbing) is performed according the description (Figure 4.9) chemicals alike the ones used for gas treatment at incineration may be assumed. The potential impact from chemicals use in gas treatment at incineration is therefore included (not visible in the GaBi plan in Figure 5.16).

The full GaBi model for HTP with heat drying is shown in Figure 5.16 and the slightly different model including solar drying is shown in Appendix 11 (Figure A11.1).

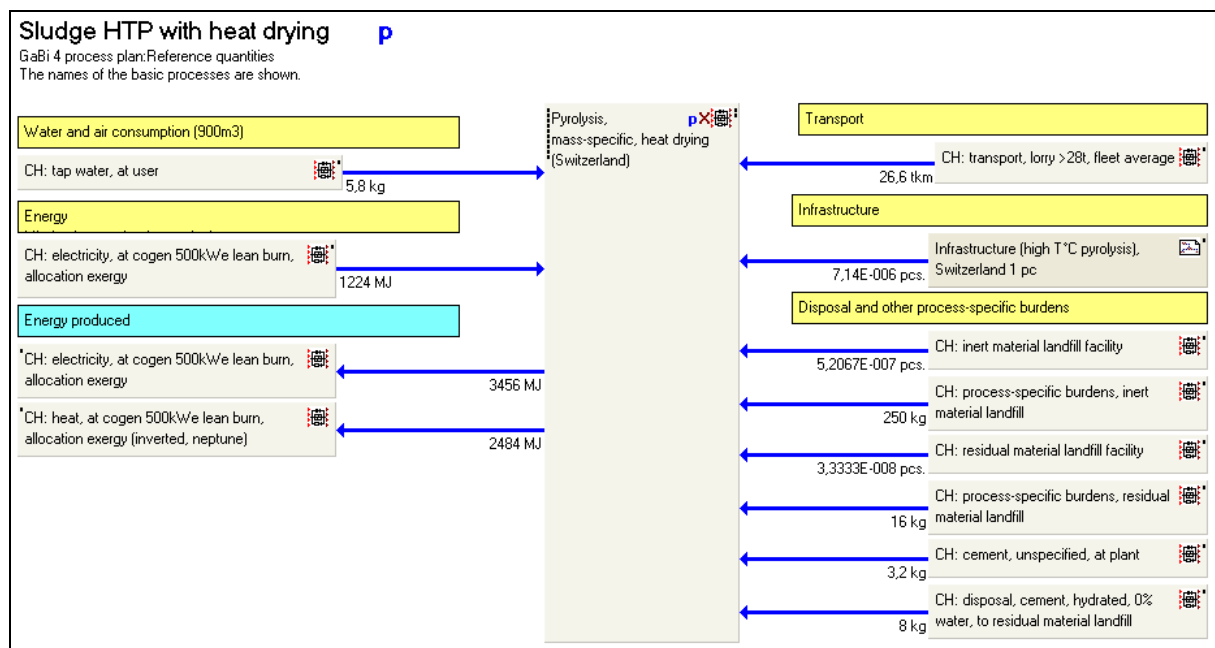


Figure 5.16 GaBi plan showing full physical inventory model for HTP with heat drying

5.3.4 Middle temperature pyrolysis (gasification) of sludge

The gasification process transforms the sludge into gaseous compounds and solid residues. It operates at a temperature of 850 - 880 °C. The sludge needs drying to a level of 70% - 85% DM before treatment by gasification. The drying is here done by either heat drying (partly fuel based) or solar drying (consuming electricity). The physical inventory of gasification is described shortly below and in more details in Appendix 12.

Infrastructure: The infrastructure materials are partly based on an inventory list from the gasification plant in Switzerland, see Appendix 12. The infrastructure is modeled in a similar way as other processes in NEPTUNE and shown in Figure 5.17.

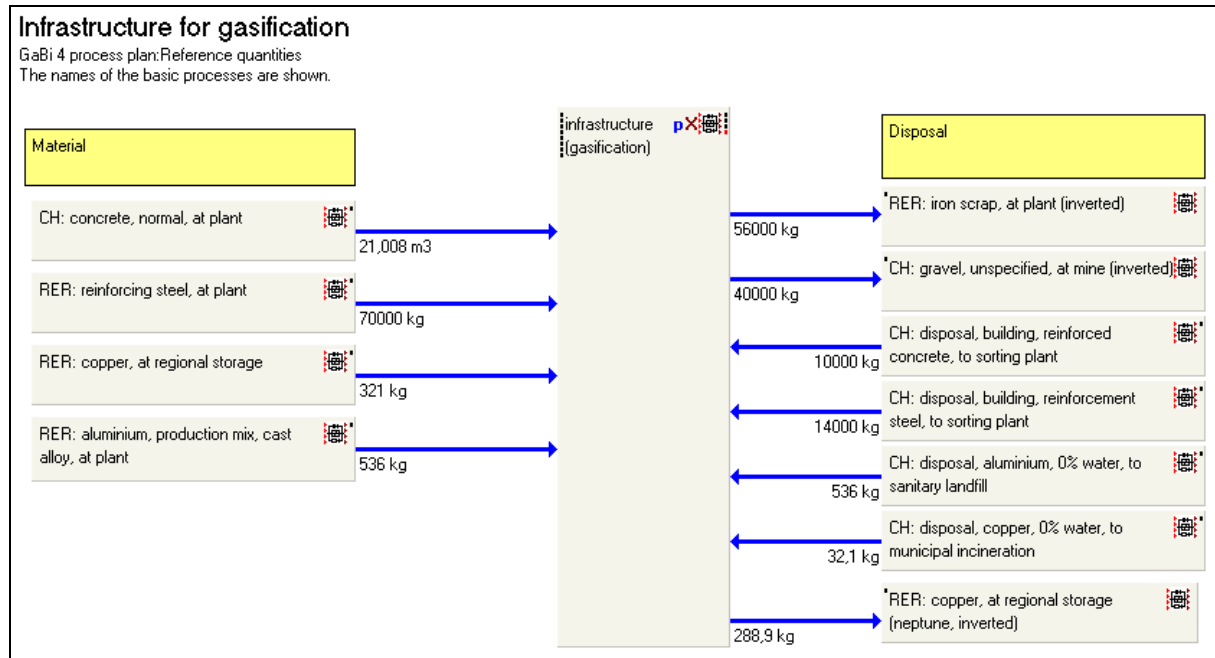


Figure 5.17 GaBi plan showing infrastructure model for gasification of sludge

Energy: The plant needs externally supplied electricity for dewatering and running the pyrolysis but also produces electricity and heat from burning the syngas. The energy exchanges are modeled in a similar way as described for other NEPTUNE processes. Both a balance including heat drying and one including solar drying is calculated in Appendix 12 and results in the following figures (see Table 5.4B)

Table 5.4B Electricity consumption for gasification of sludge (kilo Watt hour per ton dry matter, kWh/tDM)

Gasification	Heat drying	Solar drying
Electricity consumption	120 kWh/tDM	200 kWh/tDM
Electricity production	- 500 kWh/tDM	- 500 kWh/tDM
Total electricity consumption	- 380 kWh/tDM	- 300 kWh/tDM

Regarding the heat balance gasification with heat drying has a deficit of 440 kWh/ton DM whereas gasification with solar drying has a surplus of 1000 kWh/ ton DM.

Ancillary: No chemicals are included in the inventory. Apparently no gas cleaning (scrubbing) is performed (needed), see Appendix 12.

The full GaBi model for gasification with heat drying is shown in Figure 5.18 and the slightly different model including solar drying is shown in Appendix 12 (Figure A12.2).

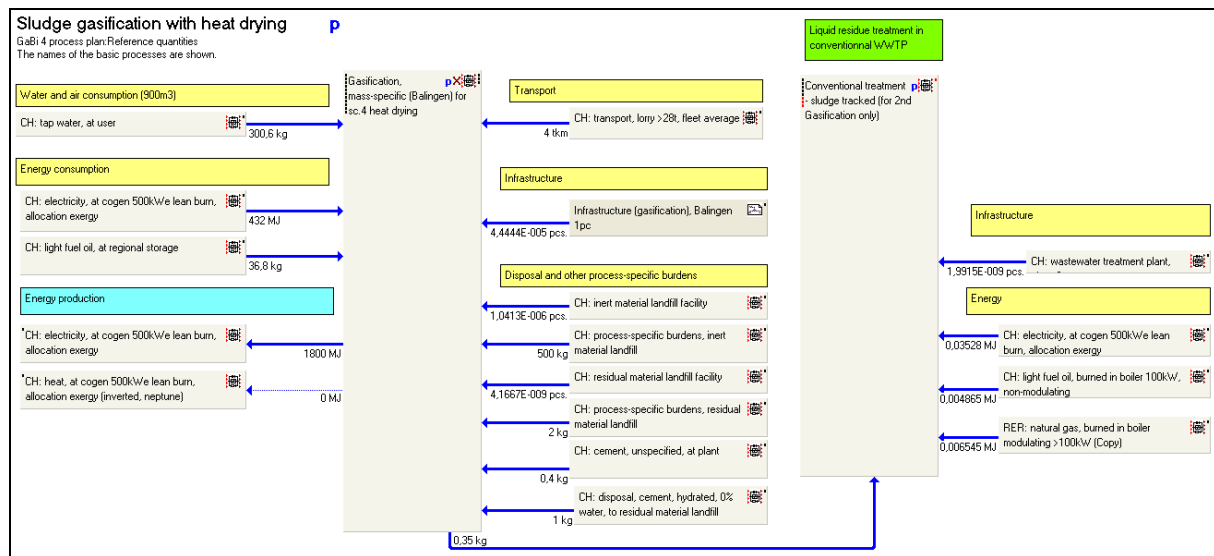


Figure 5.18 GaBi plan showing full physical inventory model for gasification with heat drying

5.4 Physical inventory: Cluster 4 - sludge triage incl. disintegration

Sludge triage consists in treating differently primary and secondary sludge, instead of treating them together as mixed sludge. Secondary sludge is assumed to be richer in nutrients than primary sludge as reported in Deliverable 1.3 (Bagnuolo et al. 2009). Thus, application of only treated secondary sludge on agricultural land may be an environmental advantage.

The reference scenario A1 is the treatment of mixed sludge by incineration (on-site) as shown in Figure 5.19. The alternative processes are shown in Figure 5.20 and comprises the treatment of primary sludge by incineration, and the treatment of secondary sludge by:

- B: short aerobic thermophilic treatment with intermittent feed (5 days; 45°C)
- C: ultrasound disintegration + anaerobic digestion
- D: thermal disintegration + anaerobic digestion

For further details about the processes, please refer to Deliverable 1.3 (Bagnuolo et al. 2009).

The process steps in common when comparing sludge triage with sludge incineration (and the different sludge triage methods against each other) include the following:

- Primary settling
- Biological treatment
- Thickening
- Mechanical dewatering (after anaerobic digestion or aerobic treatment)

These process steps will be excluded from the LCA modelling leaving the following to be taken into account (Table 5.5).

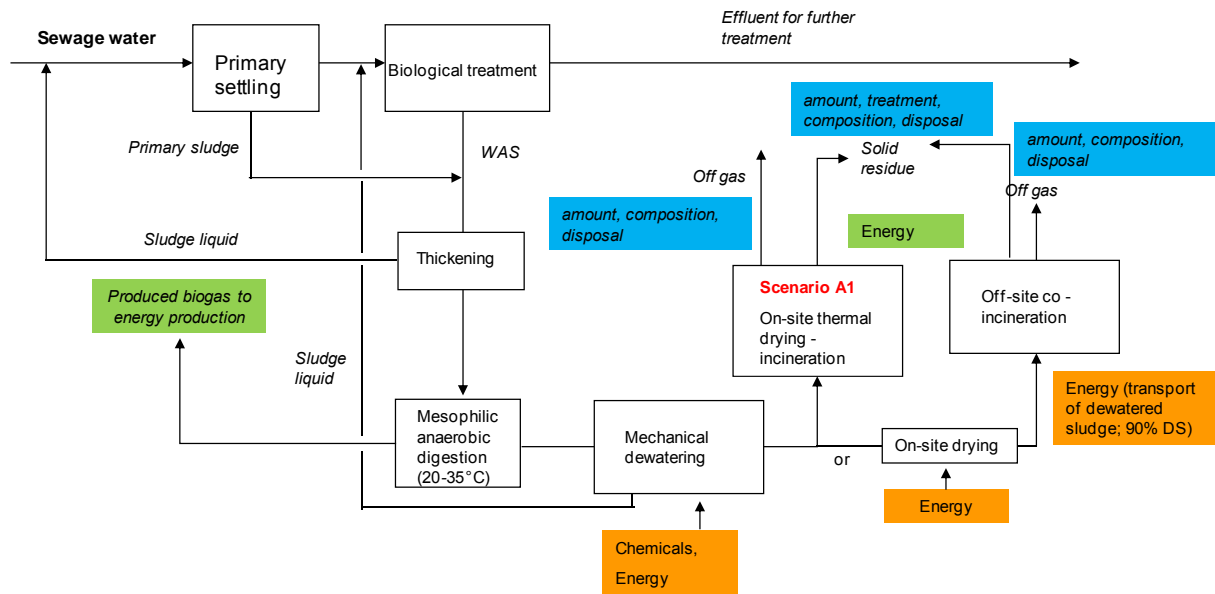


Figure 5.19 Mixed sludge incineration (scenario A1 in bold red)

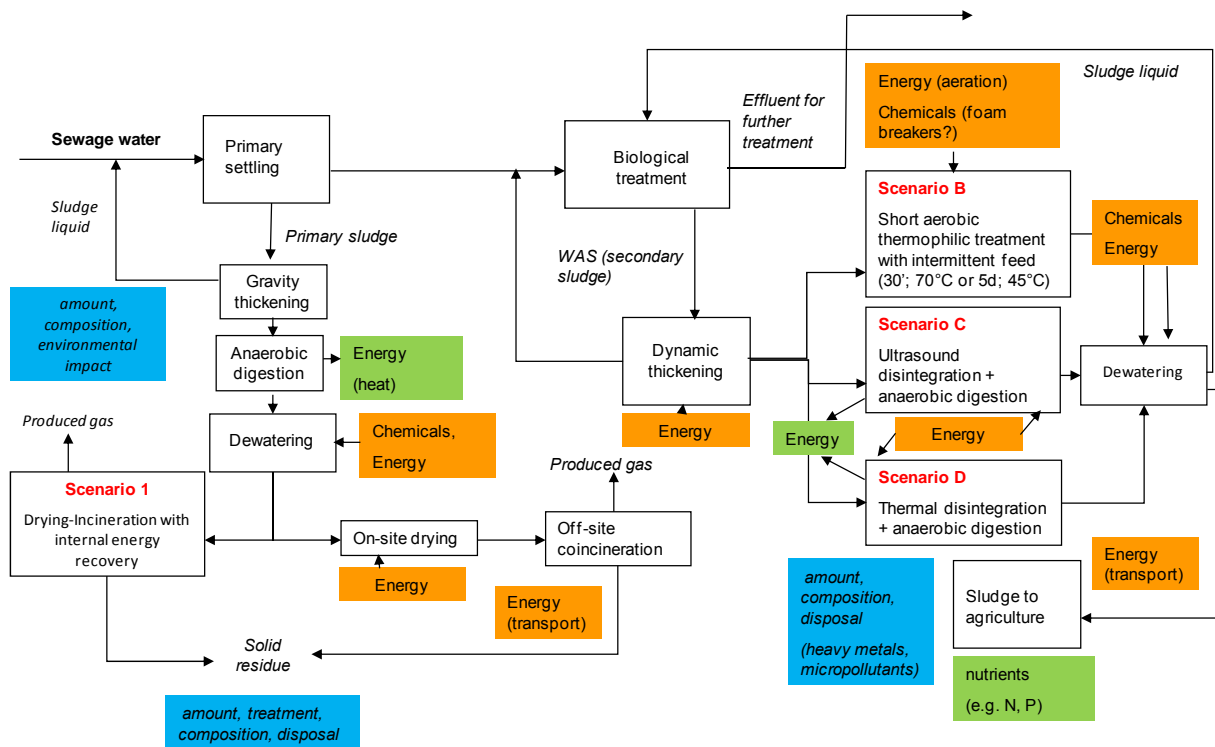


Figure 5.20 Sludge triage (scenario 1, B, C, D in bold red)

Table 5.5 Processes included in the LCA modelling

Type of sludge	Incineration - A 1	Sludge triage - B,C,D 1			
	Mixed	Primary 56%	Secondary 44%	Secondary 44%	Secondary 44%
Process (box)		1	B	C	D
Mesophilic (37°C) anaerobic digestion	X	X		X	X
On-site thermal drying-incineration (1)	X	X			
Short aerobic thermophilic (45°C) treatment (B)			X		
Ultrasound disintegration(C)				X	
Thermal disintegration (D)					X

Infrastructure: It has not been possible to get infrastructure data on the special process steps included here. Anyway, the infrastructure materials used for the different process steps compared, i.e. thermal disintegrator, ultrasound disintegrator and thermophilic treatment equipment are assumed not to be that different as having a significant impact on the results. As regards comparison to on-site incineration the infrastructure of the special processes are most probably negligible. Regarding the anaerobic digester (only excluded in scenario B) the favouring of all other special processes (scenario C and D) should be kept in mind when comparing environmental sustainability among the scenarios. For incineration the infrastructure (and the full model for the incineration part) is based on the model described in Section 5.3.1.

Energy: During anaerobic digestion both electricity (produced from biogas) and heat is produced whereas on-site incineration has a negative energy balance. Ultrasound disintegration and thermophilic disintegration also needs energy for running but increases the digestible part off the sludge and hereby also increases the energy production by anaerobic digestion. Short aerobic thermophilic treatment (scenario B) does not increase the energy production by anaerobic digestion (only included for primary sludge) as its main aim is to remove pathogens. The energy balances for the different scenarios are shown in the tables below and primarily based on data from NEPTUNE partner IRSA (Deliverable 1.3: Bagnuolo et al. 2009, including excel sheet from Giuseppe Mininni) and NEPTUNE partner AWMC (Batstone 2006). The balances are described more detailed in Appendix 14.

Table 5.6 Electricity and heat balance for scenario A1 (mixed sludge digested and on-site incinerated)

Type	Process	kWh/ton DM
Electricity consumption	Anaerobic digestion	70
	On-site incineration	350
Electricity production	Anaerobic digestion	- 500
Total electricity consumption		- 80
Heat consumption	Anaerobic digestion	320
	On-site incineration	200
Heat production	Anaerobic digestion	- 1000
Total heat consumption		- 480

The energy balance for treating one ton dry matter mixed sludge by anaerobic digestion followed by on-site incineration is shown in Table 5.6. More energy is produced than consumed in this scenario. This is also the case when primary sludge is treated the same way as evident from Table 5.7. The figures in Table 5.7 covers only the 56% of the dry matter (i.e. primary sludge) going directly to anaerobic digestion (followed by incineration) while the rest (44% of total dry matter, i.e. secondary sludge) is going to scenario B, C or D.

Table 5.7 Electricity and heat balance for scenario 1 (primary sludge digested and on-site incinerated). Only the scenario 1 share (56%) of the total 1 ton dry matter treated by triage is shown

Type	Process	kWh/ton DM
Electricity consumption	Anaerobic digestion*	40
	On-site incineration*	200
Electricity production	Anaerobic digestion	- 420
Total electricity consumption based on 0.56 ton DM primary		- 180
Heat consumption	Anaerobic digestion*	180
	On-site incineration*	110
Heat production	Anaerobic digestion	- 830
Total heat consumption based on 0.56 ton DM primary		- 540

* Similar to consumption for mixed sludge but related to 0.56 tonDM instead of 1 tonDM

The energy balance for combining anaerobic digestion of primary sludge followed by incineration (scenario 1) with aerobic thermophilic treatment (scenario B) is shown in Table 5.8. Also in this case both more electricity and heat is produced than consumed.

Table 5.8 Electricity and heat balance for scenario 1 (primary sludge digested and on-site incinerated) combined with scenario B (aerobic thermophilic treatment of secondary sludge).

Type	Process	kWh/ton DM
Electricity consumption	Aerobic thermophilic treatment (44% DM)	10*
Electricity production	Anaerobic digestion+incinera. (56% DM)	-180
Total electricity consumption for 1 ton DM (scenario 1 + B)		- 170
Heat consumption	Aerobic thermophilic treatment (44% DM)	40*
	Anaerobic digestion+incinera. (56% DM)	- 540
Total heat consumption for 1 ton DM (scenario 1 + B)		- 500

* Assumed to be ¼ of anaerobic digestion energy consumption

When scenario 1 is combined with scenario C (ultrasound treatment followed by digestion) the energy balance shown in Table 5.9 results. Same picture as for scenario 1 (alone) and scenario "1 + B" but higher net electricity production and heat production due to ultrasound treatment increasing the digestable part of secondary sludge.

Table 5.9 Electricity and heat balance for scenario 1 (primary sludge digested and on-site incinerated) combined with scenario C (ultrasound disintegration followed by anaerobic digestion, AD).

Type	Process	kWh/ton DM
Electricity consumption	Ultrasound disintegration (44% DM)	60
	AD of secondary sludge (44% DM)	30
Electricity production	AD+incin. of primary sludge (56% DM)	- 180
	AD of secondary sludge (44% DM)	- 180
Total electricity consumption for 1 ton DM (scenario 1 + C)		- 270
Heat consumption	AD of secondary sludge (44% DM)	140
Heat production	AD+incin. of primary sludge (56% DM)	- 540
	AD of secondary sludge (44% DM)	- 350
Total heat consumption for 1 ton DM (scenario 1 + C)		- 750

Finally, for the combination of scenario 1 with scenario D (thermal disintegration) the energy balance outcome is shown in Table 5.10. Same general picture as in Table 5.8 and Table 5.9 but the surplus electricity (i.e. negative consumption) is a bit lower and the surplus heat production a bit higher than for scenario 1 combined with ultrasound disintegration.

Table 5.10 Electricity and heat balance for scenario 1 (primary sludge digested and on-site incinerated) combined with scenario D (thermal disintegration followed by anaerobic digestion, AD).

Type	Process	kWh/ton DM
Electricity consumption	Thermal disintegration (44% DM)	180
	AD of secondary sludge (44% DM)	30
Electricity production	AD+incin. of primary sludge (56% DM)	- 180
	AD of secondary sludge (44% DM)	- 210
Total electricity consumption for 1 ton DM (scenario 1 + D)		- 180
Heat consumption	AD of secondary sludge (44% DM)	140
Heat production	AD+incin. of primary sludge (56% DM)	- 540
	AD of secondary sludge (44% DM)	- 430
Total heat consumption for 1 ton DM (scenario 1 + D)		- 830

Ancillary: Besides the ones included in on-site incineration (scenario A1 and scenario 1) no chemicals are included in the inventory (see Appendix 14). The chemicals involved in on-site incineration (off-gas cleaning) are described in Section 5.3.1.

The full GaBi models for the included scenarios are shown below in Figure 5.21, Figure 5.22, Figure 5.23 and Figure 5.24, and some details are described in Appendix 14.

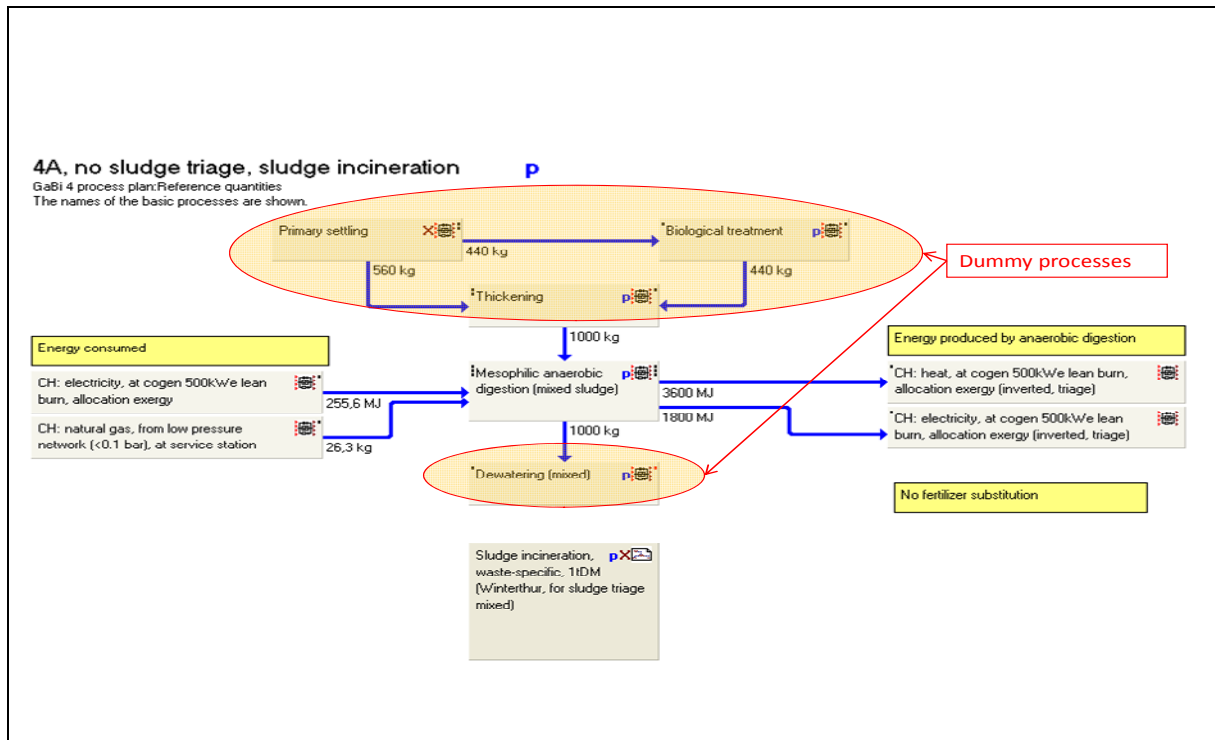


Figure 5.21 GaBi plan showing infrastructure model for scenario A1: On-site incineration of mixed sludge as used in the comparison with sludge triage scenarios

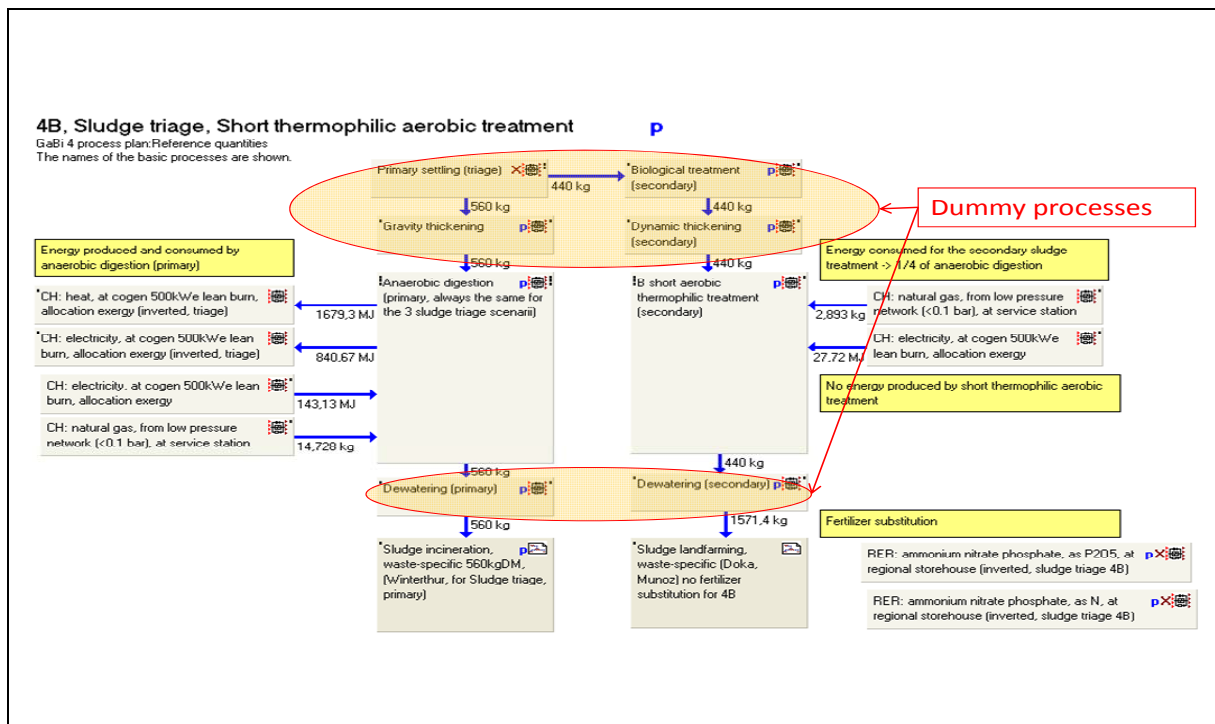


Figure 5.22 GaBi plan showing infrastructure model for sludge triage scenario 1 + B: Primary sludge digested and on-site incinerated combined with aerobic thermophilic treatment of secondary sludge disposed as fertilizer on agricultural land

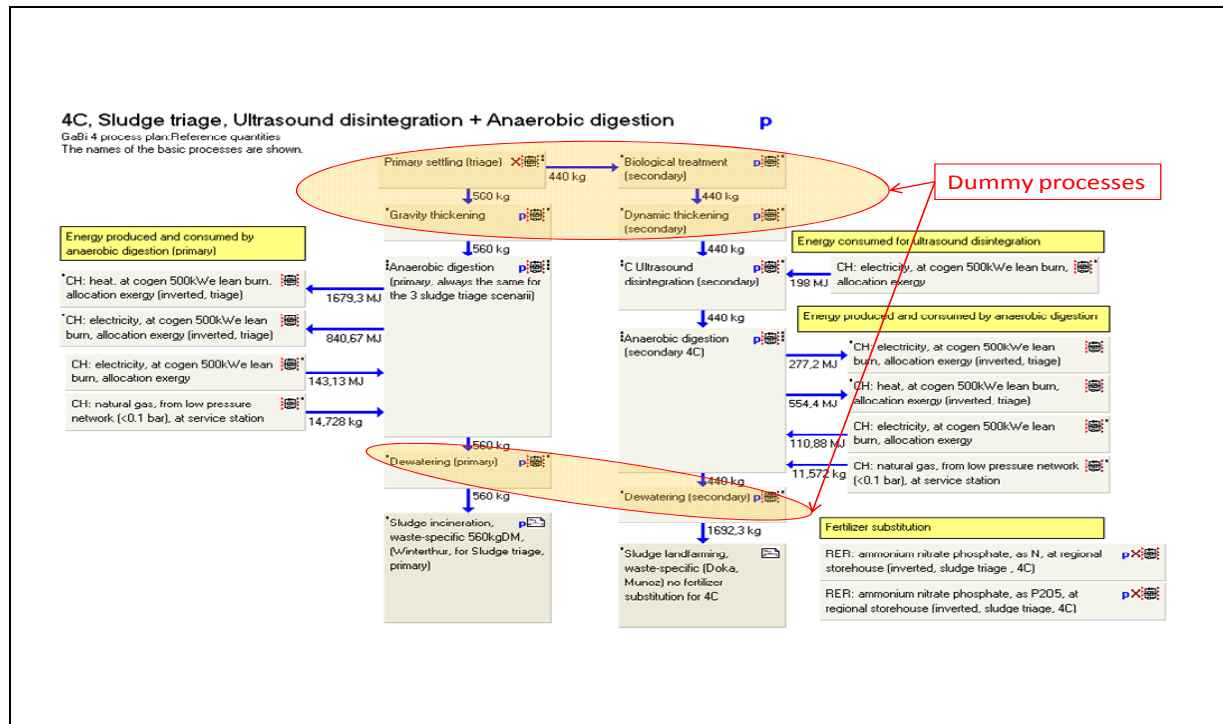


Figure 5.23 GaBi plan showing infrastructure model for sludge triage scenario 1 + C: Primary sludge digested and on-site incinerated combined with ultrasound disintegration of secondary sludge followed by AD and final disposal as fertilizer on agricultural land

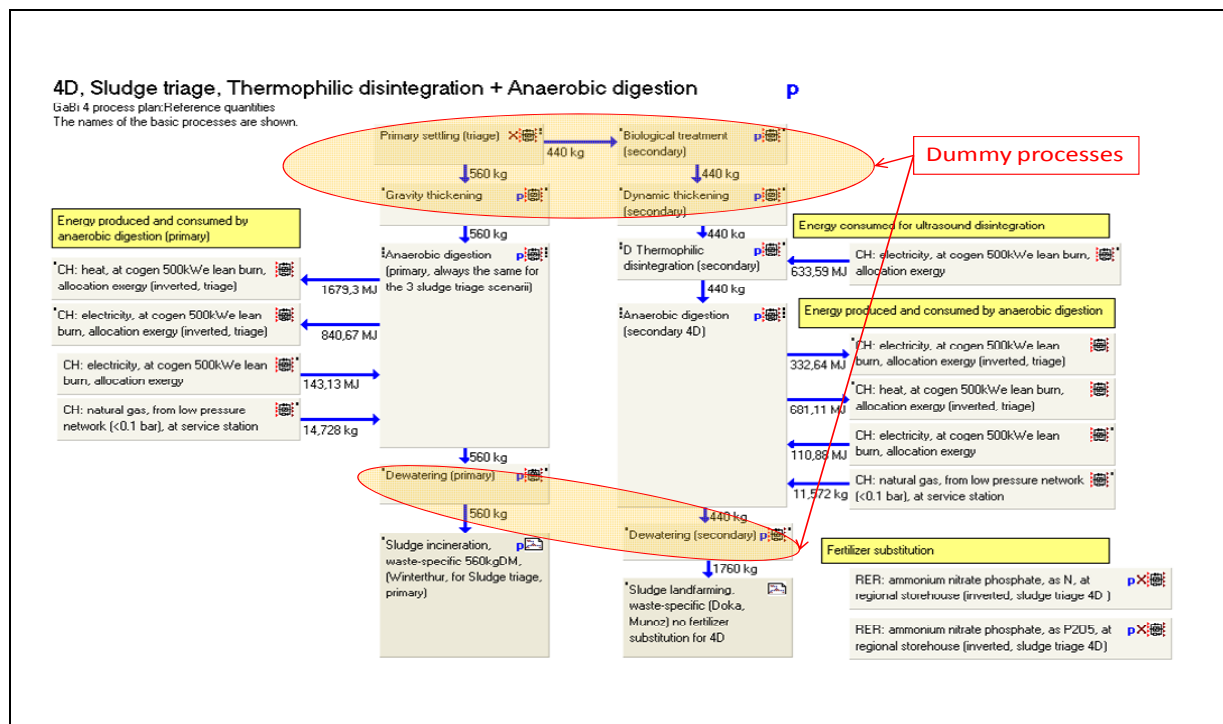


Figure 5.24 GaBi plan showing infrastructure model for sludge triage scenario 1 + D: Primary sludge digested and on-site incinerated combined with thermophilic disintegration of secondary sludge followed by AD and final disposal as fertilizer on agricultural land

5.5 Functional inventory

The functional inventory is mainly based on data from the NEPTUNE partners DPU, Bfg, Hunziker, Eawag, Pyromex, IRSA and Aquafin. Furthermore, literature data from especially Doka (20007b) is included. The inventory consists of influent concentrations and concentrations in sludge, and removal rates of micropollutants in conventional treatment and specific removal rates for each investigated technology and scenarios.

The inlet (sewage) and the outlet (after conventional treatment) concentration for micropollutants used in the reference modeling (cluster 2) is shown in Table 5.11. The outlet concentration (secondary effluent) is used as input to the modeling in cluster 1.

Table 5.11 Concentrations and removal rates for micropollutants and nutrients in model waste water used in LCA modelling of conventional WWT

Substance	Inlet conc. (kg/m ³)	Removal rate (%)	Outlet conc. (kg/m ³)
Atenolol	2,52E-06	37	1,59E-06
Bezafibrat	7,37E-07	88	8,23E-08
Carbamazepin	7,88E-07	16	7,13E-07
Clarithromycin	3,69E-07	47	1,69E-07
Clindamycin	5,90E-08	43	3,36E-08
Clofibrac acid	1,30E-07	37	7,25E-08
Diatrizoate	2,34E-06	21	1,85E-06
Diclofenac	2,00E-06	16	1,55E-06
Erythromycin	2,30E-07	48	9,91E-08
Ibuprofen	5,68E-06	98	9,09E-08
Iohexol	1,79E-06	82	1,85E-07
Iopamidol	3,64E-06	48	1,13E-06
Iopromid	8,28E-06	73	1,78E-06
Metoprolol	5,29E-07	22	4,13E-07
Naproxen	4,24E-06	83	2,33E-07
NDMA	1,32E-07	57	5,68E-08
Primidon	2,52E-07	29	1,72E-07
Propanolol	1,07E-07	11	9,52E-08
Roxithromycin	1,05E-07	57	4,96E-08
Sotalol	4,76E-07	9	4,33E-07
Sulfamethoxazol	8,93E-07	41	4,97E-07
Trimethoprim	3,52E-07	49	1,31E-07
As	9,00E-07	22	7,02E-07
Cd	2,81E-07	50	1,40E-07
Cr	1,22E-05	50	6,12E-06
Cu	3,74E-05	75	9,36E-06
Hg	2,00E-07	70	6,00E-08
Mn	5,30E-05	50	2,65E-05
Ni	6,59E-06	40	3,95E-06
Pb	8,63E-06	90	8,63E-07
Zn	1,09E-04	70	3,28E-05
Tot-P	3,02E-03	72	8,46E-04
Tot-N	2,75E-02	27	2,01E-02

The figures in Table 5.11 regarding metals and nutrients are based on Doka (2007b). Average values based on NEPTUNE data from DPU, Eawag and Bfg is used for the organics in Table 5.11 (calculated as shown in Appendix 5). Other data on influent concentrations and removal rates for conventional treatment are available from other literature sources such as Miege et al. (2009) but it is here chosen to use an average of the measurements performed within NEPTUNE in order to get as relevant concentrations as possible for the comparison among treatment technologies.

The figures for conventional waste water presented in Table 5.11 are considered pretty representative of municipal waste water (sewage and secondary effluent). Removal rates values presented below for the different treatment technologies are more scattered due to the high number of samples required to identify a pattern and the fact that some of the micropollutants are present at concentrations near detection limits. This is especially true for the ozonation data of which some are based on only one sample campaign for each case (see Appendix 6).

The concentrations of micropollutants (i.e. metals) in the reference (standard) sludge types used here are shown in Table 5.12A and Table 5.12B.

Table 5.12A Concentrations of metals in sludge, Italy (based on Bagnuolo et al. 2009)

Substance	Mixed sludge calculated (56% primary and 44% secondary) (mg/kg DM)	Primary sludge (mg/kg DM)	Secondary sludge (mg/kg DM)
As	15	17	13
Cd	0.33	0.33	0.32
Cr	53	57	47
Cu	210	190	230
Hg	0.57	0.57	0.56
Mn	300	200	420
Ni	19	19	19
Pb	100	110	100
Zn	1300	2000	480

The metal concentrations for sludge shown in Table 5.12A are average values based on measurement by IRSA at two Italian waste water treatment plants and reported in Deliverable 1.3 (Bagnuolo et al. 2009). They are about at the same level as values reported by Clauson-Kaas (2006) and Doka (2007b) except especially for cadmium being at the lower end and arsenic at the higher end. These values are used in the LCA modeling of sand filtration (removal of TSS) and in the sludge triage modeling based on inventory data from Italy.

Based on data from NEPTUNE partner Aquafin (Weemaes 2008) European average values for metal content in sludge have been calculated. The Aquafin data comprises samples from European WWTPs in 2007. The raw data (which are confidential – only for use in NEPTUNE) comprises 906 samples of sewage sludge, which can be subdivided into 28 samples from primary sludge only, 728 samples from secondary (or biological) sludge only and 150 samples from primary and secondary sludge mixed. The calculated average values are shown in Table 5.12B.

Table 5.12B Concentrations of metals in sludge, European average (based on Weemaes 2008)

Substance	Mixed sludge calculated (56% primary and 44% secondary) (mg/kg DM)	Primary sludge (mg/kg DM)	Secondary sludge (mg/kg DM)	Mixed sludge (mg/kgDM)
As	11	9.1	13	13
Cd	3.0	2.9	3.3	3.4
Cr	58	59	58	62
Cu	240	220	260	260
Hg	0.7	0.9	0.5	0.6
Ni	22	22	23	26
Pb	130	130	130	150
Zn	1200	1300	1200	1300

The values in Table 5.12B regarding “mixed sludge” are used in the LCA modeling of sludge inertization methods.

Generally the substances included in the study (Table 5.11 and Table 5.12A+B) regarding waste water and sludge are the ones for which useable measurements have been performed within NEPTUNE and for which it has been possible to calculate characterization factors (see Section 6) or in a few cases characterization factors all ready exists . Furthermore, focus has been on substances in common within the different clusters. A few extra substances are included in some of the sensitivity analysis.

Standard deviations are not presented along with the results here. It would however be possible to calculate standard deviations in some cases and in some of the references used, standard deviations are given for some of the figures. However, for the majority of data included here (both physical and functional inventory) standard deviations are not available at all. It is therefore chosen to deal with average data (values) and “uncertainty“ only through sensitivity analysis.

5.5.1 Functional inventory: Cluster 1

In this cluster different technologies for micropollutant removal (water phase) are compared and the reference is direct emission of secondary effluent as defined in Table 5.11 (Outlet conc.). In Table 5.13 the included micropollutants for each of the investigated technologies, i.e. PAC addition in biology (PAC in bio), PAC addition to effluent followed by sand filtration (PAC in effluent + SF), ozonation, sand filtration and ozonation followed by sand filtration (Ozonation + SF), are shown.

Table 5.13 Micropollutants and nutrients included in the functional inventory of the different assessed technologies in cluster 1

Substance	Reference (sec.effl.)	PAC in bio	PAC in effluent+SF	Ozonation	Sand filter	Ozonation + SF
Atenolol	X			X	X	X
Bezafibrat	X	X	X	X	X	X
Carbamazepin	X	X	X	X	X	X
Clarithromycin	X	X	X	X	X	X
Clindamycin	X			X	X	X
Clofibric acid	X	X	X	X	X	X
Diatrizoate	X	X	X	X	X	X
Diclofenac	X	X	X	X	X	X
Erythromycin	X	X	X	X	X	X
Ibuprofen	X	X	X	X	X	X
Iohexol	X	X	X	X	X	X
Iopamidol	X	X	X	X	X	X
Iopromid	X	X	X	X	X	X
Metoprolol	X			X	X	X
Naproxen	X	X	X	X	X	X
NDMA	X			X	X	X
Primidon	X	X	X	X	X	X
Propranolol	X			X	X	X
Roxithromycin	X	X	X	X	X	X
Sotalol	X			X	X	X
Sulfamethoxazol	X	X	X	X	X	X
Trimethoprim	X	X		X	X	X
As	X		X ¹		X ¹	X ¹
Cd	X		X ¹		X ¹	X ¹
Cr	X		X ¹		X ¹	X ¹
Cu	X		X ¹		X ¹	X ¹
Hg	X		X ¹		X ¹	X ¹
Mn	X		X ¹		X ¹	X ¹
Ni	X		X ¹		X ¹	X ¹
Pb	X		X ¹		X ¹	X ¹
Zn	X		X ¹		X ¹	X ¹
Tot-P			X ¹		X ¹	X ¹

¹ Alternative scenarios including the removal of suspended matter by the sand filter

PAC addition in biology: The functional inventory (i.e. removal rates) used for pulverized activated carbon (PAC) addition in biology (see Figure 4.1) is shown in Table 5.14 and based on measurements by DPU at a pilot plant at the WWTP Neuss Süd (Ante et al. 2009)

PAC addition to effluent followed by sand filtration: The functional inventory (i.e. removal rates) used for PAC addition to the effluent followed by sand filtration (see Figure 4.2) is shown in Table 5.15 and based on measurements by DPU at a pilot plant at the WWTP Neuss Süd (Siegrist 2009a).

Table 5.14 Functional inventory (removal rates) for PAC addition in biology

Substance	20 g PAC/m ³ WW Removal rate (%)	40 g PAC/m ³ WW Removal rate (%)	80 g PAC/m ³ WW Removal rate (%)
Bezafibrat	38	54	82
Carbamazepin	79	91	99
Clarithromycin	57	78	97
Clofibric acid	42	28	54
Diatrizoate	12	36	36
Diclofenac	42	57	93
Erythromycin	50	78	n.d.
Ibuprofen	21	0	0
Iohexol	0	0	0
Iopamidol	39	35	52
Iopromid	0	0	0
Naproxen	0	22	72
Primidon	48	59	86
Roxithromycin	53	73	90
Sulfamethoxazol	43	42	66
Trimethoprim	50	90	95

n.d.: No detection

Table 5.15 Functional inventory (removal rates) for PAC addition to effluent followed by sand filtration

Substance	20 g PAC/m ³ WW + sand filter Removal rate (%)
Bezafibrat	94
Carbamazepin	88
Clarithromycin	80
Clofibric acid	41
Diatrizoate	13
Diclofenac	46
Erythromycin	44
Ibuprofen	73
Iohexol	27
Iopamidol	12
Iopromid	35
Naproxen	92
Primidon	40
Roxithromycin	71
Sulfamethoxazol	16

Ozonation of effluent: The functional inventory (i.e. removal rates) used for ozonation (see Figure 4.3) is shown in Appendix 6 and based on measurements by Eawag at a pilot plant at the WWTP ARA Regensdorf (Hollender 2009) also reported in Hollender et al. (2009) and BAFU/Eawag/AWEL/BMG/Hunziker (2009). The removal rates for three of the tested ozone doses are shown in Table 5.16.

Table 5.16 Functional inventory (removal rates in %) for three of the eight tested ozone doses

Substance	1.6g O ₃ /m ³ WW ¹	3.2g O ₃ /m ³ WW ²	5.3g O ₃ /m ³ WW ³	In common all ⁴
Atenolol	32	80	100	x
Bezafibrat	38	62	89	x
Carbamazepin	97	100	100	x
Clarithromycin	83	96	99	x
Clindamycin	n.d.	95	96	
Clofibrac acid	n.d.	66	86	
Diatrizoate	0	0	n.d.	
Diclofenac	97	100	99	x
Erythromycin	0	80	n.d.	
Ibuprofen	n.d.	0	n.d.	
Iohexol	0	0	n.d.	
Iopamidol	46	24	n.d.	
Iopromid	56	26	49	x
Metoprolol	41	88	97	x
Naproxen	97	99	97	x
NDMA	-124	-171	-4	x
Primidon	60	62	91	x
Propranolol	95	90	97	x
Roxithromycin	n.d.	82	n.d.	
Sotalol	96	98	98	x
Sulfamethoxazol	81	95	96	x
Trimethoprim	96	98	97	x

¹ 4.7 mg DOC/L ² 5.5 mg DOC/L ³ 4.6 mg DOC/L ⁴ All eight tested ozone doses (1.6; 1.9; 2.8; 3.2; 3.3; 3.6; 3.7; 5.3)

Sand filtration of effluent: Sand filtration is only included here as a step after ozonation (See Figure 4.3). However, Eawag have done measurements making it possible to estimate removal rates of micropollutants on post ozonation effluent for sand filtration as shown in Appendix 6. Average removal rates for sand filtration (after ozonation) is shown in Table 5.17 and based on the data in Appendix 6.

Sand filtration has a known effect on the suspended matter content of secondary effluent. The removal rate is typically at the level 75% and the total suspended solids (TSS) of secondary effluent is typically at a level of 8 g TSS/m³, according to Siegrist (2008b). The effect on the total-P content is also significant with a typical removal rate of about 62.5% and with a tot-P concentration of 0.8 mg/L for the ingoing secondary effluent, also according to Siegrist (2008b). These levels are confirmed (but at the lower end) by the ones used in the Clauson-Kaas study (Clauson-Kaas et al. 2006) and by Doka (2007b). The TSS is assumed to have the same composition as secondary sludge and therefore metal contents as the ones shown in Table 5.12A (including As and Mn). This removal of TSS by sand filtration, and hereby phosphorus and metals, is included in alternative scenarios for sand filtration alone and PAC addition combined with sand filtration, and ozonation combined with sand filtration.

Ozonation followed by sand filtration: The functional inventory (i.e. removal rates) used for ozonation followed by sand filtration (see Figure 4.3) is shown in Appendix 6 and also based on measurements by Eawag at the pilot plant at the WWTP ARA Regensdorf (Hollender 2009). The removal rates for three of the tested ozone doses are shown in Table 5.18.

Table 5.17 Functional inventory (removal rates in %) for sand filtration of post ozonation effluent (average values of for all eight tested ozone doses)

Substance	Sand filtration (after ozonation) Average removal rates (%)
Atenolol	17
Bezafibrat	23
Carbamazepin	0
Clarithromycin	39
Clindamycin	42
Clofibrac acid	13
Diatrizoate	21
Diclofenac	0
Erythromycin	74
Ibuprofen	87
Iohexol	40
Iopamidol	4
Iopromid	10
Metoprolol	11
Naproxen	0
NDMA	50
Primidon	10
Propranolol	21
Roxithromycin	0
Sotalol	0
Sulfamethoxazol	14
Trimethoprim	0

Table 5.18 Functional inventory (removal rates in %) for ozonation combined with sand filtration

Substance	1.6g O ₃ /m ³ WW ¹ + sand filtration	3.2g O ₃ /m ³ WW ² + sand filtration	5.3g O ₃ /m ³ WW ³ + sand filtration	In common all ⁴
Atenolol	17	90	100	x
Bezafibrat	23	87	90	x
Carbamazepin	94	100	100	x
Clarithromycin	21	99	99	x
Clindamycin	n.d.	95	96	
Clofibrac acid	n.d.	55	n.d.	
Diatrizoate	0	0	n.d.	
Diclofenac	31	100	99	x
Erythromycin	0	80	n.d.	
Ibuprofen	n.d.	82	n.d.	
Iohexol	n.d.	0	n.d.	
Iopamidol	53	35	n.d.	
Iopromid	46	29	58	x
Metoprolol	12	92	97	x
Naproxen	59	98	97	x
NDMA	-22	-54	76	x
Primidon	50	66	91	x
Propranolol	55	90	97	x
Roxithromycin	n.d.	82	n.d.	
Sotalol	47	99	98	x
Sulfamethoxazol	46	96	97	x
Trimethoprim	56	98	97	x

n.d.: No detection ¹ 4.7 mg DOC/L ² 5.5 mg DOC/L ³ 4.6 mg DOC/L ⁴ All eight tested ozone doses (1.6; 1.9; 2.8; 3.2; 3.3; 3.6; 3.7; 5.3)

5.5.2 Functional inventory: Cluster 2

Only conventional waste water treatment and anammox is included here as ICA is dealt with in Deliverable 1.2.

Conventional waste water treatment: The functional inventory for conventional WWT includes all substances presented by Doka (2007b) (nutrients, organics and metals as described in Hansen 2008) including the metals and nutrients as described in Table 5.11. In addition, the “new” 22 organics (mostly pharmaceuticals) presented in Table 5.11 are included as well with their corresponding removal rates.

Regarding micropollutants ending up in the sludge, and therefore later on part of the incineration, only metals are included and transfer coefficients based on data from Doka (2007b) – identical to the removal rates shown in Table 1. Organic micropollutants are assumed to be destroyed during incineration.

Anammox: The anammox case study is based on a limited set of data delivered by the NEPTUNE partner Eawag (Miladinovic 2008a, see Appendix 8) and boils down to a comparison of the following parameters based on an energy and mass balance per functional unit (1 m³ WW):

- Compared to conventional treatment, the installation of an anammox reactor in the supernatant return line, has the following effects:
 - A decrease of 0.092 kWh in electricity consumption
 - An increase of 75.43 mg N₂O emitted to the air
 - An increase in infrastructure input for the anammox reactor

In this case, the energy and infrastructure parameters correspond to the physical inventory while the N₂O emissions are analogous to the functional inventory as they are a function of the water composition, as well as the process.

5.5.3 Functional inventory: Cluster 3

As stated in Section 3.2 the functional unit for sludge treatment is 1 ton (dry matter basis) of anaerobically digested sludge with 4% total solid content.

The functional inventory in this cluster will focus on seven metals (cadmium, chromium, copper, mercury, nickel, lead, zinc) common in the 4 sludge inertization process inventories. The concentrations of the metals in the “standard/reference” sludge used for all processes included are shown in Table 5.12B (mixed). The emissions to air, water and solid waste of these heavy metals is estimated by transfer coefficients, see below for each method.

On-site incineration of sludge: The mass balance used for the NEPTUNE on-incineration process is summarized in Table 5.19 and further detailed in Appendix 9.

Table 5.19 Overall mass balance for on-site incineration of sludge

	Flow	Quantity	Comment
Input	Sludge	1 ton DM	After dewatering (30% DM)
Outputs	Slag	515 kg	Bottom ash and fly ash
	Liquid residue	12 250 L	Returned to WWTP
	Gas	9 250 Nm ³	Total exhaust gas production

In order to model the fate of the metal content in the sludge, transfer coefficients have been estimated for transfer to air (off gas), water via WWTP (liquid residue) and land filling (ash). These distribution factors are shown in Table 5.20 and further described in Appendix 9.

Table 5.20 Transfer coefficients used for modelling fate of metals in sludge incinerated at an on-site incinerator

	Off gas (%)	Liquid residue (%)	Ash (%)
Arsenic (As)	0.00	45.08	54.92
Cadmium (Cd)	0.19	0.78	99.03
Chromium (Cr)	0.11	2.13	97.76
Copper (Cu)	0.01	0.26	99.73
Mercury (Hg)	38.19	56.33	5.48
Nickel (Ni)	0.13	4.01	95.86
Lead (Pb)	0.05	1.42	98.53
Zinc(Zn)	0.23	0.09	99.68
Antimony (Sb)	0.00	39.19	64.81

With the exception of mercury (and to a lesser degree As and Sb) the major part of the metals in sludge incinerated ends up in the ash (see Table 5.20) according to the data used for the estimating (see Appendix 9). The transfer coefficients for arsenic and antimony is not included in the life cycle impact assessment (LCIA) modelling as only the seven other metals are in common with the availability for the other sludge inertization methods.

Wet oxidation of sludge: The mass balance used for the NEPTUNE WO process is summarized in Table 5.21 and further detailed in Appendix 10.

Table 5.21 Overall mass balance for WO of sludge

	Flow	Quantity	Comment
Input	Sludge	1 ton DM	After dewatering (10% DM)
Outputs	Mineral residue	750 kg	Produced during cleaning of liquid
	Solid residue	0 kg	Produced in gas filter
	Rich liquid residue	9 600 L	Digested - producing biogas
	Gas	400 kg	From the reactor

In order to model the fate of the metal content in the sludge, transfer coefficients have been estimated for the transfer to air (off gas), digester (liquid residue to WWTP) and land filling (mineral residue). These distribution factors are shown in Table 5.22 and further described in Appendix 10.

Table 5.22 Transfer coefficients used for modelling fate of metals in sludge treated by wet oxidation

	Off gas (%)	Liquid residue (%)	Mineral residue (%)
Cadmium (Cd)	0.0272	0.66	99.31
Chromium (Cr)	0.0041	0.11	99.88
Copper (Cu)	0.0005	0.10	99.90
Mercury (Hg)	2.2128	0.00	97.79
Nickel (Ni)	0.0176	0.27	99.71
Lead (Pb)	0.0033	0.14	99.86
Zinc(Zn)	0.0000	0.07	99.93

Except for mercury more than 99% of the metals end up in the mineral residue (going to a landfill in the model). According to the data used no significant amounts of mercury end up in the liquid residue and for the other metals it is also (well) below 1%.

High temperature pyrolysis (HTP) of sludge: The mass balance used for the NEPTUNE HTP process is summarized in Table 5.23 and further detailed in Appendix 11.

Table 5.23 Overall mass balance for HTP of sludge

	Flow	Quantity	Comment
Input	Sludge	1 ton DM	After drying (70%-85% DM))
Outputs	Inert solid residue	250 kg	Produced in the pyrolysis reactor
	Solid residue	8 kg	Produced in gas cooling pipes
	Gas	10000 Nm ³	From energetical gas utilisation unit, assumed to be identical to incineration

In order to model the fate of the metal content in the sludge, transfer coefficients have been estimated for the transfer to air (off gas), inert material landfill (inert residue) and residual material landfill (solid residue). These distribution factors are shown in Table 5.24 and further described in Appendix 11.

Table 5.24 Transfer coefficients used for modelling fate of metals in sludge treated by high temperature pyrolysis

	Off gas (%)	Inert residue (%)	Solid residue (%)
Cadmium (Cd)	0.50	97.0	2.5
Chromium (Cr)	0.16	87.3	12.6
Copper (Cu)	0.09	97.3	2.6
Mercury (Hg)	3.33	-	-
Nickel (Ni)	0.25	89.8	9.9
Lead (Pb)	0.07	84.5	15.2
Zinc(Zn)	0.13	49.4	50.5

Except for mercury (for which data are missing) and zinc more than 84% of the metals end up in the inert residue (going to an inert material land fill).

Middle temperature pyrolysis (gasification) of sludge: The mass balance used for the NEPTUNE gasification process is summarized in Table 5.25 and further detailed in Appendix 12.

Table 5.25 Overall mass balance for gasification of sludge

	Flow	Quantity	Comment
Input	Sludge	1 ton DM	After drying (70%-85% DM))
Outputs	Mineral granulate	500 kg	Produced in the gasification reactor
	Solid residue	1 kg	Produced in gas filter
	Condensate after gas drying	350 L	Returned to WWTP
	Off-gas	1 765Nm ³	From energetical gas utilisation unit

In order to model the fate of the metal content in the sludge transfer coefficients have been estimated for the transfer to air (off gas), inert material landfill (mineral granulate), residual material landfill (solid residue/gas filter residue) and WWTP (condensate after gas drying). These distribution factors are shown in Table 5.26 and further described in Appendix 12.

Table 5.26 Transfer coefficients used for modelling fate of metals in sludge gasification

	Off gas (%)	Mineral granulate (%)	Solid residue (%)	Condensate (%)
Arsenic (As)	0.20	99.44	-	0.36
Cadmium (Cd)	0.00	15.63	84.20	0.18
Chromium (Cr)	0.11	99.84	-	0.05
Copper (Cu)	0.01	99.91	-	0.07
Mercury (Hg)	0.15	0.28	99.55	0.01
Nickel (Ni)	0.13	99.75	-	0.12
Lead (Pb)	0.01	20.83	79,10	0.06
Zinc(Zn)	0.00	99.87	-	0.13

The major part of all metals ends up in landfills according to the transfer coefficient balance in Table 5.26.

5.5.4 Functional inventory: Cluster 4

The functional inventory for the sludge triage assessments only relates to metal emissions and nutrients (tot-N and tot-P) in the treated sludge applied on agricultural land substituting mineral fertilizers.

As regards metals the concentrations in the different types of sludge defined in Table 5.12A is used. These data are included in the modelling of sludge application on agricultural land based on the model by Doka (2007b) in a simplified version developed by Munoz et al. (2007) (“Sludge landfarming, waste-specific (Doka, Munoz) no fertilizer substitution for 4C”). Besides the impacts related to spreading metals (in the sludge) on soil, this model includes impacts from transporting the sludge from the WWTP to the agricultural land (56 km) and impacts from the “spreading equipment” fuel consumption and more. Regarding the sludge part going to on-site incineration the relevant metal concentrations of Table 5.12A is included in the model developed for on-site incineration (see Section 5.3.1).

The phosphorus and nitrogen content of the secondary sludge after the different treatments, i.e. short aerobic thermophilic, ultrasound + AD and thermal + AD, is based on the mass balances from NEPTUNE partner IRSA (Deliverable 1.3: Bagnuolo et al. 2009, including excel sheet from Giuseppe Mininni). These nutrient concentrations are reported in Appendix 14.

The substitution of mineral fertilizer by the nutrient content of the sludge is modelled by the following EcoInvent processes:

- RER: ammonium nitrate phosphate, as P₂O₅, at regional storehouse
- RER: ammonium nitrate phosphate, as N, at regional storehouse

See further description in Appendix 14.

6 Life cycle impact assessment (LCIA)

After completing the life cycle inventory, all flows and emissions undergo life cycle impact assessment (LCIA) through the following steps as described in Deliverable 4.2 (Larsen et al. 2009):

- Classification
- Characterization
- Normalization
- Weighting

This is done in order to express the environmental burdens associated with the inventory as quantities within the environmental impact categories presented in Section 3.3. In the present study, most of the LCIA is carried out internally by GaBi, configured as described in Section 7. However, the existing GaBi tool has not modeled and do not include characterisation factors for the 22 pharmaceuticals (and NDMA) listed in Table 5.11. This is because they did not exist before but have been developed as part of NEPTUNE and incorporated in a special NEPTUNE database as part of the software. Furthermore, the quality of some of the existing characterization factors, e.g. the metals shown in Table 6.1, was improved (updated and recalculated).

6.1 EDIP characterization factors for pharmaceuticals and more

Regarding the missing characterization factors (CFs) and the updating of some existing ones for metals, ecotoxicity characterization factors for emissions to fresh water have been calculated according to EDIP97 (Hauschild et al. 1998) taking the modification as described in Deliverable 4.2 (Larsen et al. 2009) into account. Typically when calculating ecotoxicity characterization factors electronic data bases with ecotoxicity data are used but this was not possible for the pharmaceuticals (new regarding environmental focus and data currently being produced) which for the major part had to be based on original literature. The calculated characterization factors regarding fresh water ecotoxicity for emissions to fresh water is shown in Table 6.1. For the pharmaceuticals (and NDMA) the data sources, i.e. mostly scientific papers, are compiled in Appendix 7, together with an indication of the data quality on which the CFs are calculated.

As may be seen from Table 6.1 the CFs for the included pharmaceuticals have a range of about seven orders of magnitude with the contrast agents (e.g. iohexol and iopamidol) having the least ecotoxic impact potential whereas the beta-blocker propranolol and the antibiotics (e.g. erythromycin and sulphamethoxazol) are among the substances with the highest ecotoxic impact potential. If the CFs are based on the proposed German environmental quality standards (LAWA project I+II 2004/2006) especially the pain killer diclofenac change position and achieve the third highest ecotoxic impact potential.

Table 6.1 Fresh water ecotoxicity characterisation factors (CFs) based on calculated PNECs (“standard”) or conservative PNECs (German Environmental Quality Standards)

Substance	PNECs (µg/L)		Ecotox fresh water CFs (m ³ /kg)*	
	”Standard”**	Conservative RA based**	”Standard”**	Conservative RA based**
Atenolol	330		2.99E+03	
Bezafibrat	2.3		4.35E+05	
Carbamazepin	2.5	0.5	4.00E+05	2.00E+06
Clarithromycin	0.31		3.23E+06	
Clindamycin	8.5		1.17E+05	
Clofibrac acid	25	5	4.07E+04	2.00E+05
Diatrizoate	11,000		9.09E+01	
Diclofenac	100	0.1	1.00E+04	1.00E+07
Erythromycin	0.20	0.02	5.00E+06	5.00E+07
Ibuprofen	96	3	5.21E+03	1.67E+05
Iohexol	7,400,000		1.36E-01	
Iopamidol	380,000		2.65E+00	
Iopromid	100,000		1.00E+01	
Metoprolol	76	7.3	1.32E+04	1.37E+05
Naproxen	190		5.18E+03	
NDMA (N-nitrosodimethylamin)	40		2.50E+04	
Primidon	1,400		6.94E+02	
Propanolol	0.050		2.00E+07	
Roxithromycin	2.8		3.56E+05	
Sotalol	300		3.33E+03	
Sulfamethoxazol	0.59	0.15	1.69E+06	6.67E+06
Trimethoprim	800		1.25E+03	
Cadmium (Cd)***	0.19		5.26E+06	
Nickel (Ni)***	1.8		5.56E+05	
Lead (Pb)***	0.41		2.44E+06	

* Calculation principles as described in Deliverable 4.2 (Larsen et al. 2009), for EDIP97 ** LAWA project I+II (2004/2006)

*** Clauson-Kaas et al. (2006)

6.2 Environmental quantities in GaBi related to EDIP97

While the pharmaceuticals presented in Table 6.1 are only characterized by their contribution to the EDIP97 environmental impact category on “ecotoxicity water chronic”, all other flows in Ecolnvent may be characterized within any of the EDIP categories. These categories include all the main environmental impact categories on which this study focuses, as presented in Section 3.3.

6.3 Note regarding normalization references

Normalization references in the EDIP methodology are calculated based on the average person’s environmental impact within a given impact category, a given geographical area and a given time period. To do this, emissions are inventoried from all perceived potential sources in this area over the specified time period. So, rigorously speaking, the normalization reference is valid for any emission included within the inventoried emissions and sources.

Due to the relatively new interest in the environmental impact of the pharmaceuticals listed in Table 6.1,2 these substances were not included in the original inventory (Stranddorf et al. 2005) from which the normalization reference on aquatic ecotoxicity was obtained. As some

of these pharmaceuticals have a relatively high toxicity they could potentially have a significant impact on the normalization reference for aquatic ecotoxicity. For normalized (and weighted) impact potentials (e.g. impact profiles of WWTTs) that could lead to an overestimation of the potential impact from the pharmaceuticals, i.e. the impact category on aquatic ecotoxicity, as compared to the other impact categories (e.g. global warming potential). However, this problem have been investigated as part of NEPTUNE and reported in Deliverable 4.2 (Larsen et al. 2009), showing that based on existing knowledge normalization references are not significantly changed when the pharmaceuticals are included. Normalization references for the most dominating impact categories in the LCIA of the WWTTs included in NEPTUNE are shown in Table 6.2.

Table 6.2 Normalisation references (NRs) and weighting factors (WFs) regarding EDIP97 for the eight most dominating impact categories included in NEPTUNE. The weighting factors (WFs) are not included as default in NEPTUNE instead WF = 1 for all impact categories is used

Impact category	Unit for impact	Normalisation reference (NR, 1994) 1 PE	Region	Weighting factor (WF, 2004) 1 PET
Ecotoxicity water	Cubic meter water	352,000 m ³ /capita/year	EU-15	1.18
Ecotoxicity soil	Cubic meter soil	964,000 m ³ /capita/year	EU-15	1
Human toxicity water	Cubic meter water	52,200 m ³ /capita/year	EU-15	1.3
Human toxicity soil	Cubic meter soil	127 m ³ /capita/year	EU-15	1.23
Photochemical oxidation	Kg C ₂ H ₄ -eq	25 kg/capita/year	EU-15	1.33
Nutrient enrichment	Kg NO ₃ ⁻ -eq	119 kg/capita/year	EU-15	1.22
Acidification	Kg SO ₂ -eq	74 kg/capita/year	EU-15	1.27
Global warming	Kg CO ₂ -eq	8,700 kg/capita/year	Global	1.12

(Stranddorf et al. 2005)

From Table 6.2 it is evident that the maximum difference between weighting factors is only 33%. As decided among NEPTUNE partners these weighting factors (WFs) have not been used as default in NEPTUNE – instead a general WF=1 is used putting equal weight on all impact categories. For discussions about weighting please see Deliverable 4.2 (Larsen et al. 2009).

7 Running EDIP97 on GaBi 4.3/EcoInvent 2.0 integrated

In order to carry out LCAs based on EDIP97 on GaBi 4.3 with the EcoInvent 2.0 integrated database, it is necessary to configure GaBi. The changes necessary apply in the following GaBi data categories:

- EcoInvent flows: long-term emissions
- Environmental quantities: EDIP 1997

Long-term emissions result as default in the present case from EcoInvent modeling emissions from land filled materials. In the EcoInvent models, the total potential emissions resulting from land filling processes including leaching and emissions to air are considered and modeled as long-term emissions. As the name implies, this includes emissions potentially released after 100 years in the landfill and beyond (until 60,000 years). Short-term emissions are defined as emissions appearing until 100 years (Doka G 2007a) and are included for most of the land fill types.

As proposed in Deliverable 4.2 (Larsen et al. 2009), only landfill emissions occurring in the first 100 years are to be considered along with all other inventoried emissions under the EDIP97 impact categories introduced in Section 3.3. All other emissions beyond this point should be included in a new category of stored toxicity. Therefore, it is necessary to change the GaBi configuration in order to avoid including the stored toxicity emissions in the other impact categories.

This problem is solved in the present study by removing completely all long-term emissions from the inventory. This is done in the “Define balance view” option of individual balances by showing only elementary flows and further removing completely the long-term emissions flow category. The result of this change in configuration is that the present study does not include long-term emissions (or stored toxicity) if not stated explicitly.

Environmental quantities in GaBi refer to impact categories as defined in EDIP97. As such, this data category needs to be modified to account for some errors in characterization detected when looking at the results. Therefore, the following shows which EDIP97 impact categories and which flows within those categories need to be modified. Note that the changes apply only when using the newly modified impact categories in the balance calculation.

- *Ecotoxicity in water, acute*: this impact category is special for EDIP97 as all other LCIA methods typically only includes chronic ecotoxicity. The amount of pollutants emitted per functional unit is typically very small making only chronic effects relevant. Taking this into account and with the aim of not putting too much weight on aquatic ecotoxicity (both acute and chronic effects of the same emission on the same endpoint (i.e. biota) in the same compartment, i.e. water) acute toxicity is not included here. However, it may be relevant in special cases, e.g. waste water emission of ammonia, including whole effluent toxicity and more as described in Deliverable 4.2 (Larsen et al. 2009).
- *Acidification potential*: in this impact category, the characterization factor for sulphate is set to 0. Initially it was set by GaBi as the same as sulphuric acid but the error was spotted when examining results.
- *Ecotoxicity water, chronic*: the characterization factor for magnesium emissions to water is set to 0. Initially it was set by GaBi as the same as manganese thereby overestimating its potential impact.
- *Human toxicity, soil*: the characterization factor for all benzene emissions is set to 0 because it is assessed that due to the soil absorption and the human exposure routes from soil the potential impacts will be insignificant. The same goes for chlorine emissions to soil and all iron emissions. All nitrogen oxides emissions to air have a characterization factor of 0 because they are expected to land on the soil as nitrates, with no significant toxicity effects.
- *Human toxicity, water*: the characterization factor all magnesium emissions to water is set to 0. Initially it was set by GaBi as the same as manganese thereby overestimating its potential impact. All nitrogen oxides emissions to air have a characterization factor of 0 because they are expected to land on the water as nitrates, with no significant toxicity effects.

The changes listed above are implemented when using the “Neptune” normalization reference and the “Neptune” or “Neptune (no WF)” weighting reference in the balance of a NEPTUNE plan (implemented in the GaBi database-version created as part of the NEPTUNE work). These settings are used as default when doing NEPTUNE LCIA. The “NEPTUNE (no WF)” weighting reference refers to the setting where all weighting factors are set to 1, which is also used as default here. The other weighting reference included in the GaBi database-version created as part of the NEPTUNE work uses the standard EDIP weighting factors as shown in Table 6.2.

8 Results and interpretation

The results of doing life cycle impact assessments on the included WWTTs according to the principles and data described in previous sections are presented in this section. Results are in most cases shown as impact profiles based on total normalized and weighted (WF=1) impact potentials covering all the included impact categories and expressed in units of person-equivalents-targeted (PET). In cluster 1 and cluster 2 these impacts profiles are presented as avoided against induced impacts. The presentation of the results are divided into the clusters defined in Section 4 and addresses the questions on environmental sustainability as defined in Section 3.1.

8.1 Cluster 1: WWTTs for micropollutant removal

In this section, the results for the WWT technologies on ozonation, sand filtration and PAC addition are presented in terms of avoided vs. induced impacts. But first the impact profile for the reference situation, i.e. direct emission of secondary effluent, will be shown.

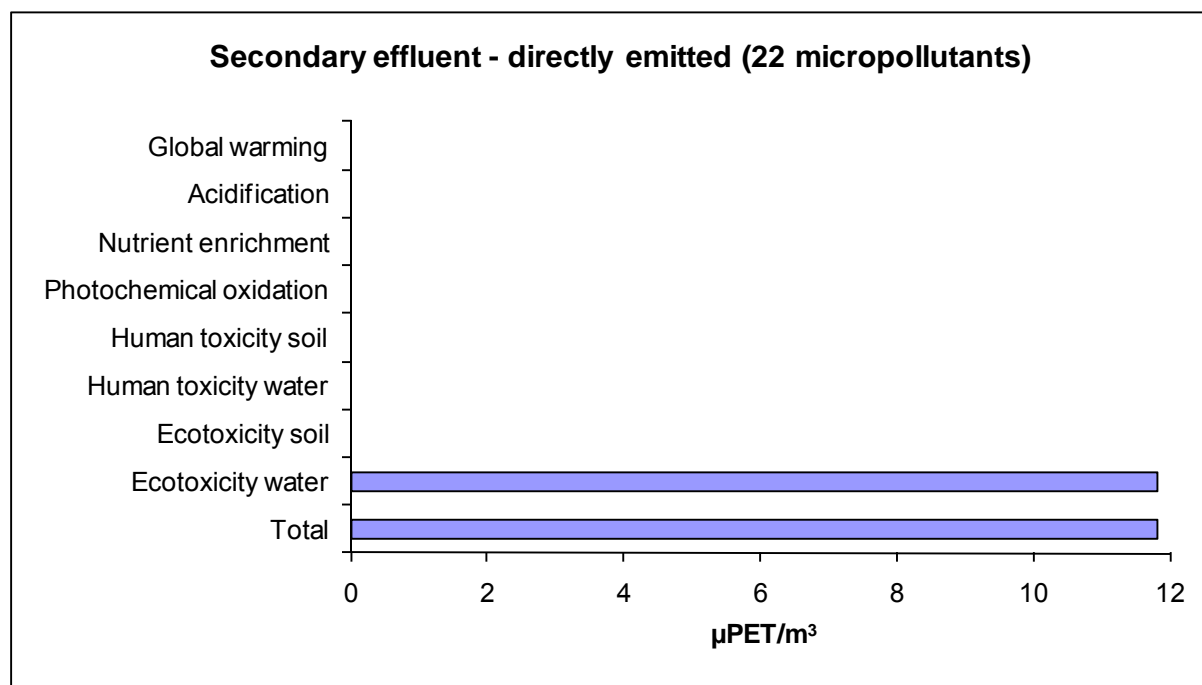


Figure 8.1 Normalized and weighed LCA impact profile for the reference situation i.e. direct emission of secondary effluent to recipient. Based on the outlet concentrations shown in Table 5.11 for the 22 organic micropollutants. Weighting factor for all impact categories equals 1

As evident from Figure 8.1 the direct emission of the 1 m³ waste water (secondary effluent) defined by the micropollutant content as described in Table 5.11 (nutrients and metals excluded) only contribute to the impact category on “Ecotoxicity water”. Therefore both the impact potential on ecotoxicity in water and the total equals 11.8 µPET/m³. As there is no infrastructure or energy consumption involved in this profile this result is also expected.

8.1.1 Ozonation

When assessing the environmental sustainability of ozonating the secondary effluent, depicted in Figure 8.1, we have to include the potential impact from building, running and demolishing the ozonator. The result is shown as an impact profile for ozonating at a dose of 3.2 g O₃/m³ in Figure 8.2, where the infrastructure, energy consumption etc. are included as described in Section 5.1.1.

As may be observed in Figure 8.2 we are now introducing potential impact in all included impact categories (especially global warming) and the total is now composed of contributions from all impact categories and amounts to 11.2 $\mu\text{PET}/\text{m}^3$. The blue (light) bars in Figure 8.2 shows the remaining potential impact (as related to “Ecotoxicity water” in Figure 8.1) that has not been removed by ozonation (i.e. 1.1 $\mu\text{PET}/\text{m}^3$) and the red (dark) bars shows the induced impact (due to electricity consumption etc.). In this case the induced impact potential therefore becomes 10.2 $\mu\text{PET}/\text{m}^3$ while the avoided part becomes 10.7 $\mu\text{PET}/\text{m}^3$ (11.8 – 1.1).

This result may be presented in another way showing the total induced impacts versus the total avoided impacts. This type of impact profile is depicted in Figure 8.3.

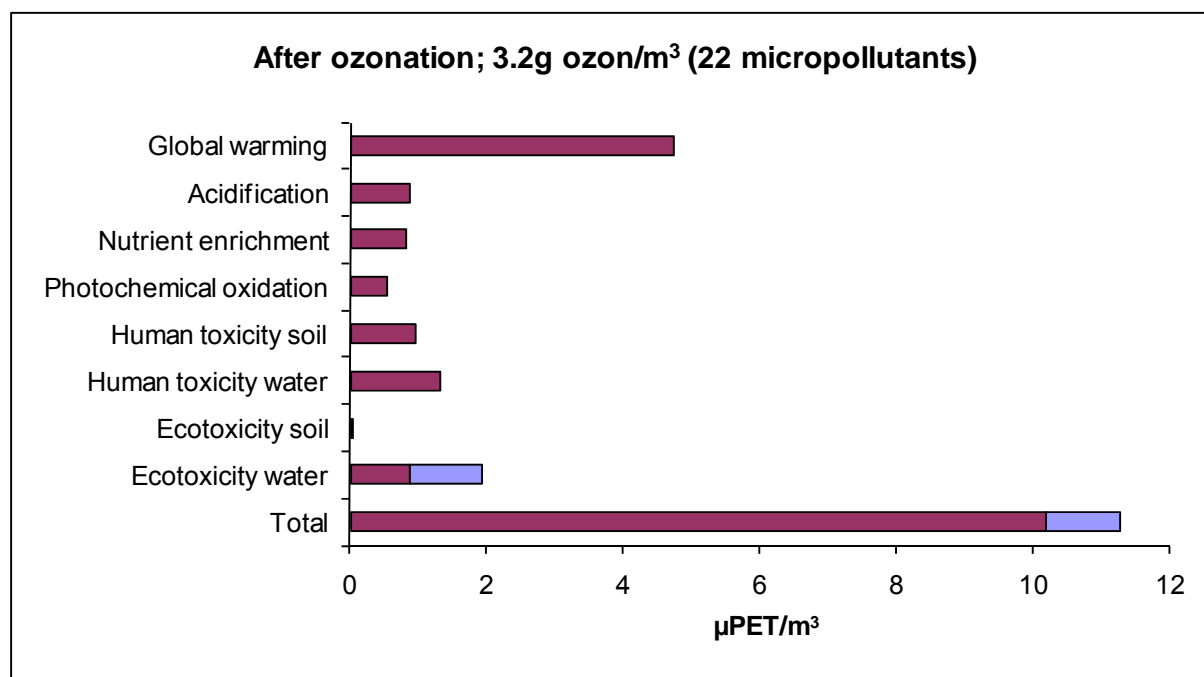


Figure 8.2 Normalized and weighed LCA impact profile for ozonating secondary effluent waste water at a dose of 3.2 g O₃/m³. Based on the functional inventory shown in Table 5.16 and the physical inventory as described in Section 5.1.1. Weighting factor for all impact categories equals 1

As is evident from Figure 8.3 the induced impacts are composed of contributions from all three process/activity categories, i.e. energy consumption, ancillary and infrastructure. Regarding the avoided part especially the removal of propanolol, sulfamethoxazole, clarithromycin, erythromycin and carbamazepin is contributing.

Besides the ozone dose of 3.2 g O₃/m³ seven other doses have been tested. The results for all doses are shown in Figure 8.4

As may be seen in Figure 8.4 the induced impact is successively increased as the ozone dose increases. This is due to the related increasing energy consumption and oxygen consumption (ancillary). On the contrary as the ozonator is the same for all doses the contribution to induced impact from infrastructure is constant. As regard the avoided impacts there is apparently also an increasing tendency but much weaker and blurred by the fact that removal rates are missing for some of the micropollutants (e.g. erythromycin) regarding some of the doses. The results shown in Figure 8.4 may be presented as avoided minus induced, see Figure 8.5, and hereby indicate that there might be a break-even (between 3.3 g O₃/m³ and 3.6 g O₃/m³) where the avoided impact neutralize the induced impact given the scoping and assumptions included in this assessment.

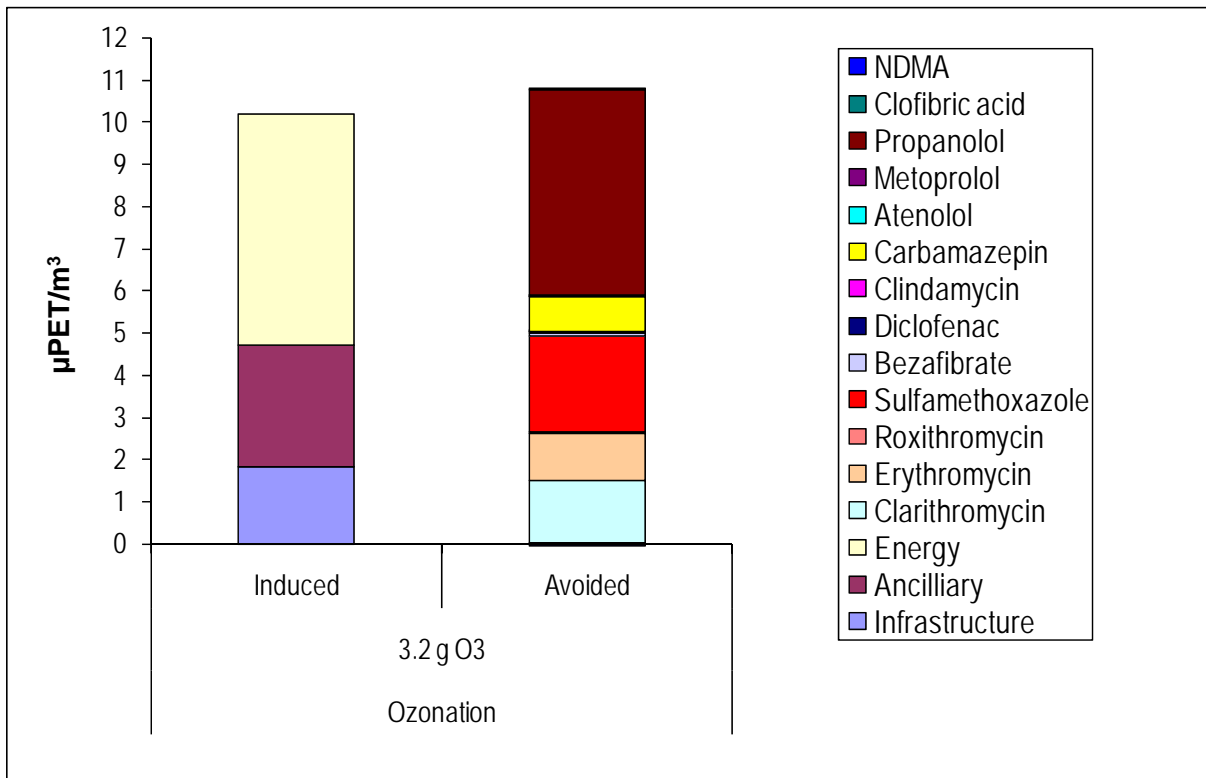


Figure 8.3 Normalized and weighed LCA impact profile showing induced and avoided impacts for ozonating secondary effluent at a dose of 3.2 g O₃/m³. Based on the functional inventory shown in Table 5.16 and the physical inventory as described in Section 5.1.1. Weighting factor for all impact categories equals 1

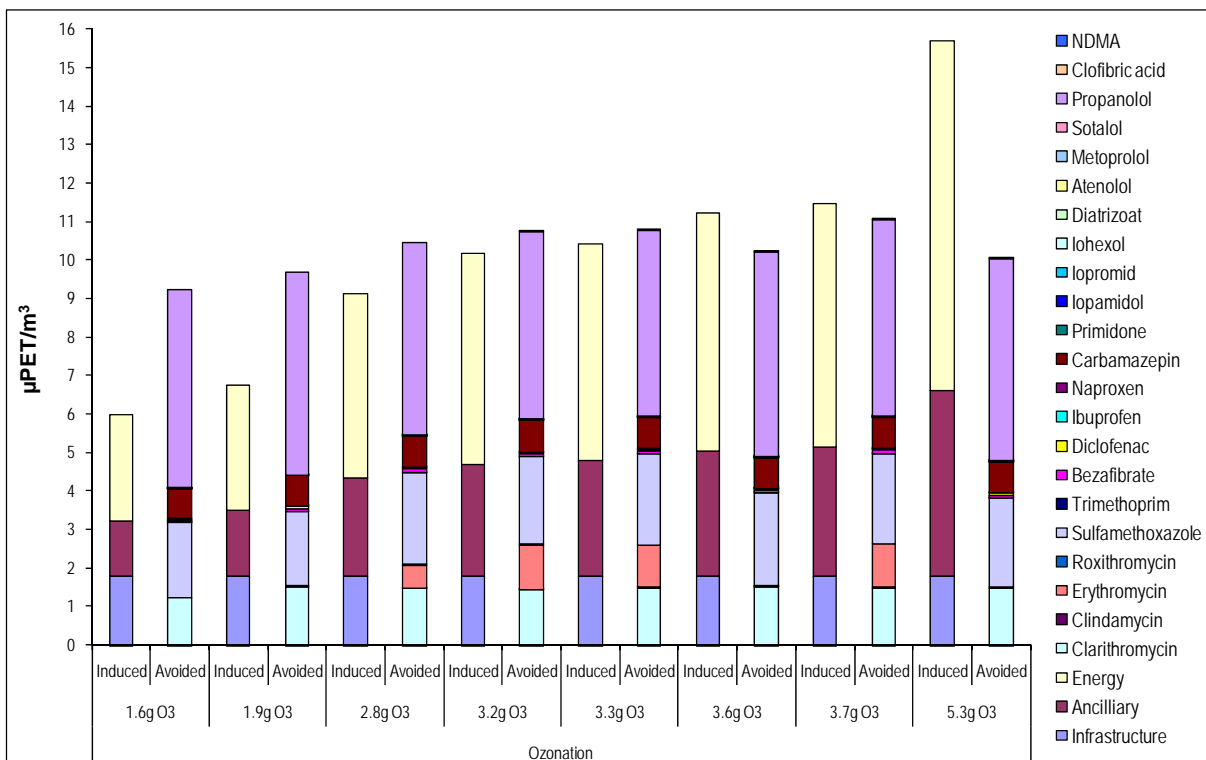


Figure 8.4 Normalized and weighed LCA impact profiles showing induced and avoided impacts for ozonating secondary effluent at all included doses. Based on the functional inventory shown in Appendix 6 and the physical inventory as described in Section 5.1.1. Weighting factor for all impact categories equals 1

As may be seen in Figure 8.4 the induced impact is successively increased as the ozone does increases. This is due to the related increasing energy consumption and oxygen consumption (ancillary). On the contrary as the ozonator is the same for all doses the contribution to induced impact from infrastructure is constant. As regard the avoided impacts there is apparently also an increasing tendency but much weaker and blurred by the fact that removal rates are missing for some of the micropollutants (e.g. erythromycin) regarding some of the doses. The results shown in Figure 8.4 may be presented as avoided minus induced, see Figure 8.5, and hereby indicate that there might be a break-even (between 3.3 g O₃/m³ and 3.6 g O₃/m³) where the avoided impact neutralize the induced impact given the scoping and assumptions included in this assessment.

It should be noted that the environmental sustainability profiles shown here (Figure 8.2 - Figure 8.5) does not include the potential effect of metabolites (e.g. aldehydes) created during the ozonation and reflected by whole effluent toxicity of the post ozonation effluent (Stalter et al 2010a; Stalter et al. 2010b). This problem is addressed by combining ozonation with sand filtration (see Section 8.14).

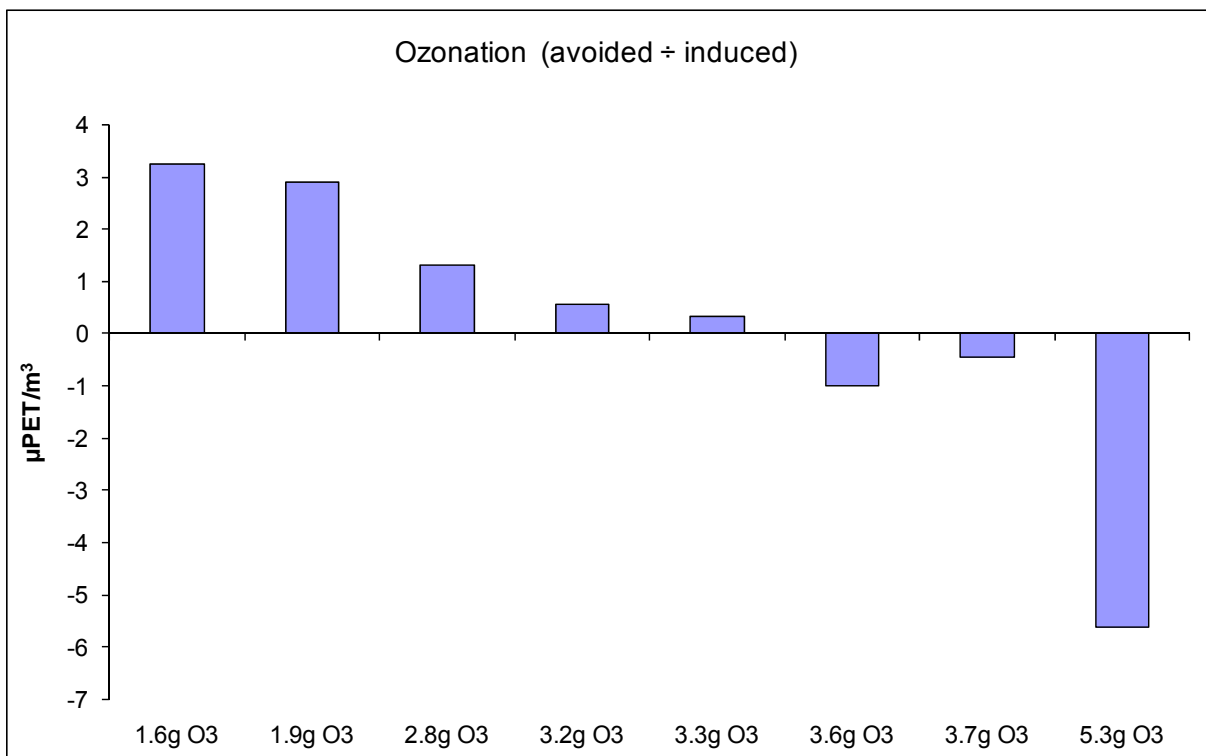


Figure 8.5 Normalized and weighed LCA impact profiles shown as induced minus avoided impacts for ozonating secondary effluent at all included doses Based on the functional inventory shown in Appendix 6 and the physical inventory as described in Section 5.1.1. Weighting factor for all impact categories equals 1

8.1.2 Sand filtration

The assessment of sand filtration is here based on removal rates observed for post ozonation effluent as compiled in Table 5.17. The results are shown in Figure 8.6.

As shown in Figure 8.6 the induced impacts are composed of contributions from only two categories, i.e. energy consumption and infrastructure, as no ancillary is involved. Regarding the avoided part especially the removal of erythromycin, propanolol, clarithromycin and sulfamethoxazole is contributing.

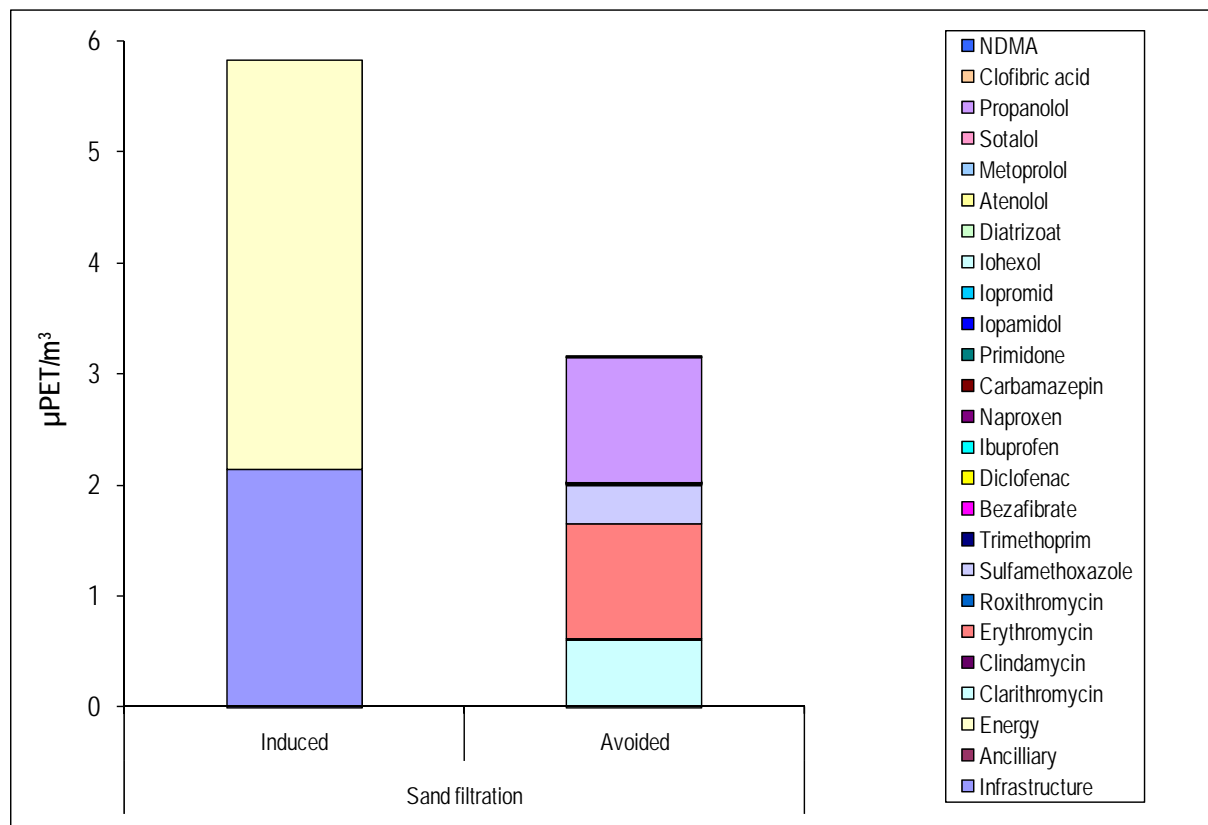


Figure 8.6 Normalized and weighed LCA impact profile showing induced and avoided impacts for sand filtration of secondary effluent. Based on the functional inventory shown in Table 5.17 and the physical inventory as described in Section 5.1.2. Weighting factor for all impact categories equals 1

8.1.3 PAC addition

The LCA impact profile for adding pulverized activated carbon (PAC) to the biological step in waste water treatment is shown in Figure 8.7 for the dose 20 g PAC/m³.

As evident from Figure 8.7 the induced impact is substantial higher than the avoided part and is dominated by the potential impact from PAC production (ancillary). Regarding the much smaller avoided part especially the removal of clarithromycin, erythromycin and carbamazepin is contributing.

The dominating role of the PAC productions becomes even more pronounced when all three tested PAC addition doses are shown, see Figure 8.8. At the 80 g PAC dose the difference between induced and avoided impacts is a factor of 73.

It should be noted that the environmental sustainability profiles shown here (Figure 8.7 and Figure 8.8) does not include the potential effect of PAC ending up in the recipient. This problem is taken care of by combining PAC addition to secondary effluent with sand filtration (see Section 8.1.5).

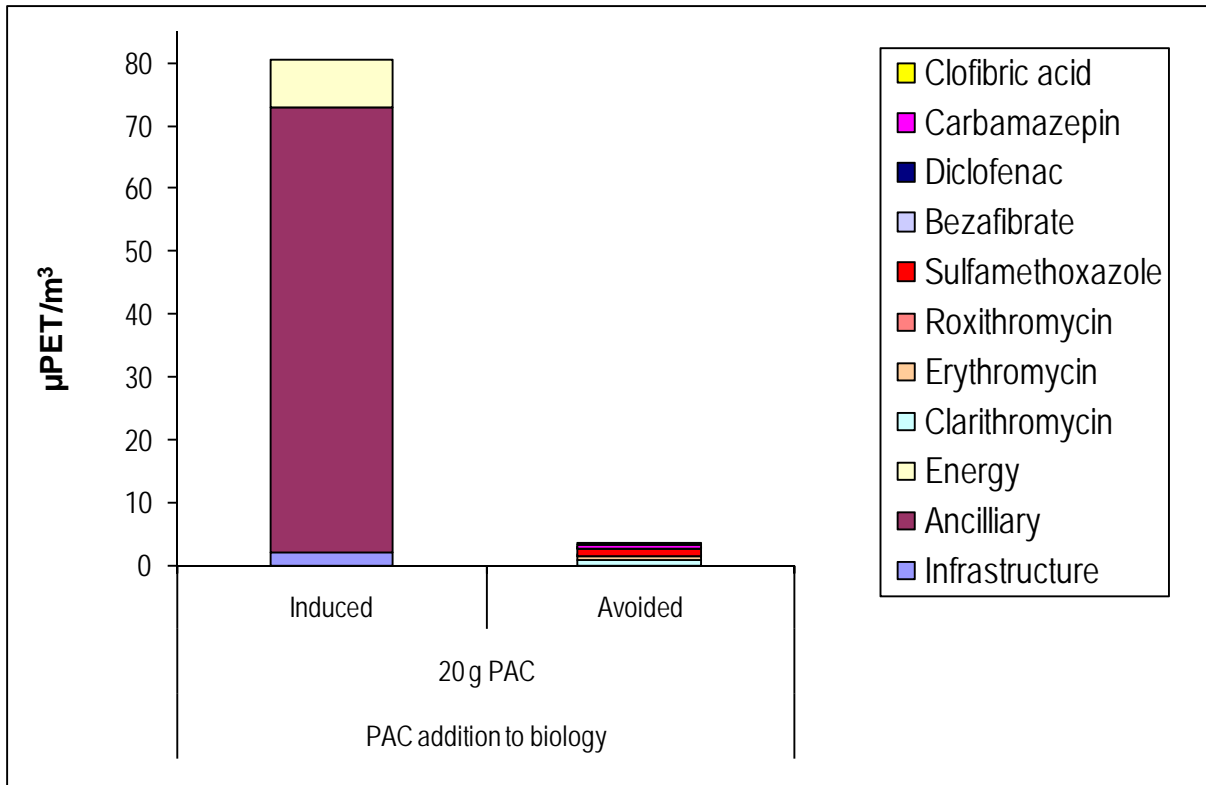


Figure 8.7 Normalized and weighed LCA impact profile showing induced and avoided impacts for PAC addition to biology at a dose of 20 g PAC/m³ regarding secondary effluent. Based on the functional inventory shown in Table 5.14 and the physical inventory as described in Section 5.1.3. Weighting factor for all impact categories equals 1

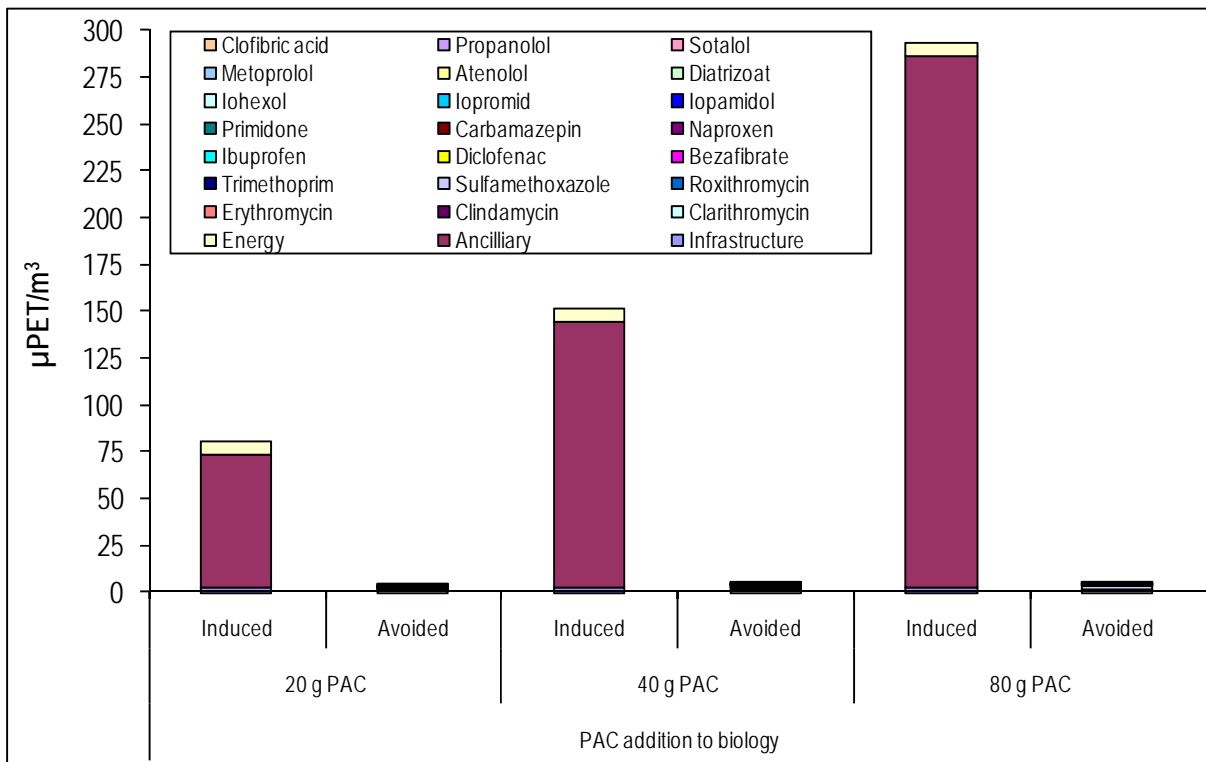


Figure 8.8 Normalized and weighed LCA impact profiles showing induced and avoided impacts for PAC addition to biology regarding secondary effluent at all included doses. Based on data as described for Figure 8.7 and weighting factor for all impact categories equals 1

8.1.4 Ozonation combined with sand filtration

With the aim of trying to solve the problem with whole effluent toxicity of post ozonation effluent (Stalter et al 2010a; Stalter et al. 2010b) ozonation have been combined with post sand filtration in NEPTUNE. The resulting LCA impact profile for the dose of 3.2 g O₃/m³ is shown in Figure 8.9.

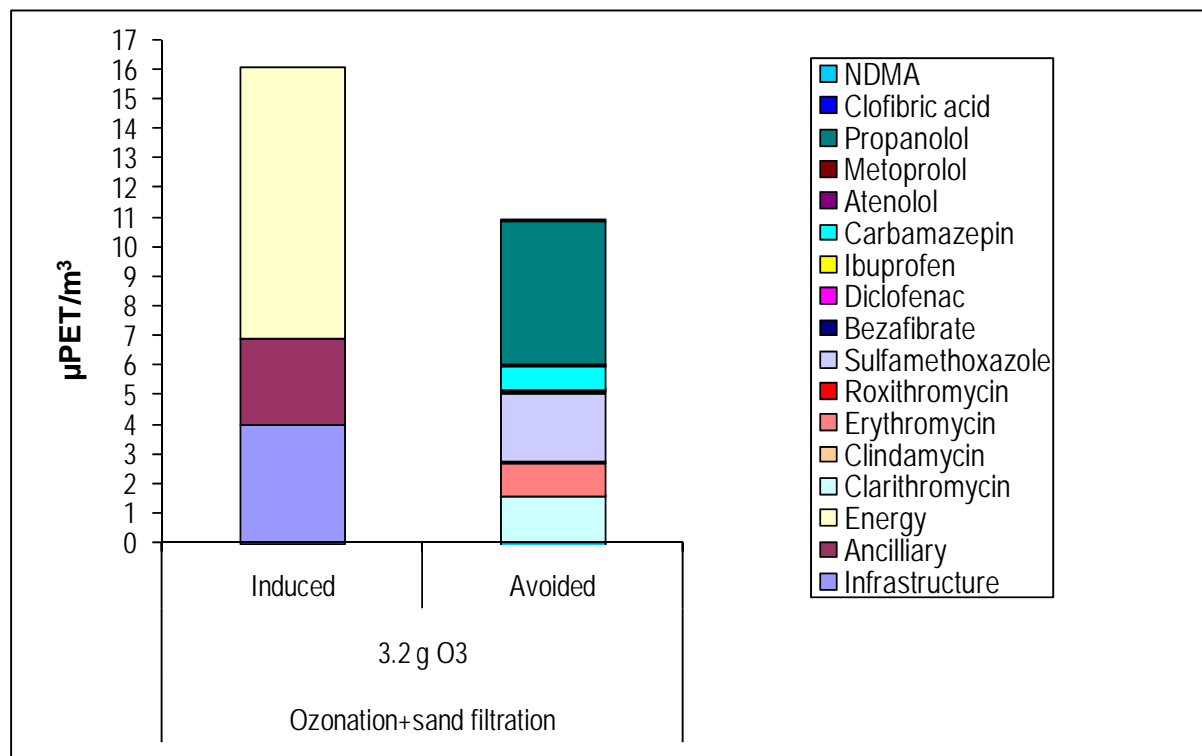


Figure 8.9 Normalized and weighed LCA impact profile showing induced and avoided impacts for ozonating secondary effluent at a dose of 3.2 g O₃/m³ combined with post sand filtration. Based on the functional inventory shown in Table 5.18 and the physical inventory as described in Section 5.1.1 and Section 5.1.2. Weighting factor for all impact categories equals 1

Due to the added infrastructure and energy consumption the impact on the profile of including sand filtration (as compared to ozonation alone) is an increase in the induced part from about 10 μPET/m³ to about 16 μPET/m³, whereas for the avoided part the increase is insignificant, see Figure 8.9 and Figure 8.3.

8.1.5 PAC addition to secondary effluent combined with sand filtration

In order to avoid activated carbon from ending up in the recipient, tests have been run on PAC addition to the secondary effluent followed by sand filtration within NEPTUNE (described in Section 4.1). Only the dose 20 g PAC/m³ has been tested and the resulting LCA impact profile is shown in Figure 8.10.

As was the case for ozonation the induced part is increased by 6 μPET/m³ when combined with sand filtration and the avoided part is only marginally changed.

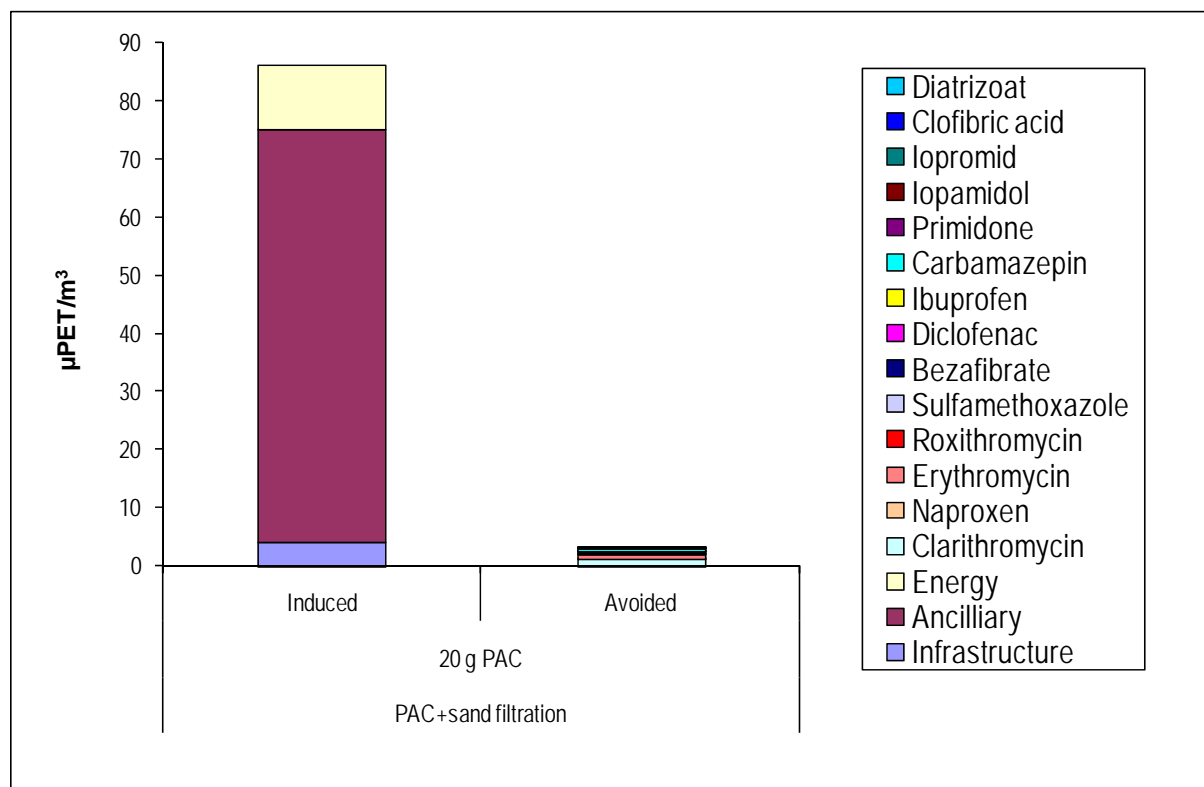


Figure 8.10 Normalized and weighed LCA impact profile showing induced and avoided impacts for PAC addition to secondary effluent at a dose of 20 g PAC/m³ combined with post sand filtration. Based on the functional inventory shown in Table 5.15 and the physical inventory as described in Section 5.1.1 and Section 5.1.3. Weighting factor for all impact categories equals 1

8.1.6 Including more micropollutants or conservative CFs

As shown in Table 5.13 it has only been possible to include a restricted number of the micropollutants that obviously appear in waste water. Maximum 22 organic micropollutants (mostly pharmaceuticals) are included in the assessment of ozonation and sand filtration, and for pulverized activated carbon (PAC) addition the number is maximum 16 for PAC addition to biology and 15 for PAC addition to effluent followed by sand filtration.

As discussed in Deliverable 4.2 (Larsen et al. 2009) we probably only cover about 1/15 of the European tonnage consumption of pharmaceuticals by including the above mentioned 22 micropollutants. In numbers the 22 pharmaceuticals are only a fraction of the approximately 3000 different pharmaceuticals that are used in Europe (Ternes and Joss 2006) supplemented by a huge number of other different substances (fragrances, antioxidants etc.) appearing as e.g. ingredients in personal care products and also ending up in the sewer.

Both ozonation and PAC addition is probably able to reduce the concentration of a major part of these “missing” substances appearing in the secondary effluent. As stated by Ternes and Joss (2006) removal of most pharmaceuticals and personal care products from water may be achieved at a level of 90%-99% for ozonation and is also shown for activated carbon.

Inclusion of more micropollutants will inevitably lead to an increase in the avoided part on the LCA impact profiles for ozonation and PAC addition. As an example we may look at substances with high endocrine disrupting properties, i.e. estrogenicity. Due to lack of useable data on removal rates for especially estradiol and ethynylestradiol when treated by ozonation and PAC addition it has not been possible to include these substances in the impact profiles. However, experiences performed within NEPTUNE and published in Stalter et al. (2010b) have shown higher than 75% removal of in vivo estrogenicity by both ozonation

and PAC addition. These measurements are done as whole effluent toxicity (WET) tests by yeast screening tests for estrogenicity (YES). A preliminary methodology for estimating characterisation factors for this type of measured estrogenicity is developed within NEPTUNE and reported in Deliverable 4.2 (Larsen et al. 2009). In this report the aquatic ecotoxicity characterisation factor ECF (WET_{YES}) for WET as regards estrogenicity measured by YES in the unit estradiol equivalents (EEQ), is estimated to be $2.0E+09$ m³/kg EEQ. This is a 100 times higher than the characterisation factor for one of the most toxic pharmaceuticals, i.e. propranolol (see Table 6.1). If we assume an incoming concentration of 2 ng EEQ/L (based on data from Miege et al. 2009) and a removal rate of 75%, the contribution to the avoided part will be: $1 * (0.75 * 2 \mu\text{g}/\text{m}^3 * 2.0E+09 \text{ m}^3/\text{kg EEQ}) / (352,000 \text{ m}^3/\text{capita}/\text{year}) = 8.5 \mu \text{ PET}/\text{m}^3$. Including this result in the LCA impact profiles will increase the avoided part by almost a factor 2 for ozonation (Figure 8.3 and Figure 8.9) and almost a factor 4 for PAC addition (Figure 8.7 and Figure 8.10). This example illustrates the high impact including more micropollutants may have on the avoided part in the LCA impact profiles on ozonation and PAC addition.

The “standard” characterisation factors (CF) shown in Table 6.1 have been used to calculate the LCA impact profile shown previously. These CFs are all calculated based on the adapted principles described in Deliverable 4.2 (Larsen et al. 2009) for EDIP97 focusing on end-points relevant for the survival of the species population and using the assessment factors described for EDIP97 in Hauschild et al. (1998). However, for seven substances more rigorous CFs are also shown. These factors are based on PNECs from tiered risk assessment (RA) using higher assessment factors in some cases and including end-points for which the relevance regarding population survival is debatable.

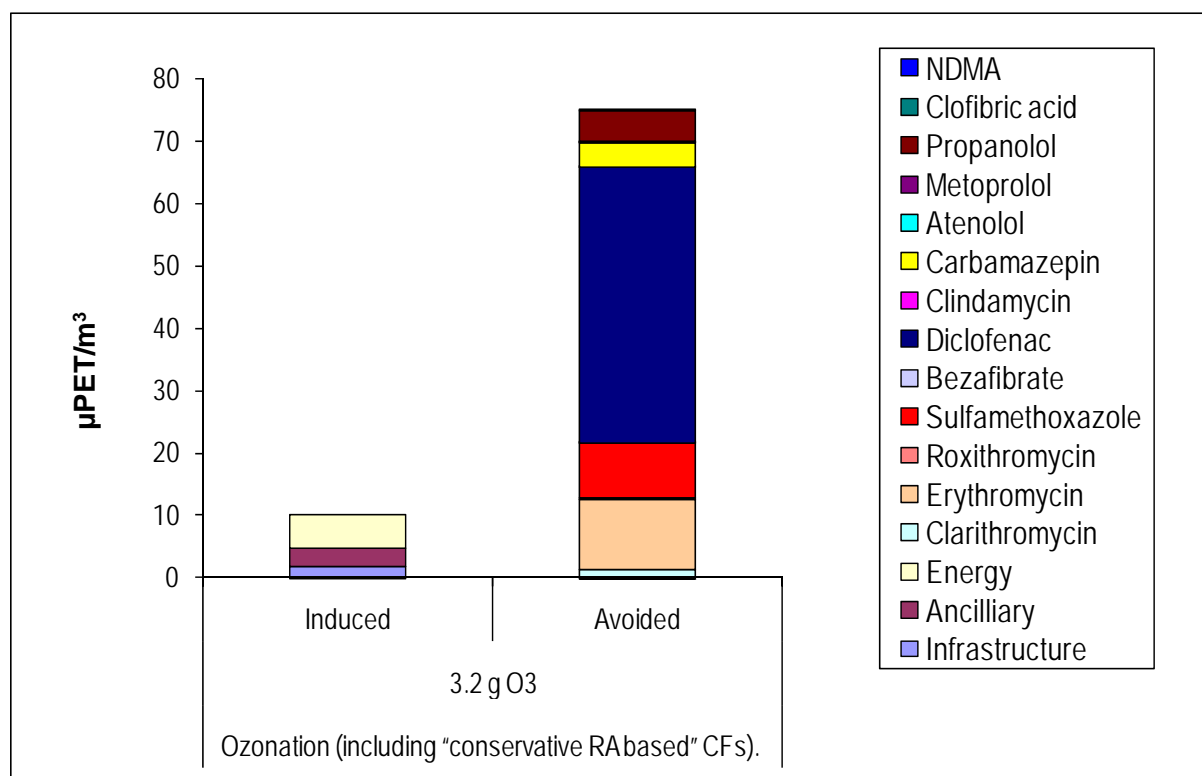


Figure 8.11 Normalized and weighed LCA impact profile showing induced and avoided impacts for ozonating secondary effluent at a dose of $3.2 \text{ g O}_3/\text{m}^3$. Based on the functional inventory shown in Table 5.16 and the physical inventory as described in Section 5.1.1. Weighting factor for all impact categories equals 1. (“conservative RA based” CFs from Table 6.1 included)

As an example we might have a closer look at diclofenac for which there is a factor of 1000 in difference between the “standard” characterisation factor and the “conservative RA

based". The "standard" CF is based on the lowest NOEC value of 1000 µg/L from a reproduction test on Ceriodaphnia dubia (Ferrari et al. 2004). As also chronic values exist for the other two trophic levels (algae and fish) an assessment factor of 10 is used leading to a PNEC value of (1000 µg/L)/10 = 100 µg/L. The "conservative RA based" CF for diclofenac is based on a PNEC value of 0.1 µg/L reported in the German LAWA projects (LAWA project I+II 2004/2006). This PNEC is based on a NOEC value of 1 µg/L from a histopathological investigation of rainbow trout (Oncorhynchus mykiss) exposed to diclofenac (Schwaiger et al 2004; Triebkorn et al. 2004) and an assessment factor of 10 is also used leading to the PNEC value of 0.1 µg/L. In the case of the "standard" CF the end-point reproduction is used which is evidently relevant for the survival of the population. Whether or not histopathological changes/cytopathology in the organs/tissue of a rainbow trout (i.e. renal lesions, necrosis of pillar cells in gills, depletion of glycogen content in liver) are relevant for the survival of the rainbow trout population is debatable. However, discussions via personal communication with Daniel Stalter (Stalter 2008) have lead the conclusion that the observed effects (multiple end-points) might be relevant from the point of view that they probably lead to increased impairment of the health conditions of the fish population and hereby increasing the sensitivity to other environmental stressors.

The effect of using the "conservative RA based" CFs for the LCA impact profiles on ozonation and PAC addition is shown in Figure 8.11 and Figure 8.12, respectively.

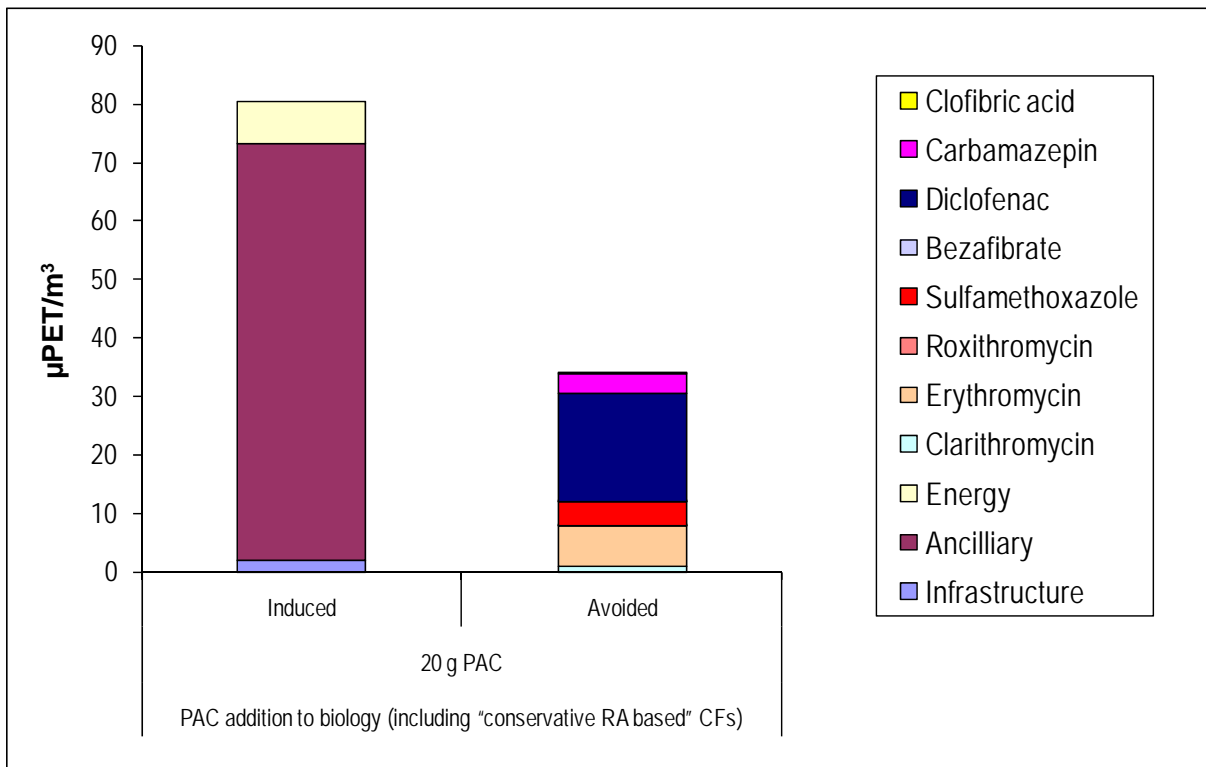


Figure 8.12 Normalized and weighed LCA impact profile showing induced and avoided impacts for PAC addition to biology at a dose of 20 g PAC/m³ regarding secondary effluent waste water. Based on the functional inventory shown in Table 5.14 and the physical inventory as described in Section 5.1.3. Weighting factor for all impact categories equals 1 ("conservative RA based" CFs from Table 6.1 included)

As is evident from both Figure 8.11 and Figure 8.12, especially diclofenac is now contributing significantly to the avoided part but also the contribution from erythromycin, sulfamethoxazole and carbamazepin is increased. Regarding ozonation the avoided part is now about seven times bigger than the induced part.

8.1.7 Including removal of metals and phosphorus by the sand filter

As described in Section 5.5.1 sand filtration has a known effect on suspended matter (TSS) and removes typically around 75% of the TSS content of the incoming water. Combining this effect with the typical content of metals and phosphorus in TSS (see Section 5.1.1 and Table 5.12A, secondary sludge) and including these data in the LCA impact profile for sand filtration gives the following result, see Figure 8.13.

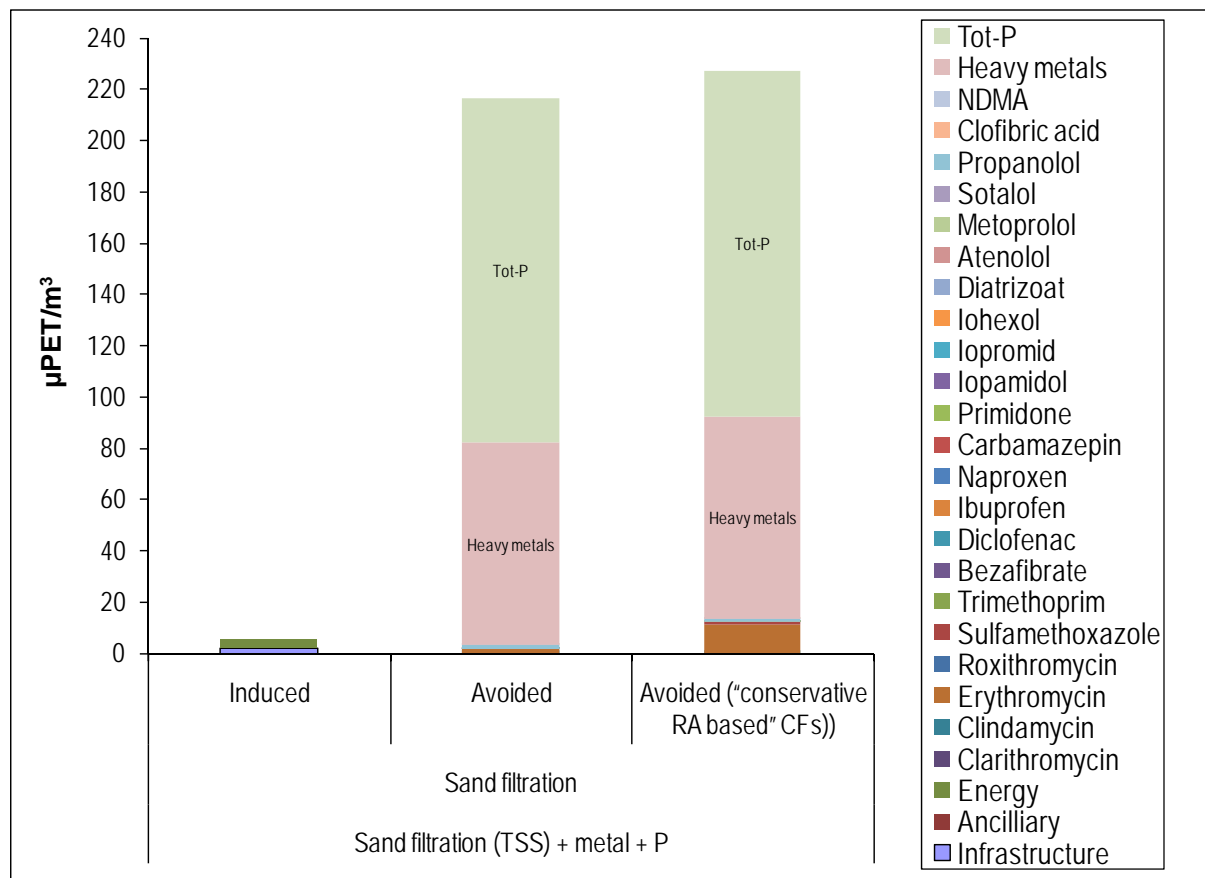


Figure 8.13 Normalized and weighed LCA impact profile showing induced and avoided impacts for sand filtration regarding secondary effluent (post ozonation removal rates used for pharmaceuticals). Removal of TSS included (i.e. metals and P). Based on the functional inventory shown in Table 5.17 and the physical inventory as described in Section 5.1.2. Weighting factor for all impact categories equals 1 (“conservative RA based” CFs from Table 6.1 included)

As may be seen from Figure 8.13 including removal of metals and total-P increase the avoided part substantially (compared with Figure 8.6) and the avoided part is now almost 40 times bigger than the induced part. This change is heavily dominated by the removal of suspended matter (i.e. metals and P) as the share of the avoided part for P is 59% - 62%, for metals is 35% - 36% and for the pharmaceuticals 1% - 6%, depending on whether or not “conservative RA based” CFs are included.

For ozonation and PAC addition combined with post sand filtration the effect of including the removal of metals and phosphorus is shown in Figure 8.14 and Figure 8.15 respectively.

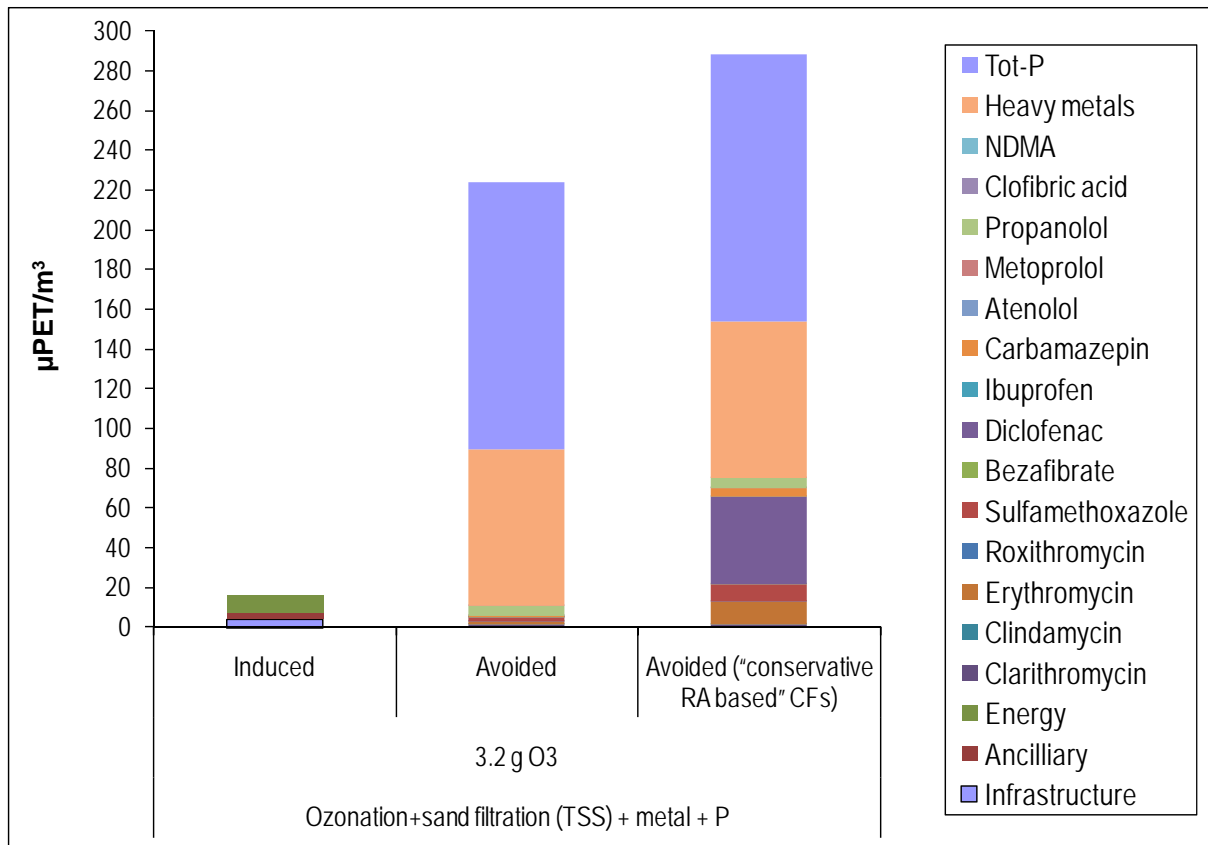


Figure 8.14 Normalized and weighed LCA impact profile showing induced and avoided impacts for ozonating secondary effluent at a dose of 3.2 g O₃/m³ combined with post sand filtration. Removal of TSS included (i.e. metals and P). Based on the functional inventory described in Section 5.5.1, and the physical inventory as described in Section 5.1.1 and Section 5.1.2. Weighting factor for all impact categories equals 1. (“conservative RA based” CFs from Table 6.1 included)

As for sand filtration alone the effect of including removal of metals and phosphorus, in the LCA impact profile for ozonation combined with sand filtration, is substantial and the avoided part is now 14 - 18 times higher than the induced part, depending on whether or not “conservative RA based” CFs are included, see Figure 8.14. The avoided part is, as for sand filtration (Figure 8.13), dominated by removal of tot-P and metals but the share covered by the pharmaceutical removal is increased to 5% - 26%, depending on whether or not “conservative RA based” CFs are included.

Also for PAC addition combined with post sand filtration the effect of including removal of metals and tot-P on the profile is significant, see Figure 8.15. But due to a much higher induced part (than for sand filtration alone or ozonation) the avoided part becomes only about 3 times higher than the induced part. The share of the avoided part covered by the pharmaceutical removal is 1% - 14%, depending on whether or not “conservative RA based” CFs are included.

It should be noted that regarding the high contribution from phosphorus removal P-deficient is assumed in the recipient. This is true for most freshwater ecosystems while coastal areas are typically N-deficient. This issue is further discussed in Deliverable 4.2 (Larsen et al. 2009)

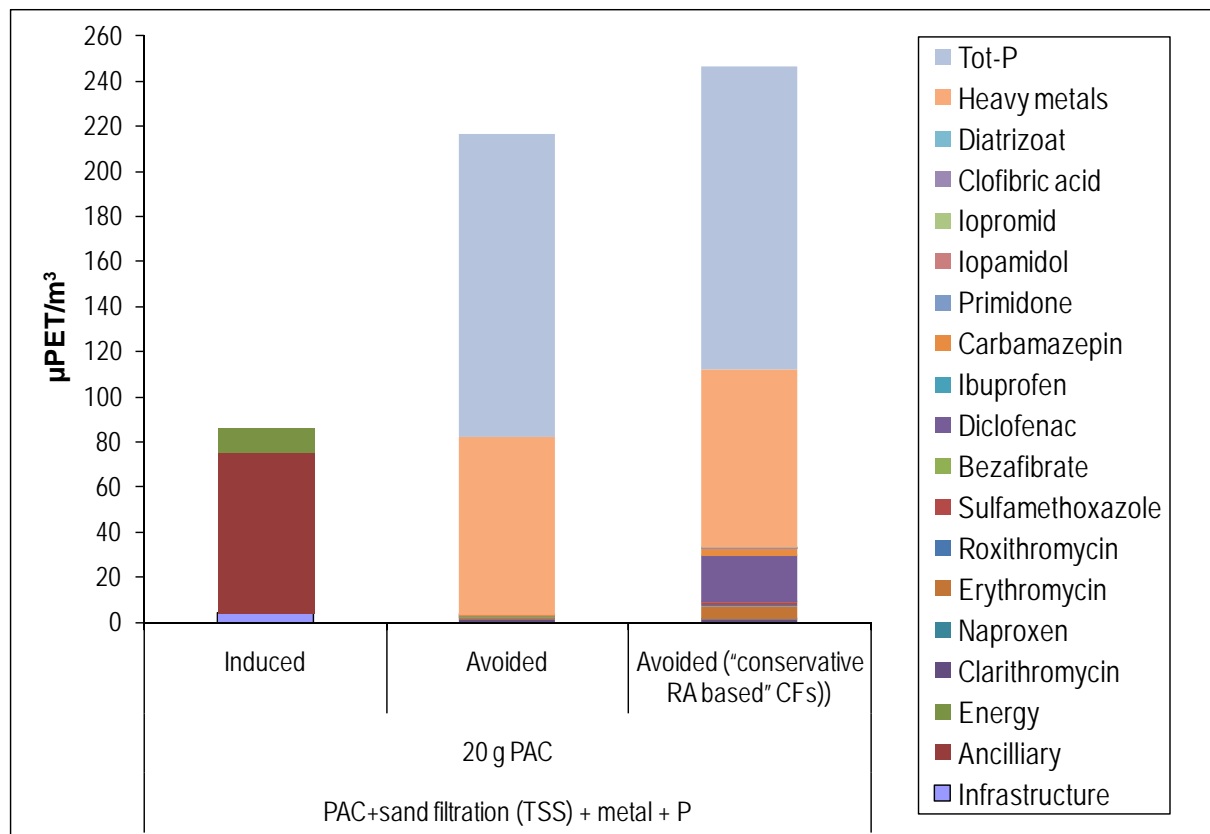


Figure 8.15 Normalized and weighed LCA impact profile showing induced and avoided impacts for PAC addition to biology combined with post sand filtration at a dose of 20 g PAC/m³ regarding secondary effluent. Removal of TSS included (i.e. metals and P). Based on the functional inventory described in Section 5.5.1 and the physical inventory as described in Section 5.1.3 and Section 5.1.2. Weighting factor for all impact categories equals 1 (“conservative RA based” CFs from Table 6.1 included)

8.1.8 Including average electricity production or long-term emissions

Choice of energy scenario for electricity production and long-term emissions from landfills are issues that could have significant effect on the size of the induced impact included in the LCA impact profiles.

As stated in Section 3.3 marginal electricity production is choosing here as default because the environmental sustainability assessment of (the possible introduction of) new waste water treatment technologies is assessed by consequential LCA. This is further discussed in Appendix 15. Even though, there are strong arguments for the statement that average electricity production should only be included in attributional LCA (i.e. identifying hot spots in existing systems), the effect of including an average approach on the NEPTUNE cases is assessed below.

As described in Appendix 15 the difference between producing one kWh by use of the natural gas (the marginal used as default here; CH: electricity, at cogen 500kWe lean burn, allocation exergy) and using a European average (UCTE: electricity, medium voltage, production UCTE, at grid) is a factor below 3. Using this average approach in the LCA impact profile on ozonation, PAC addition and sand filtration will mainly have an effect on the energy part of the induced column and to some degree on the ancillary. For example, the effect on the total induced part for ozonation (3.2 g O₃/m³) will be an increase by a factor of 2.5, for sand filtration a factor of 2.1 and for PAC addition (20 g PAC/m³) a factor 1.2. As may be concluded from looking at e.g. Figure 8.13, Figure 8.14 and Figure 8.15 changes at that level

will not affect the overall picture of the LCA impact profiles. Only if the used marginal Swiss electricity production is substituted by average Polish produced electricity (based primarily on coal) the impact is significant on the overall picture as we are now talking about an increase of a factor 11 on the induced part for ozonation ($3.2 \text{ g O}_3/\text{m}^3$). On the other hand using average Swiss produced electricity the induced part for ozonation ($3.2 \text{ g O}_3/\text{m}^3$) is reduced to less than one third (see Appendix 15 for more details).

As stated in Section 3.3 and Section 7 long-term emissions from landfills are not included in the default scenarios used here. Emissions covering 60,000 years are considered highly uncertain and out of proportion as compared to the other emissions (and impact categories) included, covering maximum a period of 100 years. If however, long-term emissions are included the effect on the induced part for ozonation ($3.2 \text{ g O}_3/\text{m}^3$) is an increase by a factor of 4.1 in the total induced part, for sand filtration a factor of 5.5 and for PAC addition ($20 \text{ g PAC}/\text{m}^3$) a factor 2.3.

Combining both the substitution by average European electricity production and including long-term emissions (“worst case scenarios”) will only change the overall picture for PAC addition combined with sand filtration (Figure 8.15) making induced impacts higher than avoided impacts while for both sand filtration alone (Figure 8.13) and ozonation combined with sand filtration (Figure 8.14) the overall picture will still be a higher avoided part than the induced part.

8.1.9 Environmental sustainability assessment regarding cluster 1

The environmental sustainability of a technology/treatment system is here assessed as to which degree (or how likely it is if comprehensiveness is improved that) the avoided potential environmental impacts are higher than the induced potential environmental impacts. Based on the results presented here it may be concluded that the results indicate that:

- **Ozonation alone** (e.g. $3.2 \text{ g O}_3/\text{m}^3$) used for removal of organic micropollutants is probably environmental sustainable but problems with toxic metabolites have to be addressed from a risk assessment point of view
- **Sand filtration alone** used for removal of organic micropollutants is probably also environmental sustainable but the LCA impact profile is far from as good as the one for ozonation mainly because of lower removal rates. If removal of metals and phosphorus is included the LCA impact profile is improved substantially
- **PAC addition to biology alone** (e.g. $20 \text{ g PAC}/\text{m}^3$) used for removal of organic micropollutants may be environmental sustainable but this needs clarification by including a lot more micropollutants and better updated inventory data on PAC production. Anyway, based on existing knowledge the LCA impact profile is not at all as good as the one for ozonation alone and the problem with PAC ending up in the recipient have to be addressed from a risk assessment point of view
- **Ozonation combined with post sand filtration** used for removal of micropollutants (including metals) and phosphorus is most probably environmental sustainable and has the best LCA impact profile of the treatment systems assessed in this cluster
- **PAC addition to effluent combined with post sand filtration** used for removal of micropollutants (including metals) and phosphorus is probably environmental sustainable. However, based on existing knowledge the LCA impact profile is not as good as the one for ozonation combined with sand filtration.

Of the waste water treatment systems for micropollutant removal assessed here ozonation combined with post sand filtrations seems to be the most optimal solutions when looking at environmental sustainability. A further advantage, which is has not been possible to include quantitatively here but see Deliverable 4.2 (Larsen et al. 2009) for preliminary methodology, is the pathogen removal observed for ozonation (BAFU/Eawag/AWEL/BMG/Hunziker 2009).

Recently compiled results from the Micropoll-project in Switzerland (Zwickenpflug 2010a) on PAC addition to effluent ($10 - 15 \text{ g/m}^3$) combined with iron precipitation ($4 - 6 \text{ g Fe/m}^3$) and recirculation of PAC to the biological step (denitrification/nitrification) shows the achievement of removal rates of pharmaceuticals at the same level as achieved by ozonation ($3.2 \text{ g O}_3/\text{m}^3$) – at least for 15 g PAC/m^3). These results point in the direction of improvement of the LCA impact profile for PAC addition (due to better removal rates than observed within NEPTUNE) but the achievement by a reduction of 5 g PAC/m^3 (as compared to 20 g PAC/m^3) is more than neutralized by the added consumption of 6 g Fe/m^3 (contributing with about $50 \mu\text{PET/m}^3$).

It may be argued that in order to have a significant difference between avoided and induced impacts at least a factor of $10 - 100$ in difference should be achieved. This is due to the relatively high uncertainty on the inventory data and the characterisation factors – especially on ecotoxicity and human toxicity. This level is only reached here for sand filtration including removal of metals and P (Figure 8.13), and for ozonation combined with sand filtration and also including removal of metals and P (Figure 8.14). Including especially more micropollutants (but also updated data on PAC production and more) may change this picture regarding PAC addition combined with sand filtration.

It should be stressed that these conclusions are based on the assumptions, scoping and constraints of this study in which average values (removal rates, incoming concentrations) observed within NEPTUNE are used and European data (EcoInvent) for the physical inventory is included. The conclusions based on this generic approach are off course sensitive to variation in the included data. If for example specific (i.e. site-specific) data on the incoming concentrations are used instead of average values, higher micropollutant concentration will generally lead to an increase in the avoided part of the LCA impact profile if the removal rates are kept unchanged. Besides the incoming concentration of phosphorus and metals (as part of TSS when sand filtration is included) the resulting LCA impact profile is especially sensitive to the concentrations of diclofenac, erythromycin, sulfamethoxazole, propranolol and carbamazepin.

With the present high focus on global warming (Carbon Footprint) it may be appropriate to comment on the results as regards weighting factors. In LCAs the global warming potential is typically related to energy production (fossil fuel based) and in NEPTUNE therefore almost exclusively related to the induced impacts as evident from the results presented here for cluster 1, see for example Figure 8.9. It may therefore be of interest to look at the size of the weighting factor needed, to achieve break-even between avoided and induced impacts, when all other impact categories than global warming is weighted by a factor 1 (the default in NEPTUNE). Looking at ozonation combined with sand filtration ($3.2 \text{ g O}_3/\text{m}^3$, 22 micropollutants + metals+P), the weighting factor to put on global warming in order to reach break-even is 30-40, depending on whether or not conservative PNEC-based CFs are included. For PAC addition combined with sand filtration (SF) the corresponding weighting factor is 5-6. So, based on the NEPTUNE project results, you have to weight global warming by a factor of 5-6 higher (for PAC + SF), or by a factor of 30-40 higher (for ozonation + SF) than all other included impact categories, in order to reach a level where the induced impacts becomes higher than the avoided. Of the total induced impacts included here global warming accounts for 45% (ozonation + SF) and 31% (PAC + SF). The rest is distributed among other impact categories and for “ozonation + SF” dominated by “Human toxicity water” (16%), “Human toxicity soil” (10%), “Acidification” (8%), “Ecotoxicity water (8%), “Nutrient enrichment” (7%) and “Photochemical oxidation” (5%). Regarding “PAC+SF” the rest is dominated by “Human toxicity soil” (18%), “Acidification” (17%), “Human toxicity water” (15%), “Ecotoxicity water (10%) and “Nutrient enrichment” (6%). As regards especially the impact categories on acidification and nutrient enrichment (and to a certain degree the one on photochemical oxidation) they are all closely related to fossil fuel consumption. The others are in this case only partly related.

8.2 Cluster 2: WWTs for nutrient removal

Only the results from the LCIA of conventional WWT (reference) and anammox will be presented here. ICA is addressed in Deliverable 1.2.

8.2.1 Conventional WWT combined with sludge incineration

The impact profile achieved if sewage water (i.e. primary effluent) is emitted directly to the recipient is shown in Figure 8.16, based on in total 45 substances including pharmaceuticals, metals and other inorganics, and nutrients. The impact profile is, not surprisingly, dominated by impact potentials related to ecotoxicity (micropollutants) and nutrient enrichment (N and P). More than two thirds of the total potential impact is related to metals (iron, copper, mercury, aluminum, zinc) whereas tot-N and tot-P contribute with below one third. Pharmaceuticals only contributes with below 0.3%, i.e. about 20 $\mu\text{PET}/\text{m}^3$ (“standard” CFs from Table 6.1 used)

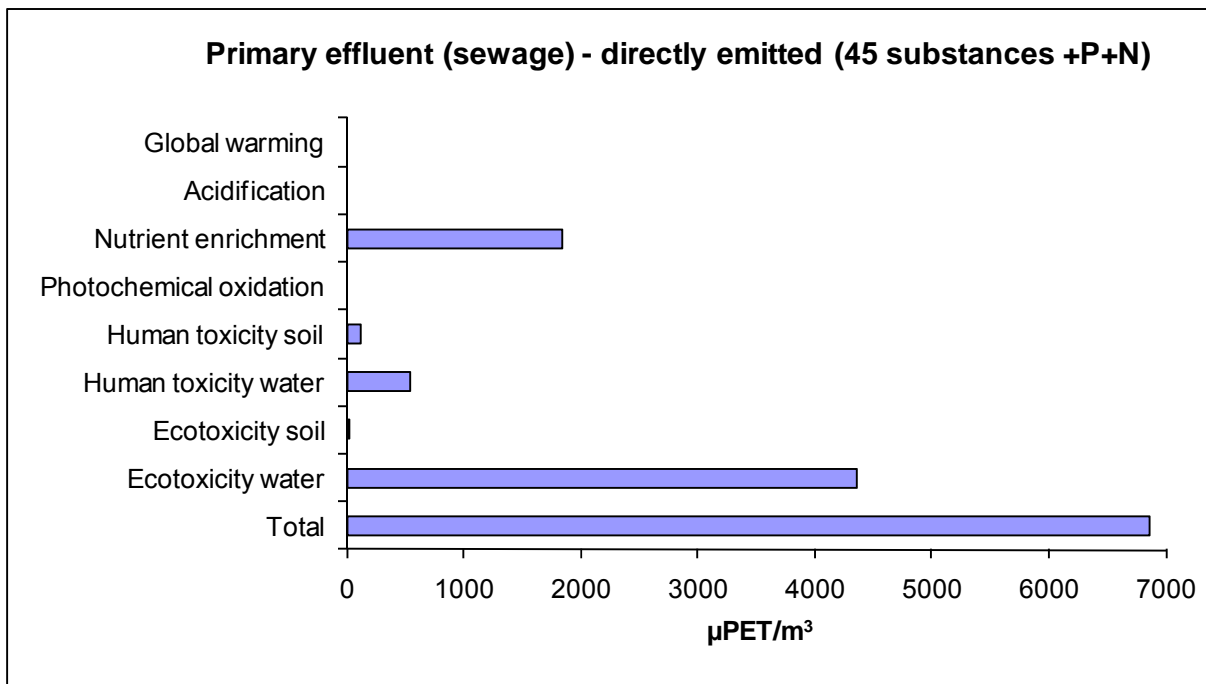


Figure 8.16 Normalized and weighed LCA impact profile for the direct emission of primary effluent (sewage water) to recipient. Based on the inlet concentrations as referred to in Section 5.5.2 – of which some are shown in Table 5.11 - in total 45 substances including pharmaceuticals, metals and other inorganics, and nutrients. Weighting factor for all impact categories equals 1

When treating the sewage water in the conventional treatment model defined in Section 5.2.1 the impact profile (including the induced part) for the treated water (secondary effluent) becomes the one shown in Figure 8.17. It should be noted that this profile do not include sludge disposal (incineration) only the water treatment and sludge digestion. As compared to Figure 8.16 the potential impact from nutrient enrichment and ecotoxicity is substantially reduced. Global warming, acidification and photochemical oxidation are now appearing. Especially for global warming potential (gwp) the contributors are gwp-gases emitted during the treatment (CO_2 , N_2O and CH_4) with a share of about 48%, but also infrastructure (about 32%) and electricity consumption (about 14%) contributes significantly.

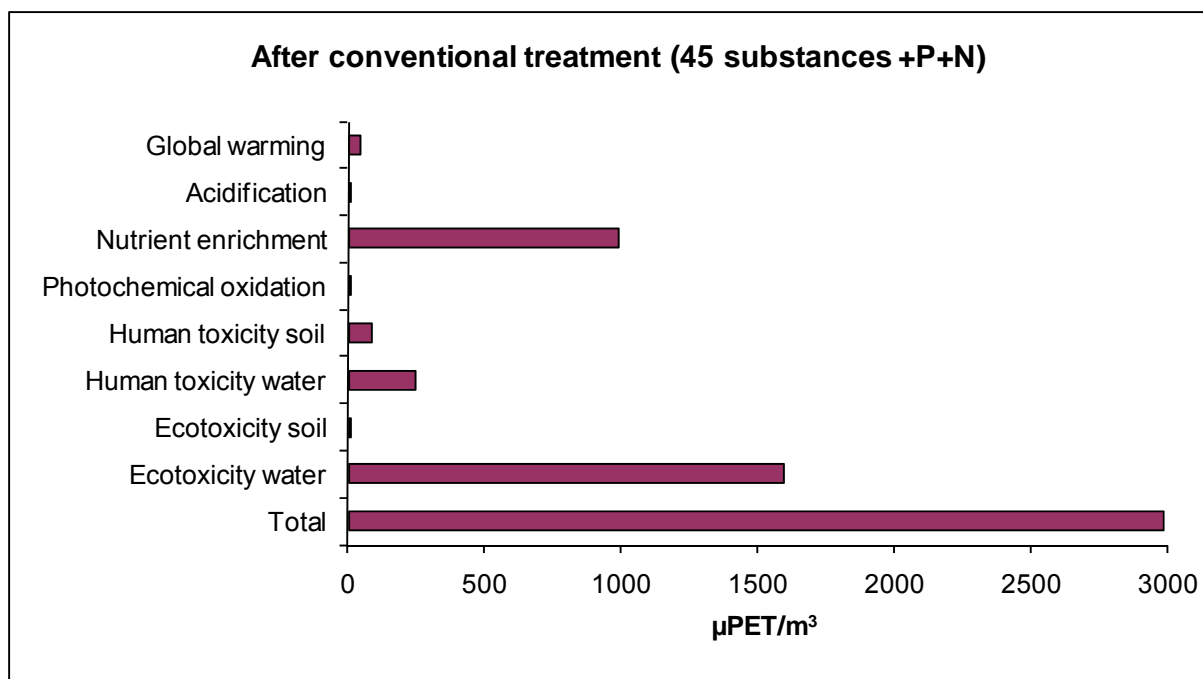


Figure 8.17 Normalized and weighted LCA impact profile for secondary effluent (sewage water treated by conventional treatment) including the induced impact from building, running and disposing the treatment plant. Sludge disposal (incineration) not included. Based on the physical inventory described in Section 4.2 and Section 5.2.1, and the inlet concentrations and removal rates as referred to in Section 5.5.2 – of which some are shown in Table 5.11 - in total 45 substances including pharmaceuticals, metals and other inorganics, and nutrients. Weighting factor for all impact categories equals 1

If the sludge disposal by municipal waste incineration is included and the conventional treatment impact profile is expressed in terms of avoided as compared to induced impacts the profile shown in Figure 8.18 appears.

It is evident from Figure 8.18 that the avoided impacts are substantially higher than the induced part, actually a factor of about 18 for the total. The reductions are, not surprisingly, especially high for nutrient enrichment (factor 106) and ecotoxicity in water (factor 97). So, the aim of conventional treatment regarding removal of nutrients and micropollutants are reflected in these results. The reductions regarding the human toxicity related impact categories are modest, i.e. a factor 4 for “human toxicity water” and a factor 2 for “human toxicity soil”. So, even though removal of metals (especially mercury and lead) by the waste water treatment contribute substantially to the avoided part (383 μPET/m³) of human toxicity in water, emissions of especially mercury related to the production of iron chloride (ancillary for P- precipitation) and the WWTP infrastructure, and emissions related to the incineration of sludge (air emissions and emissions from land filled ashes) contribute significantly to the induced part. Looking carefully at Figure 8.18 it may be realized that besides expected induced impacts in the energy-related impact categories, i.e. global warming, acidification and photochemical oxidation, also negative avoided impacts are shown (insignificant for photochemical oxidation). These negative avoided impacts appears due to the LCIA modelling, considering changes related directly to the waste water (composition) as avoided impacts and therefore include the gwp-gases emitted during the treatment (CO₂, N₂O and CH₄) as avoided - but negative.

Incineration of sludge contributes with about 10% of the total induced impacts shown in Figure 8.18. This contribution is dominated by potential impacts from air and water emissions by the incinerator and treatment/disposal of ash (e.g. metals and NO_x), and contributions related to the infrastructure.

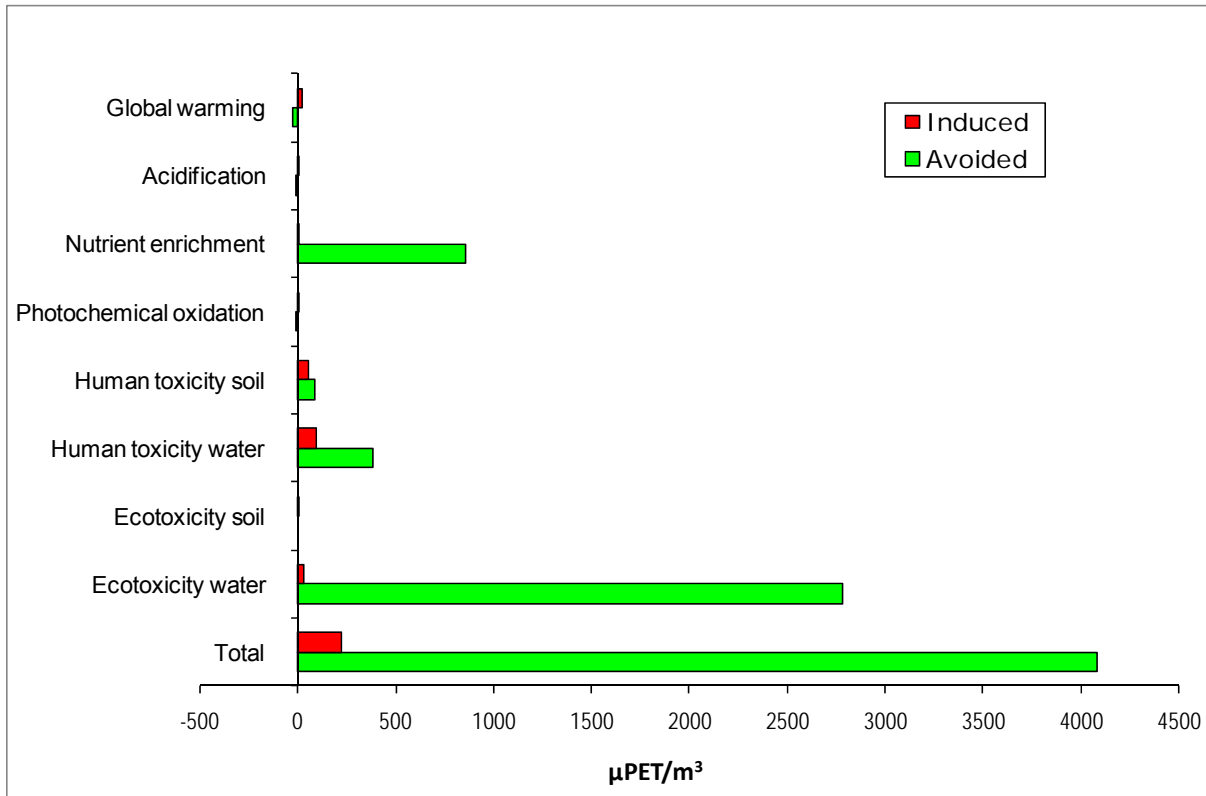


Figure 8.18 Normalized and weighted LCA impact profile showing induced and avoided impacts for conventional treatment of waste water (as defined in Section 4.2 and Section 5.2.1) including sludge incineration. Based on inlet concentrations and removal rates as referred to in Section 5.5.2 – of which some are shown in Table 5.11 - in total 45 substances including pharmaceuticals, metals and other inorganics, and nutrients. Weighting factor for all impact categories equals 1

8.2.2 Autotrophic anaerobic ammonium oxidation (anammox)

The LCA impact profile for anammox is shown in Figure 8.19. It is based on the mass balance described in Section 5.5.2.

As illustrated in Figure 8.19 introducing anammox apparently leads to reductions in all impact categories except nutrient enrichment. This is primarily due to the increased N₂O emissions included in this scenario. In total the avoided impacts are about a factor 2 higher than the induced.

According to Siegrist (2009b) recent research shows a balance a lot more in favor of treatment systems with anammox than treatment systems without. This “new” mass balance is shown in Table 8.1.

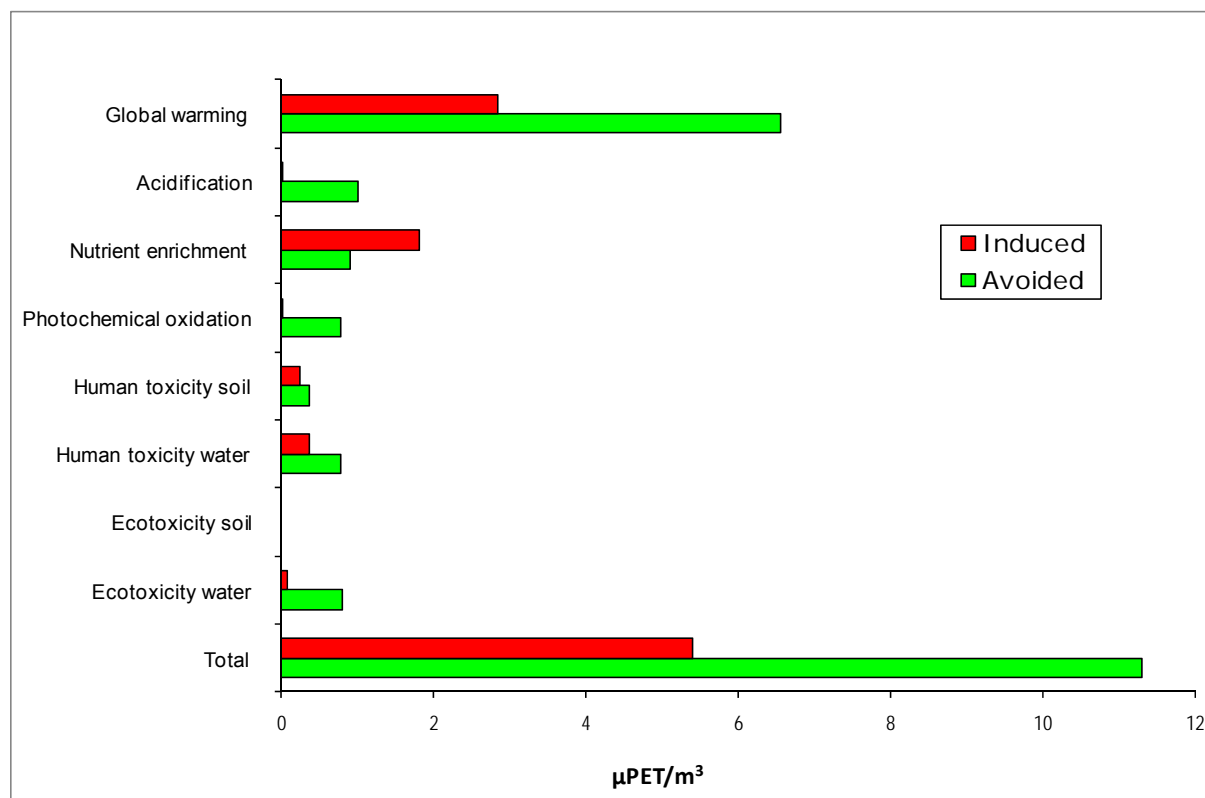


Figure 8.19 Normalized and weighted LCA impact profile showing induced and avoided impacts due to the introduction of anammox at a generic conventional waste water treatment plant. Emissions of nutrients (Tot-N) and COD are kept constant leading to changes in energy consumption, see Appendix 8. Weighting factor for all impact categories equals 1

Table 8.1 Recent data on mass and energy balance for treatment with or without anammox. Emission of nutrients and COD identical in the two cases (as shown in Appendix 8 for the dataset used in Figure 8.19)

		Electricity consumption net (kWh/m ³)	N ₂ O emission (mg N ₂ O/m ³)
Conventional alone	Nitrification/denitrification	0.024	172
	Biogas production	0	0
	Total consumption/emission	0.024	172
Anammox included	Nitrificat./denitrificat. + Anammox	0.011	43
	Biogas production	-0.023	0
	Total	-0.012	43
Difference	Anammox incl. - Conv. alone	-0.036	-129

(Siegrist 2009b)

Using the figures in Table 8.1 for calculation of an LCA impact profile gives the result shown in Figure 8.20. As expected the profile is now much more in favor of including anammox, as the avoided part becomes a factor of about 15 higher than the induced part.

Other research on anammox and N₂O emissions (Kampschreur et al. 2008) indicate that introducing anammox at conventional treatment plants does not change the total N₂O emissions as “the N₂O emissions in the reject water treatment seem to be in the same range as for the main stream of activated sludge processes” (Kampschreur et al. 2008). However, it is also stated that preliminary results indicate that the N₂O emissions from a one reactor anammox system (as used in Figure 8.20) is lower than in a two reactor system used in the

study by Kampschreur et al (2008). This statement point in the direction of saving N₂O emission when introducing (one reactor) anammox as reflected in Figure 8.20.

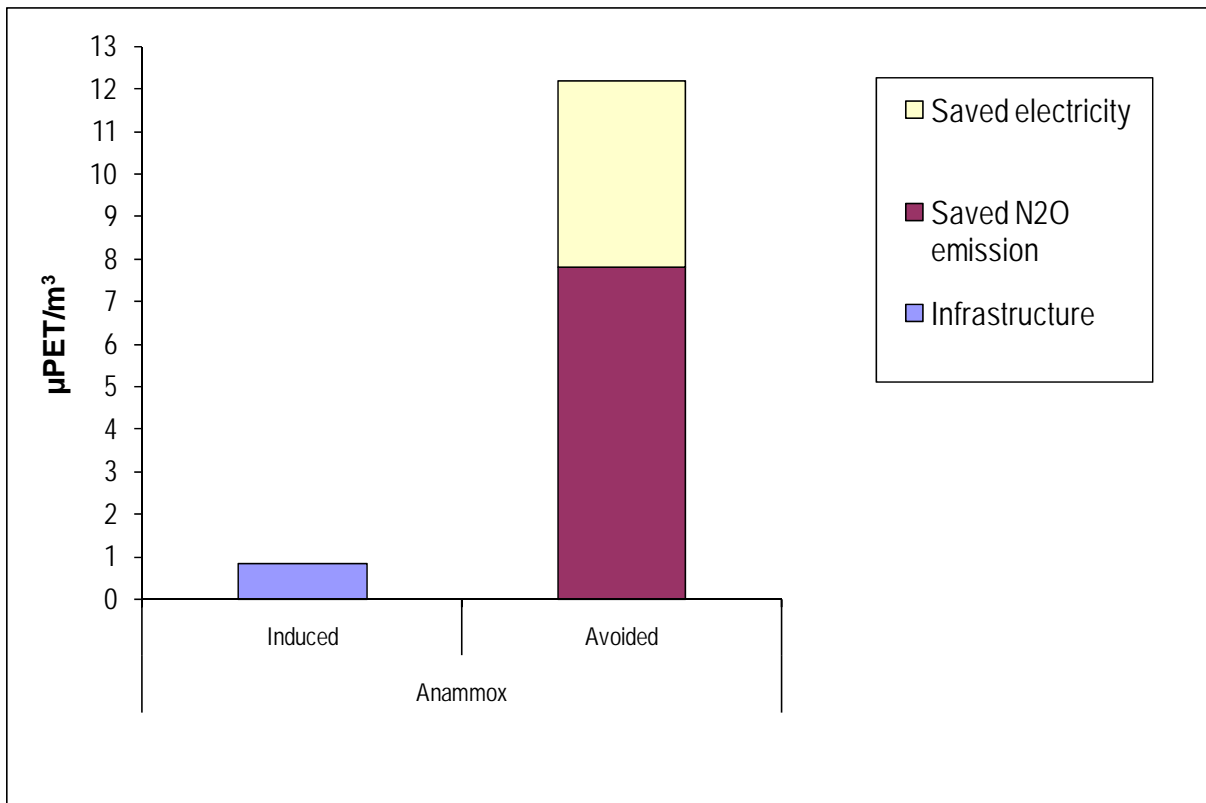


Figure 8.20 Normalized and weighted LCA impact profile showing induced and avoided impacts due to the introduction of anammox at a conventional waste water treatment plant according to the data in Table 8.1. Weighting factor for all impact categories equals 1

On average an N₂O-N emission of 2.3% of N-total load was found in the Kampschreur study (two reactor anammox system). In the study by Siegrist 2% of N-total load is used for conventional and 0.5% of N-total load for conventional combined with anammox. These figures are quite high as compared to the fraction used in the “conventional treatment” case described in Section 8.2.1 based on data from Doka (2007b) where 0.037% of N-total is used. However, if the value from the Kampschreur study is used in the LCA impact profile for the conventional treatment model shown in Figure 8.18 (i.e. 2.3% instead of 0.037%) the negative avoided impact related to global warming will only increase by a factor of about five. This is due to the fact that only about 6% of the negative avoided impact for global warming shown in Figure 8.18 is based on denitrification-related N₂O emission (the rest is mainly related to CO₂ and CH₄ emissions).

8.2.3 Environmental sustainability assessment regarding cluster 2

The environmental sustainability of a technology/treatment system is here assessed in the same way as for cluster 1. Based on the results presented here it may be concluded that the results indicate that:

- Conventional treatment of waste water as defined here is most probably environmentally sustainable
- Introducing anammox in conventional treatment is probably environmentally sustainable but the result is very sensitive to the N₂O emission balance

As metals emissions play a very important role in the impact profile on conventional treatment (but also on sand filtration alone and combined with ozonation or PAC, cluster 1) and only factors for three metals have been updated during the NEPTUNE work (see Table 6.1), there is a need for better characterisation factors on metal emissions in order to confirm their high importance. For at least some of the included metals (e.g. iron) their potential impact may be overestimated and distinguishing between different species and bioavailable fractions are needed. This is not a special problem for waste water but a general problem within LCIA and is addressed in the recently started EU projects LC-Impact (terrestrial ecotoxicity) and PROSUITE (aquatic ecotoxicity).

It may be argued that including both total phosphorus and total nitrogen at the same time in the potential impacts related to nutrient enrichment doesn't make sense as the recipient is either N-deficient or P-deficient. However, as discussed in Deliverable 4.2 (Larsen et al. 2009) the deficient type may change due to seasons and nutrients emitted to typical P-deficient fresh water may sooner or later reach N-deficient coastal areas. This issue is also in focus regarding general LCIA and is addressed in the recently started EU projects LC-Impact where new models on both a global and a site-dependent scale are going to be developed.

Despite the acknowledge general short-comings regarding modelling potential impacts of metals and nutrients, it is assessed that based on existing knowledge conventional treatment of waste water (as defined here, see Figure 4.4) most probably is environmentally sustainable, i.e. total avoided impacts higher than total induced impacts. That only a fraction of total number of micropollutants exposed to treatment is included here further strengthens this conclusion.

8.3 Cluster 3: Sludge inertization as disposal of WW sludge

Sludge inertization aims to reduce the volume of sludge before final disposal as well as the degradation of organic compounds still present in the sludge.

As the reference scenario here is on-site incineration, looking at the environmental sustainability of the different inertization technologies on their own doesn't make sense – only in comparison with incineration. The questions addressed here are therefore:

- Do the sludge inertization methods investigated (WO, pyrolysis, gasification) decrease the potential environmental impact as compared to on-site incineration of sludge as final disposal?
- Which of the investigated inertization methods creates the lowest potential environmental impact?

8.3.1 On-site sludge incineration

The LCA impact profile for on-site incineration of sludge is shown in Figure 8.21. As evident from the figure the potential environmental impacts from on-site incineration is dominated by emissions related to the incinerator. However, also treatment of the liquid residue at the WWTP is contributing significantly. Especially mercury emissions to air and water are dominating giving rise to potential impacts on humans via water and soil.

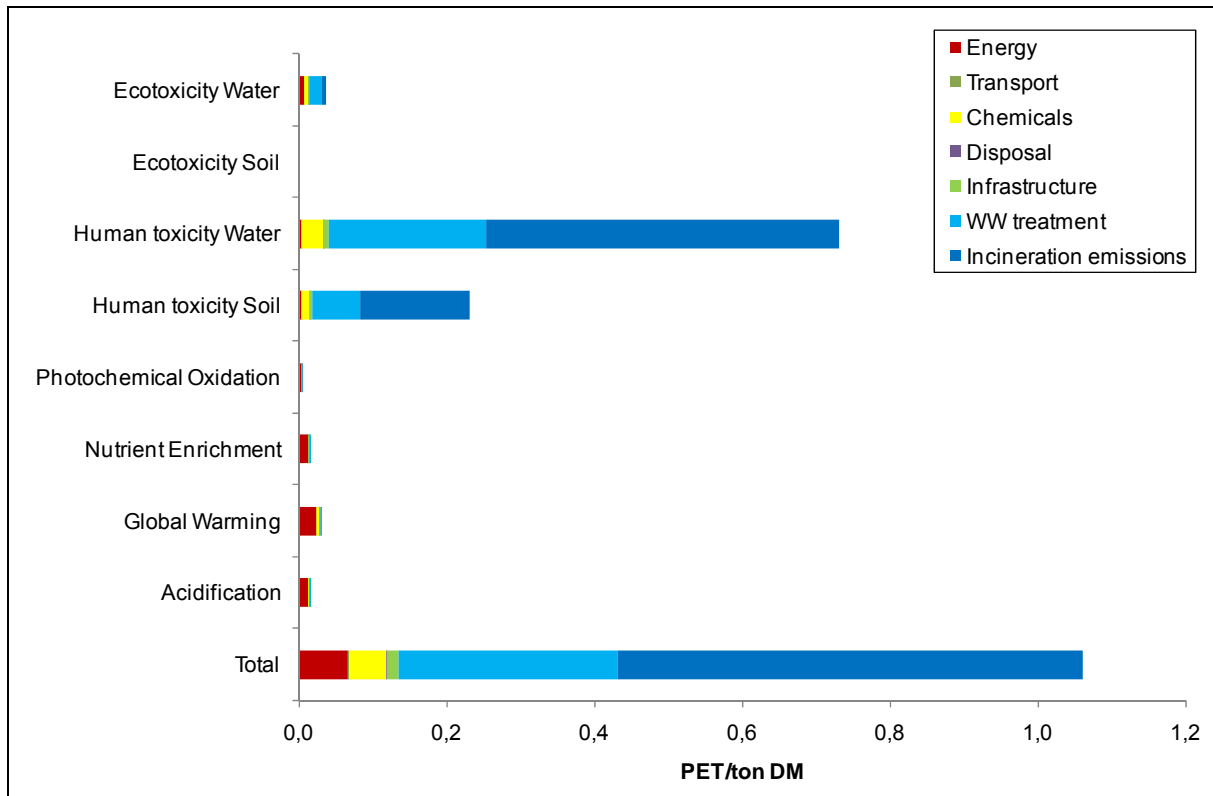


Figure 8.21 Normalized and weighted LCA impact profile showing (induced) impacts due to on-site incineration of 1 ton dry matter (DM) sludge as defined in Section 5.3.1 and Section 5.5.3. Weighting factor for all impact categories equals 1

8.3.2 Wet oxidation (WO) of sludge

The LCA impact profile for WO of sludge is shown in Figure 8.22.

As evident from Figure 8.22 the potential environmental impacts from WO is lower than for on-site incineration and dominated by emissions related to the oxygen consumption (part of “Energy” accounting for more than 50% of the total potential impact). Anyway, emissions related to the process itself (“WO emissions) are also contributing significantly and as for on-site incineration air emissions of mercury is dominating.

The potential impact from the metal content of the liquid residue going back to the WWTP is not included in the GaBi modeling. Estimations based on the modeling of gasification, where potential impact from metals in liquid residue given back to the WWTP is included, have however been done. The result shows that the contribution is about 0.008 PET/ton DM, only increasing the total impact in Figure 8.22 with about 4%.

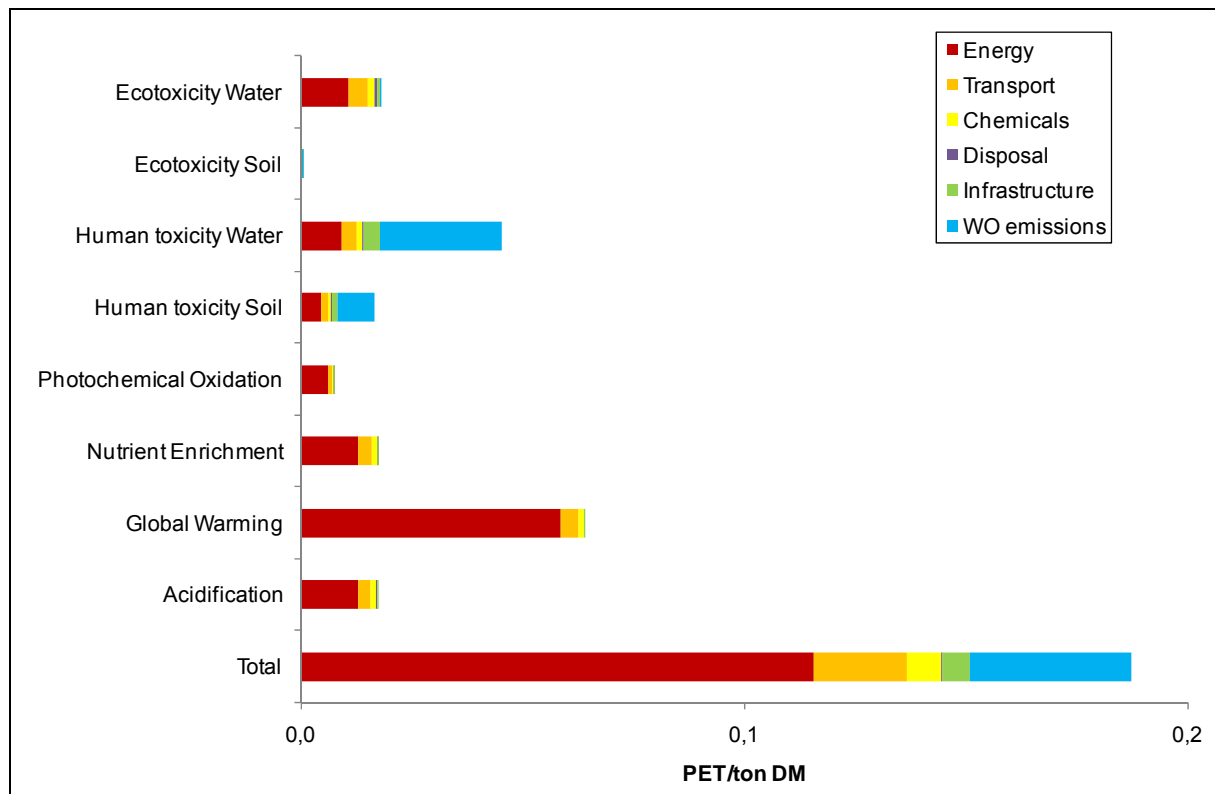


Figure 8.22 Normalized and weighted LCA impact profile showing (induced) impacts due to wet oxidation of 1 ton dry matter (DM) sludge as defined in Section 5.3.2 and Section 5.5.3. Weighting factor for all impact categories equals 1

8.3.3 High temperature pyrolysis (HTP, pyrolysis) of sludge

The LCA impact profile for HTP including sludge drying by heating is shown in Figure 8.23.

The net potential impact balance amounts to 0.027 PET/ton DM and is therefore lower than for on-site incineration (1.06 PET/ton DM). The main contributors are related to air emissions (dominated by Hg) from the burning of the syngas and emissions related to the production of chemicals used for off gas cleaning. The surplus produced electricity and heat contribute “negatively” in the balance, i.e. by substituting fossil fuel based energy, improving the net impact profile of the process. Please be aware that the energy surplus production is based on a theoretical maximum – but even removing this part fully will only increase the total potential impact to 0.21 PET/tonDM.

If the sludge is dried by solar heating instead the impact profile shown in Figure 8.24 results. The net potential impact balance now amounts to only 0.0035 PET/tandem even though more electricity in total is used when including solar drying. This is because all heat is now used externally substituting fossil fuel based heat.

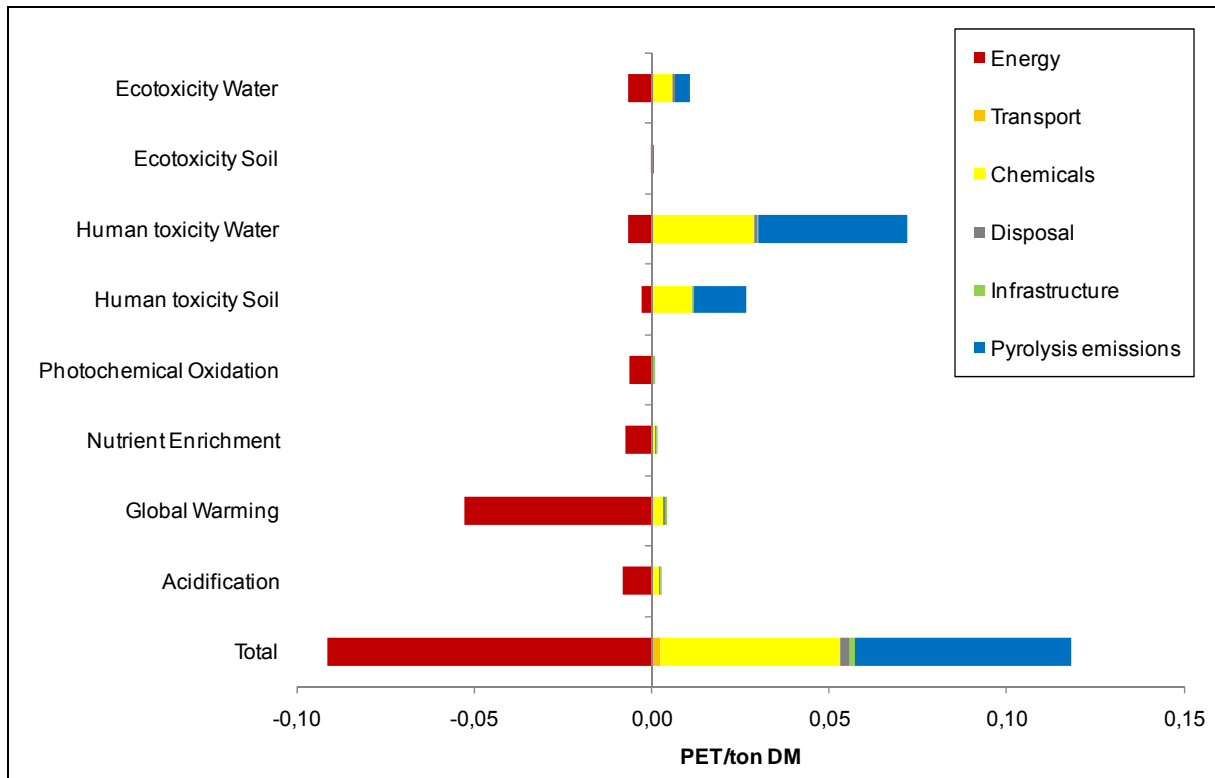


Figure 8.23 Normalized and weighted LCA impact profile showing (induced) impacts due to HTP with heat drying of 1 ton dry matter (DM) sludge as defined in Section 5.3.3 and Section 5.5.3. Weighting factor for all impact categories equals 1

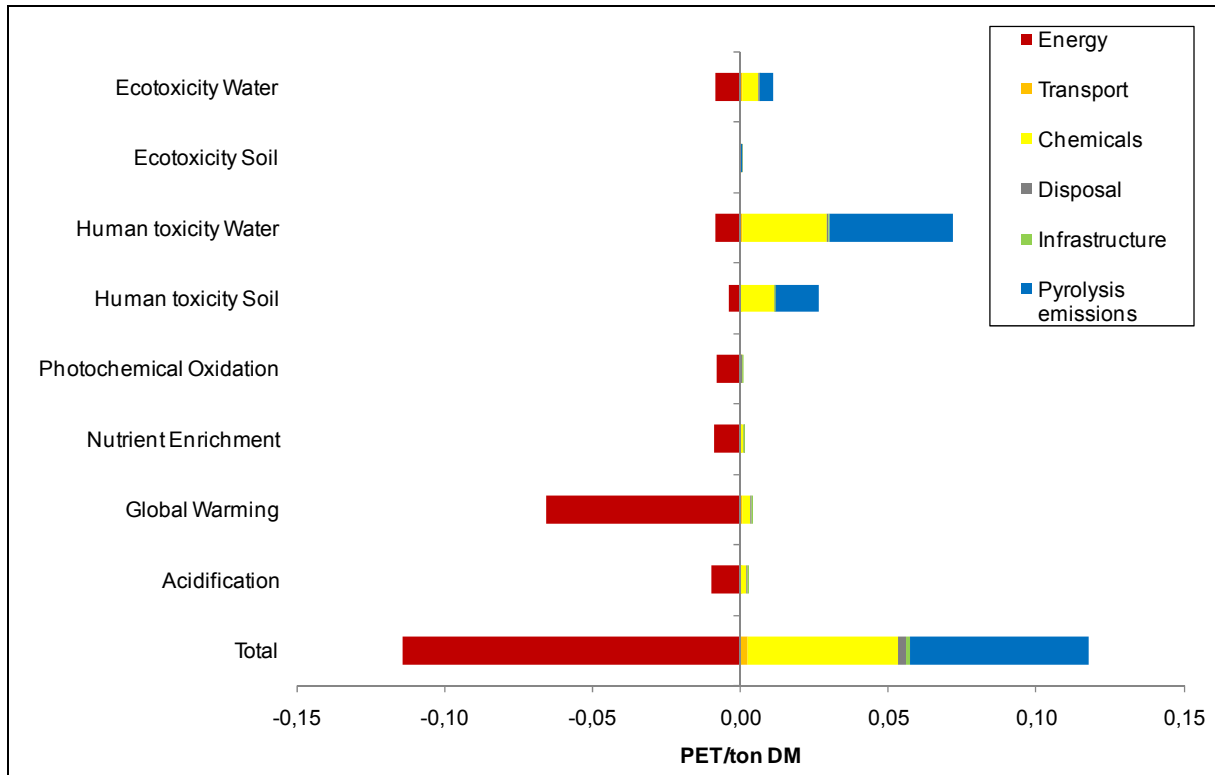


Figure 8.24 Normalized and weighted LCA impact profile showing (induced) impacts due to HTP with solar drying of 1 ton dry matter (DM) sludge as defined in Section 5.3.3 and Section 5.5.3. Weighting factor for all impact categories equals 1

8.3.4 Middle temperature pyrolysis (gasification) of sludge

The LCA impact profile for gasification including sludge drying by heating is shown in Figure 8.25.

The net potential impact balance is negative and amounts to -0.0012 PET/tonDM and is low as compared to on-site incineration (1.06 PET/ton DM). The main contributors are related to the infrastructure but also emissions (dominated by Hg) from the condensate returned to the WWTP but also, the air emissions from the gas utilisation unit and the disposals to landfills are contributing significantly. The surplus produced electricity contributes “negatively”, i.e. by substituting fossil fuel based energy, improving the net impact profile of the process. Anyway, the fuel oil consumption (not shown separately in Figure 8.25) contributes positively reducing the size of the negative energy balance.

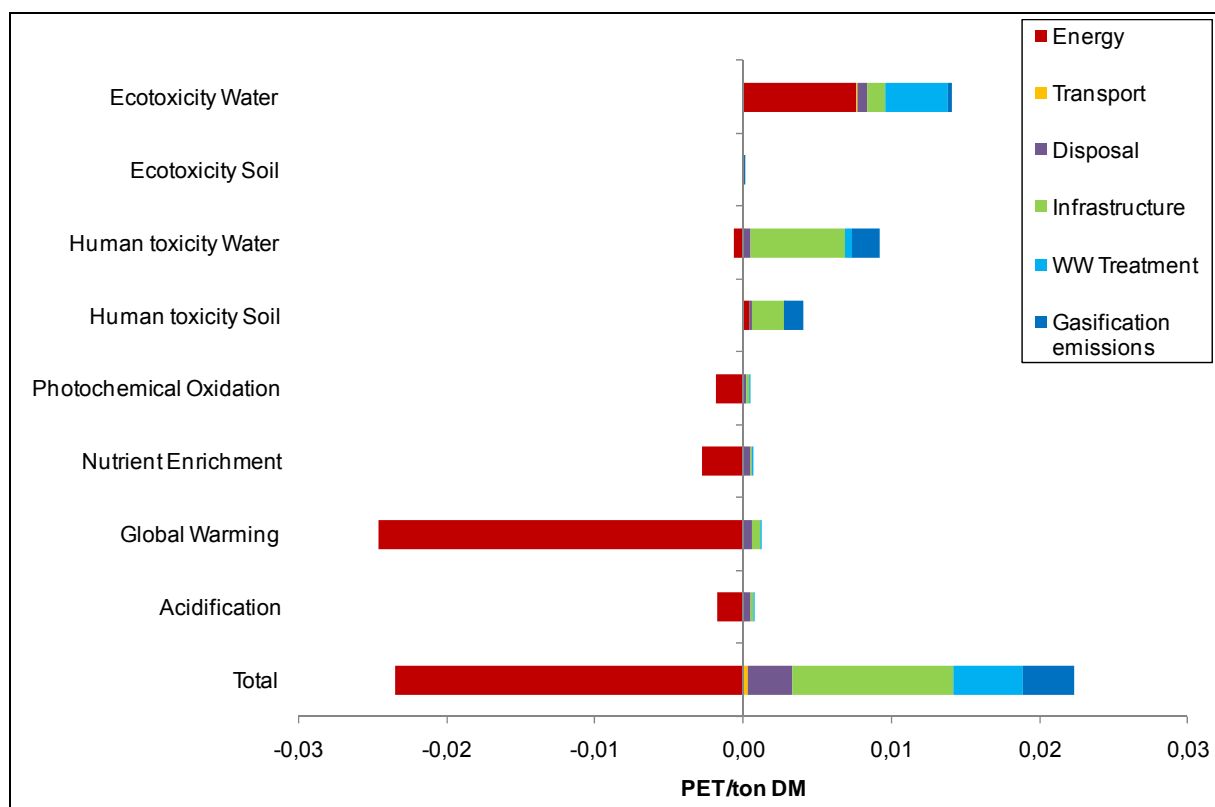


Figure 8.25 Normalized and weighted LCA impact profile showing (induced) impacts due to gasification with heat drying of 1 ton dry matter (DM) sludge as defined in Section 5.3.4 and Section 5.5.3. Weighting factor for all impact categories equals 1

If the sludge is dried by solar heating instead the impact profile shown in Figure 8.26 results. The net potential impact balance now achieves a higher negative value of - 0.036 PET/tonDM. This is due to the assumption that surplus heat is used externally (and more than compensate for the extra electricity consumption) and substitute's fossil fuel based heat.

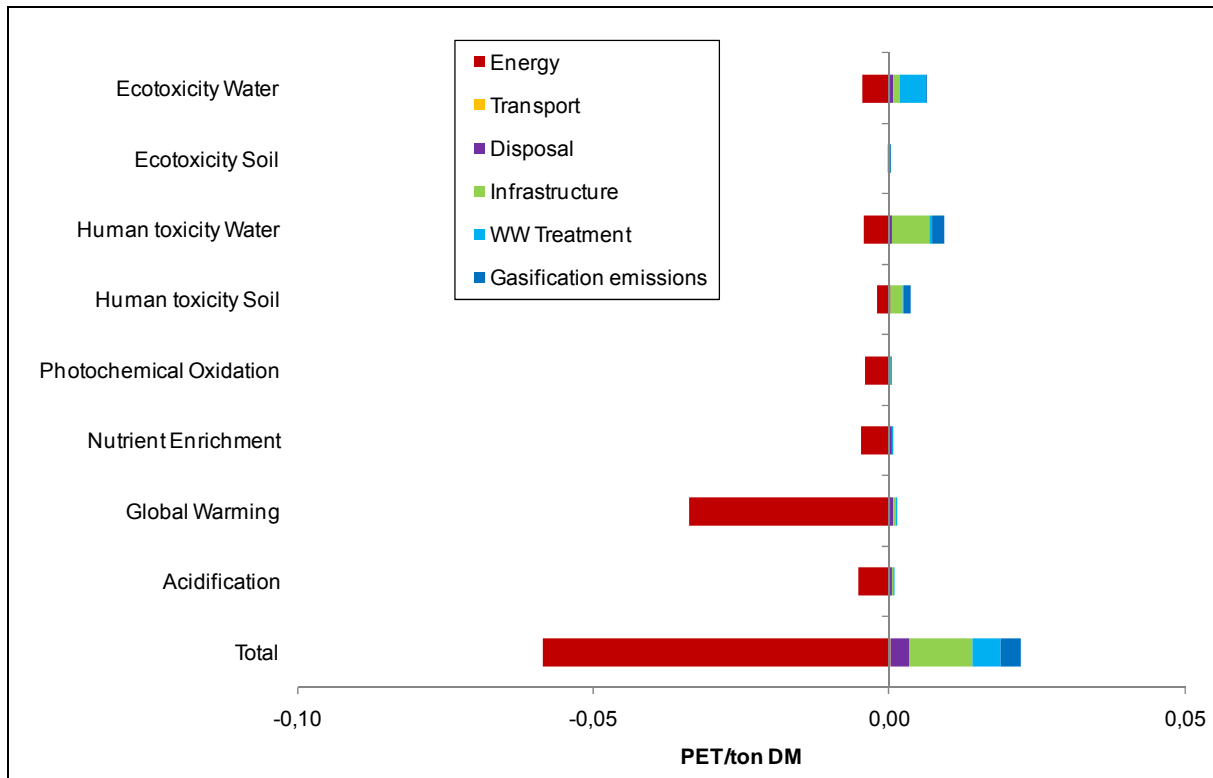


Figure 8.26 Normalized and weighted LCA impact profile showing (induced) impacts due to gasification with solar drying of 1 ton dry matter (DM) sludge as defined in Section 5.3.4 and Section 5.5.3. Weighting factor for all impact categories equals 1

8.3.5 Environmental sustainability assessment regarding cluster 3

A comparison of impact profiles among the sludge inertization methods included in this cluster is shown in Figure 8.27.

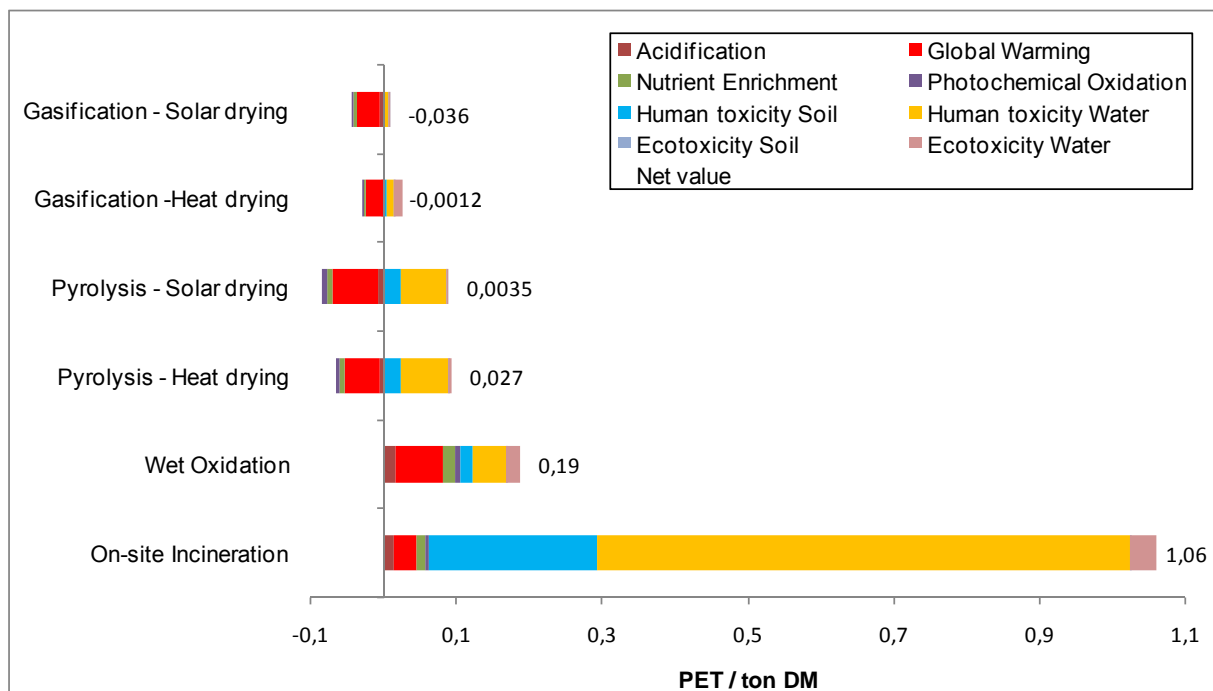


Figure 8.27 Normalized and weighted LCA impact profiles for the sludge inertization methods regarding treatment of 1 ton dry matter (DM) sludge. Weighting factor for all impact categories equals 1

As evident from Figure 8.27 treating 1 ton sludge dry matter by the on-site incineration scenario gives rise to higher potential environmental impacts than for the other five methods – especially for gasification and pyrolysis. The dominating impact categories are in almost all cases “human toxicity water” and human toxicity soil”. These impacts are in this case almost entirely determined by mercury emissions to air (and water). The transfer coefficients used (especially for mercury to air) therefore becomes very critical for the outcome.

From reading Appendix 9 to Appendix 12 it may be observed that the transfer coefficients for the metals (and especially mercury) are based on data of low or varying quality and missing data are substituted by assumptions. Based on the different estimation approaches min and max values for the mercury transfer to air may be calculated (see Appendix 13). The result is shown in Figure 8.28.

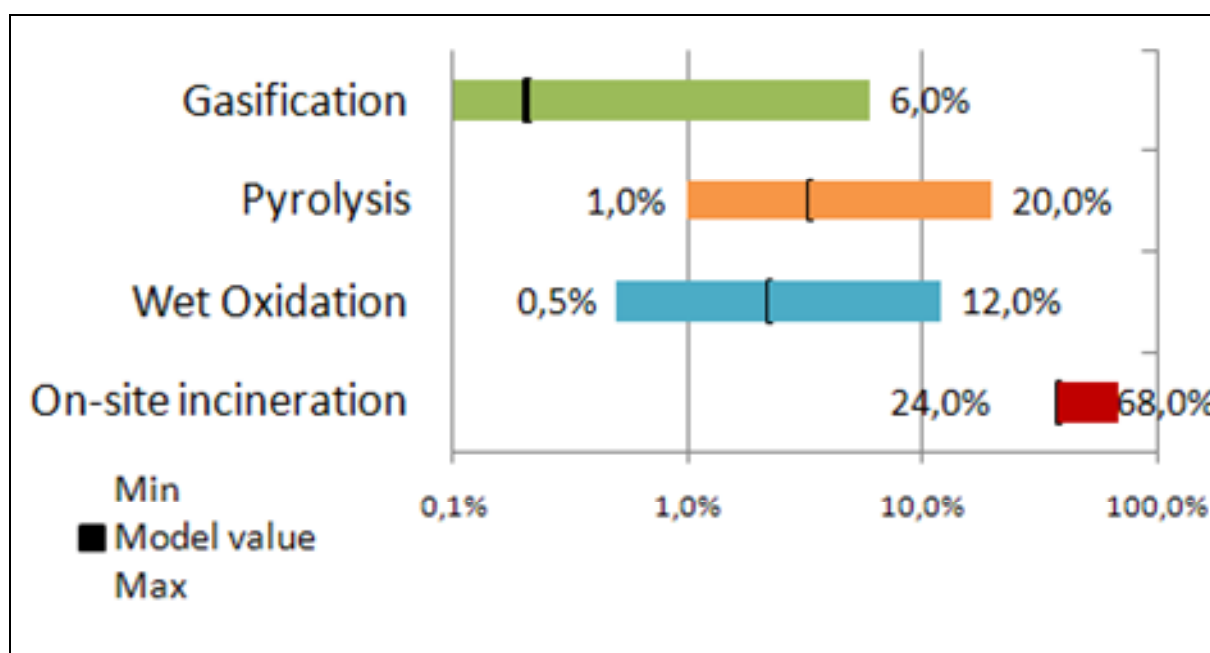


Figure 8.28 Variations in estimations of transfer coefficients for mercury to air (off-gas) for the different sludge inertization methods

The transfer coefficients for mercury to off-gas used in the models are 38.2% (on-site incineration), 3.3% (HTP), 2.2% (WO) and 0.2% (gasification). The ranges shown in Figure 8.28 indicate that differences in potential environmental impacts among pyrolysis, WO and gasification can't be identified – apparently only the on-site incineration scenario is different.

It may be argued that surplus heat from the processes assessed (especially pyrolysis and gasification combined with solar drying) is not to be allocated to the process as substituting fossil fuel based heat because that might not be the case. Anyway, erasing this advantage for the processes in question would not change the overall picture in Figure 8.27, i.e. that besides distinguishing between on-site incineration and the others no significant differences can be identified.

Another issue that are presently in focus regarding incineration of sludge is N₂O emissions, potentially contributing significantly as N₂O being a very potent gwp-gas. According to research done by Kuno et al. (2009) emissions of about 3 kg N₂O/tonDM (at 850 °C) to about 8 kg N₂O/tonDM (at 800 °C) may occur during sludge incineration. Besides global warming N₂O also contribute to the impact category on nutrient enrichment (and human toxicity but insignificant in this case). The contribution to incineration is shown in Table 8.2.

Table 8.2 Normalized and weighted impact potentials for min and max value of emitted laughing gas (N₂O) during sludge incineration. Weighting factor for all impact categories equals 1

Impact category (PET/tDM)	3 kg N ₂ O/ton DM emitted	7 kg N ₂ O/ton DM emitted
Global warming	0.11	0.26
Nutrient enrichment	0.07	0.17
Total	0.18	0.42

Including the min value from Table 8.2 in the impact profile for on-site incineration will increase the total impact from 1.06 PET/tDM to 1.24 PET/tDM (~17%) whereas using the max value will increase the total impact by ~40% (1.48 PET/tDM). Including these results in the comparison within the cluster (and implicitly assuming that no N₂O is emitted from the other processes) increases the observed difference between the size of the total environmental impacts for on-site incineration as compared to the other methods.

Though Figure 8.27 indicate a ranking from top to bottom, i.e. gasification showing negative potential impact and on-site incineration showing the highest potential impact, it is assessed that this ranking is not fully valid on an environmental sustainability scale. So, the observed differences in total potential impact among the alternative sludge handling methods does not create a basis for prioritisation regarding environmental sustainability due to high uncertainty on the input data (especially mercury emissions). The quality of a significant part of the available data is low leading to (high) uncertainty and assumptions in demand of a higher difference in potential impacts among alternatives. It can therefore only be concluded that the results indicate that gasification, pyrolysis and maybe wet oxidation, as modelled here, probably are more environmental sustainable than on-site incineration.

The electricity balances (Figure 8.29) and heat balances (Figure 8.30) for the investigated sludge inertization methods are shown below.

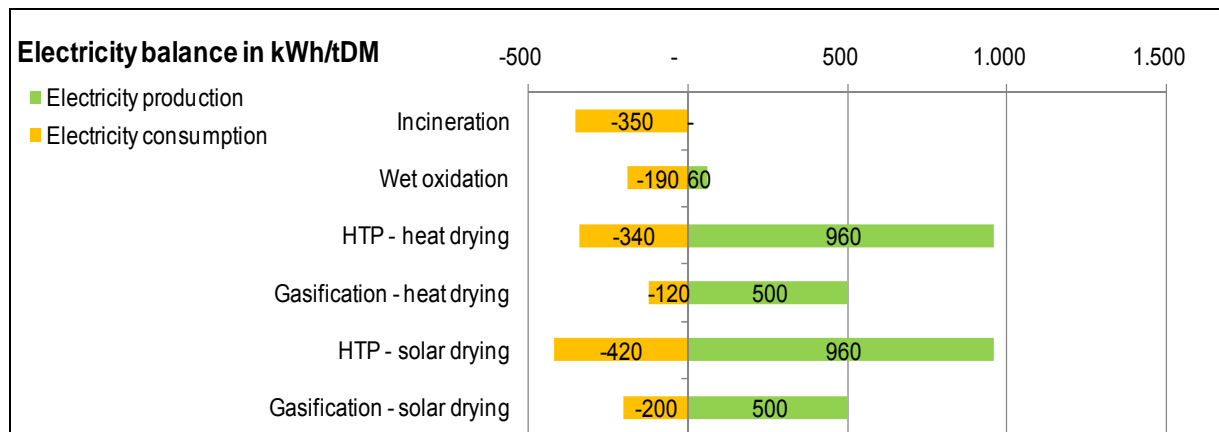


Figure 8.29 Electricity consumption and production by the different sludge inertization method scenarios

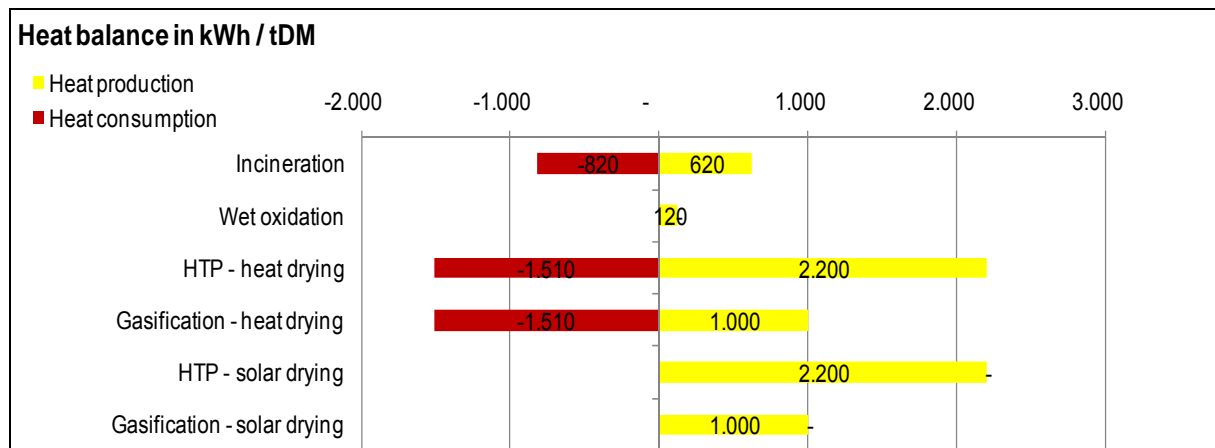


Figure 8.30 Heat consumption and production by the different sludge inertization method scenarios

8.4 Cluster 4: Sludge triage including disintegration methods

The results of the life cycle impact assessment of disposing mixed digested sludge directly on agricultural land or disposing by on-site incineration as compared to the three included alternative sludge triage technologies are shown in Figure 8.31. The dominating contributing potential impacts are in all cases associated with the impact categories “Human toxicity water” and “Human toxicity soil”. In Figure 8.32 it is evident that these impacts are related to the application of sludge on agricultural land and emissions from the on-site incineration. As for sludge inertization the most dominating contributing emissions are air emissions of mercury from the incinerator and mercury in the sludge emitted (applied) on soil.

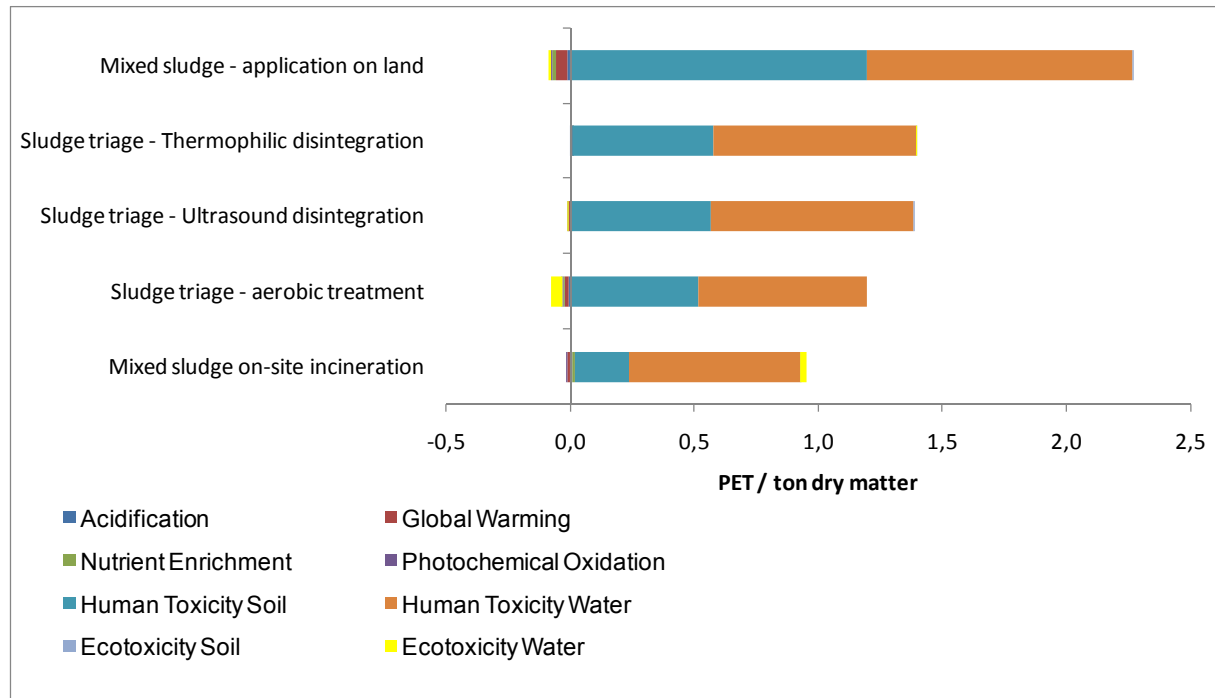


Figure 8.31 Normalized and weighted LCA impact profiles for the sludge triage scenarios and alternatives showing distribution among impact categories. Weighting factor for all impact categories equals 1

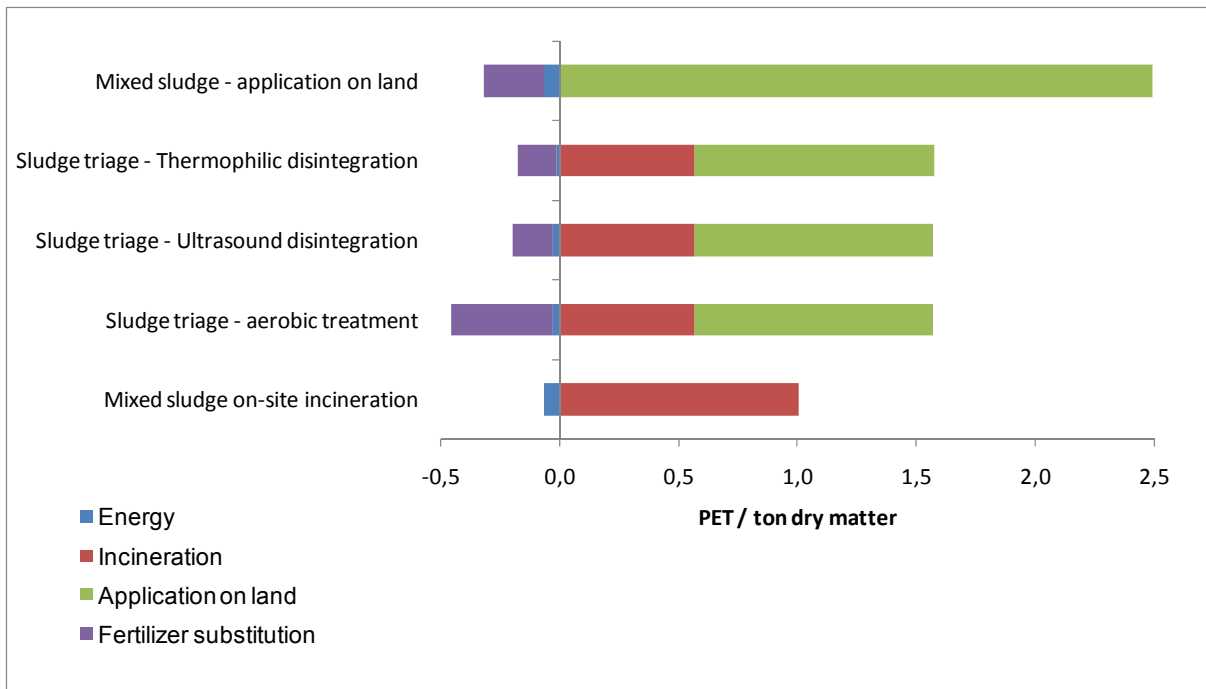


Figure 8.32 Normalized and weighted LCA impact profiles for the sludge triage scenarios and alternatives showing distribution among processes/activities. Weighting factor for all impact categories equals 1

8.4.1 Environmental sustainability assessment regarding cluster 4

Due to the dominating role of primarily metal emissions (mercury) from on-site incineration of primary sludge and metal (mercury) emissions to soil (secondary sludge application on land), it is not possible to identify any significant difference in environmental sustainability among the three assessed sludge triage methods (range 1.1 - 1.4 PET/tDM as total net value). Though both Figure 8.31 and Figure 8.32 indicate a ranking from bottom to top, i.e. on-site incineration of mixed sludge showing the lowest potential impact while mixed sludge application on land shows the highest, it is assessed that due to the high sensitivity to mercury emission and the assumptions due to lack of data (e.g. about infrastructure) no significant difference in environmental sustainability among the five scenarios can be identified. If the sludge triage methods are compared with mixed sludge on-site incineration on a net total impact potential basis, the difference is only a factor of 1.2 – 1.5 in favour of on-site incineration (mainly due to the avoided impacts (metals) from the application of secondary sludge on soil). The largest difference on a net total impact potential basis appears between mixed digested sludge applied on land and mixed sludge on-site incinerated and amounts to a factor of 2.3. It is assessed that these results only indicate that there might be a difference in environmental sustainability among the assessed scenarios but confirmation based on a more comprehensive data set of higher quality is needed. So, no significant differences in environmental sustainability among any of the technologies included in cluster 4 is possible to identify.

For sludge application on agricultural land the main purpose is substitution of mineral fertilizers. Therefore it may also be relevant to look at the metal (impurities) content of mineral fertilizers. As the dominating nutrient is the LCA impact profile is phosphorus it is relevant to look at phosphorus mineral fertilizers (or NPK fertilizers). The contents of metals as related to the total phosphorus content for the mixed sludge used in the modelling and for mineral fertilizers are shown in Table 8.3.

Table 8.3 Contents of metals in mixed sludge and mineral fertilizer as related to the total phosphorus content.

Metal	Content in mixed sludge		Content in mineral fertilizer (European average)*	Difference
	(mg/kg DM)	(mg/kg tot-P)	(mg/ kg tot-P)	(mg/kg tot-P)
Cd	0.33	15.6	82.7	-67.1
Cr	52.5	2480	1100	1380
Cu	209	9890	-	-
Hg	0.57	27.0	0.063 **	26.9
Ni	19	899	190	709
Pb	104	4920	55.3	4860
Zn	1340	63400	2290	61100

* Nziguheba and Smolders (2008) ** Based on US average of five different fertilizers from Schaffer (2001)

According to Table 8.3 all metals aside from cadmium have lower average concentrations in mineral phosphorus fertilizers than in mixed sludge. As mentioned in Section 5.5 the cadmium concentration used for mixed sludge here (based on Deliverable 1.3, Bagnuolo et al. 2009) is apparently not representative as a European average, where 3.4 mg/kg DM may be more valid (see Table 5.12B). Using this value for Cd will make the cadmium amount per kg tot-P higher for mixed sludge than for average mineral fertilizer in Table 8.3 (difference + 78 g/kg tot-P).

The main contributors to the potential impact regarding application of mixed sludge on land (see Figure 8.32) are the metal content of the sludge with a share of almost 98%. The contribution from mercury is dominating with about 64%, followed by chromium (24%), zinc (7%) and lead (4%). Cadmium only contributes with about 0.6%. If 3.4 mg/kgDM is used for cadmium instead of 0.33 mg/kgDM the contribution from cadmium will increase to about 6%.

As the dominating contributors to the potential impacts related to application on land are mercury and chromium (about 88% of total) which have significant higher concentrations in mixed sludge than in average mineral fertilizer (see Table 8.3), compensating mixed sludge used as fertilizer, for the metal content of mineral fertilizer, will not significantly change the picture in Figure 8.32. Anyway, a mercury concentration of 5 mg/kg fertilizer (assuming 20% tot-P in the product this corresponds to 25 mg/kg tot-P) is proposed as maximum permissible concentration in Australia by the Fertilizer Industry Federation of Australia (FIFA 2010). If this value is used it will have a significant effect on the potential impact from application of mixed (and secondary) sludge on land. The impact related to sludge application ("Application on land in Figure 8.32) will be reduced by about 2/3 and hereby significantly improving the impact profile for sludge application on agricultural land.

Besides metals, sludge (as waste water) contains a huge number of other micropollutants. Though it has only been possible to include seven metals in the NEPTUNE modelling of sludge triage, literature values exists for a number of organics that might be included when looking at mixed sludge application on land. For LAS, nonyl phenol, DEHP and PAH (represented by benzo(a)pyrene) ecotoxicity characterisation factors for emissions to soil also exists making it possible to calculate impact potentials (unfortunately human toxicity characterisation factors for emissions to soil only exists for benzo(a)pyrene.). In Table 8.3 the impact potentials for these four extra micropollutants are shown based on the Danish limit values for sludge applied on agricultural land.

Table 8.3 Potential contribution to the impact category on “Ecotoxicity in soil” from organic micropollutants in mixed sludge based on Danish limit values.

Substance	kg/t DM*	Ecotoxicity soil CF (m ³ /kg) **	Impact potentials (m ³ /tDM)	Normalized and weighted impact potentials (PET/tDM)#
LAS	1.3	29.0	37.7	3.91E-05
Nonylphenol	0.01	969	9.69	1.01E-05
DEHP	0.05	10.3	0.515	5.34E-07
PAH (benzo(a)pyrene)	0.003	2560	7.68	7.97E-06

* Danish limit values for application on agricultural land – the levels confirmed by actual measurements on sludge (Knudsen et al. 2000)

** Characterisations factors (CFs) from Clauson-Kaas et al. (2006)

Normalisation reference: 964,000 m³/capita/year (see Table 6.2). Weighting factor = 1

As may be evident from Table 8.3 the contribution from the four organic micropollutants (in total about 50 μPET/tDM) is insignificant as compared to the total contribution from the metals (i.e. 2.4 PET/tDM in total) but at the same level as the metals contribution to “Ecotoxicity in soil” (i.e. 21 μPET/t DM). The contribution from benz(a)pyrene to “Human toxicity soil” amounts to 43 μPET/tDM ((0.003 kg/tDM*1.8 m³/kg)/(127 m³/capita/year)*1).

The electricity balance and the heat balance for the included technologies are shown in Figure 8.33 and Figure 8.34, respectively. The highest surplus electricity production (430 kWh/ton DM) is achieved by anaerobic digestion of mixed sludge followed by application on land, and the lowest (80 kWh/ton DM) by anaerobic digestion of mixed sludge followed by on-site incineration instead. Regarding heat, the highest surplus production (830 kWh/ton DM) is achieved by combining on-site incineration of digested primary sludge with thermophilic disintegration of secondary sludge followed by anaerobic digestion and final disposal on agricultural land. The lowest surplus heat production (480 kWh/ton DM) is achieved by anaerobic digestion of mixed sludge followed by on-site incineration.

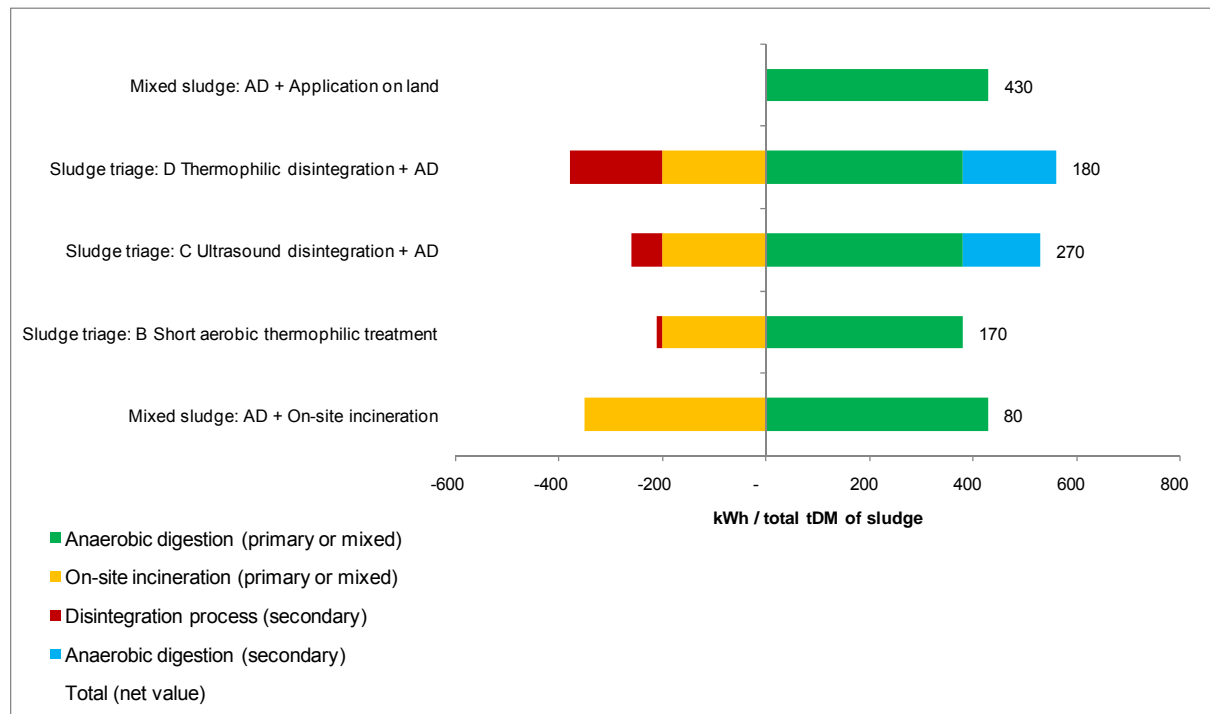


Figure 8.33 Electricity consumption (-) and production (+) by the different sludge triage scenarios and alternatives

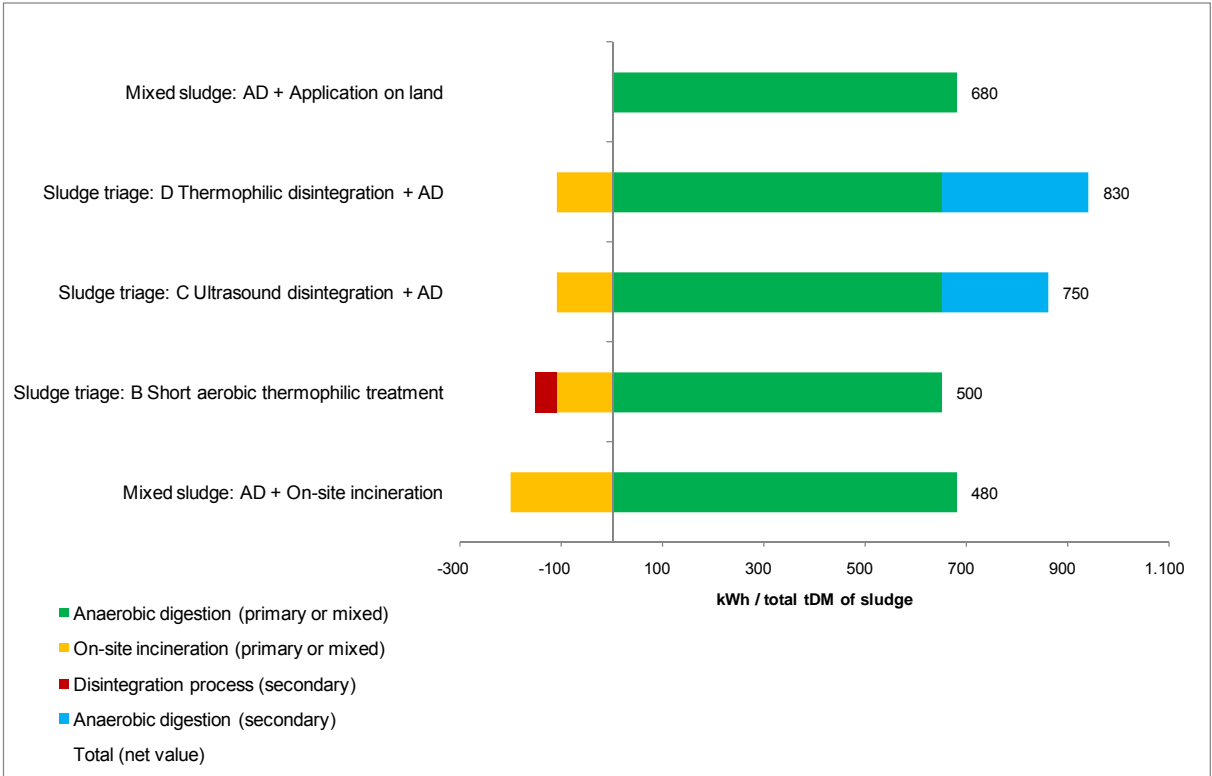


Figure 8.34 Heat consumption (-) and production (+) by the different sludge triage scenarios and alternatives

9 Cost/efficiency assessment

Within NEPTUNE useable costs have been calculated for a few of the included waste water treatment technologies. Actually, only relevant data for cluster 1, i.e. ozonation and PAC addition combined with sand filtration, have been provided.

The estimated cost/efficiency for the technologies in cluster 1 are shown in Table 9.1

Table 9.1 Cost/efficiency of cluster 1 technologies. Number of micropollutants included depending on scenario

Technology	Specification	Induced impacts	Avoided impacts	Net avoided impacts	Costs*	Cost-efficiency
		(μ PET/ m^3)	(μ PET/ m^3)	(μ PET/ m^3)		
Ozonation	22 micropollutants, 3.2gO ₃ /m ³ WW	10	11	0.55	0.07	7.9
Ozonation + SF	22 micropollutants+P+ 9 metals, 3.2gO ₃ /m ³ WW	16	220	210	0.15	1400
PAC addition	13 micropollutants, 20gPAC/m ³ WW	80	3.3	-77	0.15	-510
PAC addition + SF	15 micropollutants+P+ 9 metals, 20gPAC/m ³ WW	86	220	130	0.25	520

* Based on data for 30,000 p.e. sized ozonation (0.7-1 €/m³), ozonation combined with sand filtration, SF (0.15-2 €/m³), PAC addition (0.15-2 €/m³) and PAC addition combined with SF (0.25-3 €/m³) (Zwickenpflug (2010b))

As evident from Table 9.1 ozonation combined with sand filtration gives the highest cost/efficiency, i.e. the highest reduction in potential environmental impacts per EURO spend. However, by comparing on the basis of unequal numbers and types of micropollutants included in the modelling of the different technologies, we run the risk of introducing bias, i.e. favouring the scenarios with most micropollutants. Cost/efficiency based only on micropollutants in common for cluster 1 is therefore presented in Table 9.2.

Table 9.2 Cost/efficiency of cluster 1 technologies. Only micropollutants in common included

Technology	Specification	Induced impacts	Avoided impacts	Net avoided impacts	Costs*	Cost-efficiency
		(μ PET/ m^3)	(μ PET/ m^3)	(μ PET/ m^3)		
Ozonation	15 micropollutants, 3.2gO ₃ /m ³ WW	10	5.8	-4.3	0.07	-60
Ozonation + SF	15 micropollutants+P+ 9 metals, 3.2gO ₃ /m ³ WW	16	220	200	0.15	1400
PAC addition	15 micropollutants, 20gPAC/m ³ WW	80	3.3	-77	0.15	-510
PAC addition + SF	15 micropollutants+P+ 9 metals, 20gPAC/m ³ WW	86	220	130	0.25	520

* Based on data for 30000 p.e. sized ozonation (0.7-1 €/m³), ozonation combined with sand filtration, SF (0.15-2 €/m³), PAC addition (0.15-2 €/m³) and PAC addition combined with SF (0.25-3 €/m³) (Zwickenpflug (2010b))

The cost efficiency for ozonation alone now becomes negative but still substantially higher than for PAC addition alone. Regarding both ozonation and PAC addition combined with sand filtration the cost efficiency does not change significantly due to the all ready in common and highly dominating phosphorus and metals, see Table 9.2.

10 Conclusions and future research needs

By use of life cycle assessment (LCA) in total about 14 “new” (partly only implemented in small scale) waste water treatment technologies/systems (WWTTs) including sludge handling systems have been assessed for environmental sustainability, i.e. do we avoid more potential environmental impact than we induce by implementing them. For example by introducing ozonation we remove micropollutants from the water going to the water recipient (avoided impact) but in order to run the ozonator, oxygen production and electricity production is needed leading to emissions having an impact on the environment (induced impact). The WWTTs assessed are divided into four clusters depending on their main aim, i.e. micropollutant removal, nutrient removal, sludge inertisation and sludge optimisation regarding utilisation of energy and nutrients (trriage). It should be stressed that the results are dependent on the assumptions, scoping and constraints of the study and are valid for the generic (European) types of WWTTs defined in this report.

10.1 Conclusions

In cluster 1, focusing on micropollutant removal, the technologies/systems; ozonation, sand filtration, PAC addition in biology, ozonation followed by sand filtration and PAC addition to effluent followed by sand filtration, have all been assessed. Among these treatment systems ozonation combined with post sand filtrations seems to be the most optimal solution when looking at environmental sustainability. PAC addition to effluent combined with post sand filtration is probably also environmental sustainable but the sustainability profile is not as good as for ozonation combined with sand filtration. This is also the case when looking at cost/efficiency showing that the obtained reduction in potential environmental impact per EURO spend is about three times higher for ozonation combined with sand filtration.

Cluster 2 focus on nutrient removal and only one process has been assessed here, i.e. autotrophic anaerobic ammonium oxidation (anammox). The subject of the assessment is here a conventional WWTP without anammox as compared to a WWTP with anammox integrated. The conclusion is that introducing anammox in conventional treatment is environmentally sustainable but the result is very sensitive to the N₂O emission balance.

Cluster 3 deals with sludge inertization methods, i.e. on-site incineration, wet oxidation, gasification and pyrolysis. Here it is assessed whether or not substituting on-site incineration by any of the other alternatives (wet oxidation, gasification and pyrolysis) would be environmentally sustainable. The conclusion is that gasification, pyrolysis and maybe also wet oxidation probably are more environmentally sustainable than on-site incineration and substitutions therefore could be an environmental advantage.

Sludge triage is the subject of cluster 4. On-site incineration of mixed digested sludge or disposal on agricultural land is both used as references (conventional techniques to compare with). Three different techniques of treating primary sludge and secondary sludge separately are introduced as alternatives. Primary sludge are digested and incinerated, whereas secondary sludge is either treated by a short aerobic thermophilic treatment or by ultrasound disintegration or by thermophilic disintegration. In the two last cases the secondary sludge is afterwards treated by anaerobic digestion and in all three cases finally disposed on agricultural land. The conclusion is that no significant difference in environmental sustainability among the five scenarios can be identified. It is assessed that the results obtained only indicate that there might be a difference in environmental sustainability among the assessed scenarios but confirmation based on a more comprehensive data set of higher quality is needed.

10.2 Future research needs

During the project work in NEPTUNE on LCA of waste water treatment technologies, shortcomings, critical parameters and more have been identified. The following list describes

the most important of these issues and the status regarding ongoing/planned research addressing these problems.

- Emission of metals has turned out to be one of the dominating factors in LCIA of many of the WWTTs assessed in NEPTUNE. Especially mercury is dominating in many of the impact profiles. There is a need for better characterisation factors on metal emissions in order to confirm their high importance. For at least some of the included metals (e.g. iron) their potential impact may be overestimated and distinguishing between different species and bioavailable fractions are needed. This is not a special problem for waste water but a general problem within LCIA and is addressed in the recently started EU projects LC-Impact (terrestrial ecotoxicity) and PROSUITE (aquatic ecotoxicity).
- Emissions of nutrients have also turned out to be one of the dominating factors in LCIA of WWTTs in NEPTUNE. This issue is also in focus regarding general LCIA and is addressed in the recently started EU projects LC-Impact where new models on both a global and a site-dependent scale are going to be developed.
- In NEPTUNE it has only been possible to include a restricted number of the micropollutants that obviously appear in waste water. That including more micropollutants in the assessments can have a significant impact on the results is shown for endocrine disrupters and heavy metals. The inclusion of more micropollutants is a challenge for future research in order to improve the reliability of the LCIA outcomes on WWTTs.
- The LCIA model for toxic impacts used in NEPTUNE is based on the PNEC approach (EDIP97). Confirming the results by using an average based model (i.e. HC50-based) like USEtox is desirable. Even though USEtox was recently included in GaBi (and other LCA modelling tools) calculations of characterisation factors for especially the pharmaceuticals are needed.
- Within NEPTUNE it has only been possible to include whole effluent toxicity (WET) as regards estrogenicity. Further research on how to include WET in LCIA is addressed in the recently started EU project LC-Impact.
- Regarding pathogens it has not been possible to include potential human effects in the LCIA of NEPTUNE due to the lack of normalisation references. This is also a very relevant future research area
- Disposal to landfills are in NEPTUNE process-specific modelled. The possibilities of including substance-specific modelling of waste going to landfills would improve the modelling

11 List of abbreviations and definitions

AC	Activated carbon
AD	Anaerobic digestion
Anammox	Autotrophic anaerobic ammonium oxidation
CF	Characterisation factor
CH	EcoInvent marker for data regionality, Switzerland
DM	Dry matter
EC50	Effect concentration for 50% of the organism
GAC	Granulated activated carbon
GWP	Global warming potential
HC50	Hazardous concentration for 50% of species
HTP	High temperature pyrolysis
ICA	Instrumentation, control and automation
LC	Life cycle
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
MAD	Mesophilic anaerobic digestion
NDMA	N-nitrosodimethylamine
NR	Normalisation reference
PAC	Powdered activated carbon
PE	Person-equivalent
PNEC	Predicted no effect concentration
RA	Risk assessment
RER	EcoInvent marker for data regionality, Europe
UCTE	EcoInvent marker for data regionality, union for the co-ordination of transmission of electricity (Europe)
WET	Whole effluent toxicity
WF	Weighting factor
WO	Wet oxidation
WP	Work package
WW	Wastewater
WWT	Waste water treatment
WWTP	Waste water treatment plant
WWTT	Waste water treatment technology

REFERENCES

Ante S, Corrominas L, Ellerhorst S, Forrez I, Larsen H, Keller J, Magdeburg A, Joss A, Lachmund C, Lettl W, Mascolo G, McArdell - Buegisser C, Miladinovic N, Minnini G, Moldovan Z, Morgan F, Moser R, Moss I, Oehlmann J, Petrov P, Pratt S, Rieger L, Schlüsener M, Seyffarth T, Schulte-Oehlmann U, Siegrist H, Stalter D, Ternes T, Vanrolleghem PA., von Gunten U, Wick A, Weemaes M (2009). 2nd Periodic Activity Report. EC Project "NEPTUNE", contract No.: 036845.

BAFU/Eawag/AWEL/BMG/Hunziker (2009). Ozonung von gereinigtem Abwasser. Schlussbericht Pilotversuch Regensdorf. Dübendorf, Eawag, 16. Juni 2009.

Bagnuolo G, Braguglia CM, Gianico A, Mascolo G, Miladinovic N, Mininni G, Weemaes M (2009). Strategies for a sustainable and safe sludge management. Deliverable 1.3. NEPTUNE (New sustainable concepts and processes for optimization and upgrading municipal wastewater and sludge treatment). FP6. EU Contract-No. 036845.

Batstone DJ (2006). Sunshine Coast Regional Biosolids Feasibility Study: Final Report, Advanced Wastewater Management Center, The University of Queensland, Australia.

Boon N, Brown S, Carballa M, Forrez I, Freguia S, Johansson P, Karlsson A, Keller J, Lant P, Lee Y, Miladinovic N, Moos I, Morgan-Sagastume F, Pratt S, Rabaey K, Siegrist H, Verstraete W, von Gunten U, Werker A, Zimmermann S (2010). New approaches for a sustainable WWTP design. Deliverable 2.2. NEPTUNE (New sustainable concepts and processes for optimization and upgrading municipal wastewater and sludge treatment). FP6. EU Contract-No. 036845.

Chua CP, Lee DJ, Chang CY (2005). Energy demand in sludge dewatering. *Water Research* 39 (2005) 1858–1868.

Clauson-Kaas J, Dahl F, Dalgaard O, Høibye L, Kjølholt J, Wenzel H, Larsen HF (2006). Videregående renseteknologier for kommunalt spildevand (Advanced treatment technologies for municipal waste water). Book in Danish with English summary. DANVA, Miljøstyrelsen, Lynettefællesskabet og Spildevandscenter Avedøre. DANVA Forsknings- og Udredningsprojekt nr. 2. DANVA, Denmark. <http://www.zmag.dk/showmag.php?mid=wqrp&preview=1&x=1>

Doka G (2007a). Part III: Landfills - Underground Deposits - Landfarming. Life cycle inventories of waste treatment services – EcoInvent report No. 13. Swiss Center for Life Cycle Inventories, Dübendorf, Switzerland, December 2007.

Doka G (2007b). Part IV: Life cycle inventory of wastewater treatment. Life cycle inventories of waste treatment services – EcoInvent report No. 13. Swiss Center for Life Cycle Inventories, Dübendorf, Switzerland, December 2007.

Doka G (2007c). Part II: Waste Incineration. Life cycle inventories of waste treatment services – EcoInvent report No. 13. Swiss Center for Life Cycle Inventories, Dübendorf, Switzerland, December 2007.

Doka G (2007d). Part V: Building material disposal. Life cycle inventories of waste treatment services - EcoInvent report No. 13. Swiss Center for Life Cycle Inventories, Dübendorf, Switzerland, December 2007.

EcoInvent (2007). EcoInvent v. 2.0 (2.01). Database on life cycle inventory data. Swiss Centre for Life Cycle Inventories. The ecoinvent Centre - a Competence Centre of ETH, PSI, Empa and ART. <http://www.ecoinvent.org/home/>

Ekvall T, Mattsson N, Unger T (2004) Complex marginal electricity in the Nordic countries, *Abstracts – 14th Annual Meeting of SETAC-Europe*, Prague, Czech Republic, April 2004, p.64 (TU5AM1/01).

EU (2005). Appendix 1. The Landfill Directive. Council Directive 1999/31/EC. October 2005: http://ec.europa.eu/environment/waste/pdf/annexes_a1.pdf

Ferrari B, Mons R, Vollat B, Fraysse B, Paxeus N, Lo Giudice R, Pollio A, Garric J (2004). Environmental risk assessment of six human pharmaceuticals: Are the current environmental risk assessment procedures sufficient for the protection of the aquatic environment? *Environmental Toxicology and Chemistry*, Vol. 23, No. 5, pp. 1344–1354

Granit (2007). Waste water Treatment Plant. Wet Oxidation Technology. Date: 12/03/2007. Correspondent: Justyna Eaves; Boris Correa. Granit Technologies SA. www.granit.net.

Hansen PA (2008). A conceptual framework for life cycle assessment of wastewater treatment systems – Master thesis, DTU Management, LCA Group, Technical University of Denmark.

Hauschild M, Wenzel H, Damborg A, Tørsløv J (1998). Ecotoxicity as a criterion in the environmental assessment of products. In: Hauschild M, Wenzel H (eds) *Environmental Assessment of Products: Volume 2: Scientific background*, pp. 203-314. Chapman & Hall, London, Great Britain.

Heijungs R, Ekvall T (2009) Advances in LCI modeling. Short course at *19th Annual Meeting of SETAC-Europe*, Gothenburg, June 2009.

Hollender J, Zimmermann SG, Koepke S, Krauss M, McArdell CS, Ort C, Singer H, von Gunten U, Siegrist H (2009). Elimination of Organic Micropollutants in a Municipal Wastewater Treatment Plant Upgraded with a Full-Scale Post-Ozonation Followed by Sand Filtration. *Environ. Sci. Technol.*, 2009, 43 (20), pp 7862–7869.

Hollender J (2009). Personal communication with Juliane Hollender, Eawag. March 2009

Kampschreur MJ, Wouter RL van der Star, Wielders HA, Mulder JW, Jetten MSM, Loosdrecht MCM van (2008). Dynamics of nitric oxide and nitrous oxide emission during full-scale reject water treatment. *Water Research* 42 (2008) 812 – 826.

Knudsen L, Kristensen GH, Jørgensen PE, Jepsen SE (2000). Reduction of content of organic micropollutants in digested sludge by post aeration process - a full scale demonstration. *Water Science and Technology* Vol. 42 No. 9 pp 111-118

Larsen HF, Olsen SI, Hauschild M, Laurant A (2009). Methodology for including specific biological effects and pathogen aspects into LCA. Deliverable 4.2. NEPTUNE (New sustainable concepts and processes for optimization and upgrading municipal wastewater and sludge treatment). FP6. EU Contract-No. 036845.

Larsen HF, Hauschild M, Wenzel H, Almemark M (2007). Homogeneous LCA methodology agreed by NEPTUNE and INNOWATECH. Deliverable 4.1. NEPTUNE (New sustainable

concepts and processes for optimization and upgrading municipal wastewater and sludge treatment). FP6. EU Contract-No. 036845.

LAWA project I+II (2004/2006). Entwicklung von Umweltqualitätsnormen zum Schutz aquatischer Biota in Oberflächewässern für flussgebietspezifische Stoffe (Development of Environmental Quality Standards for the Protection of Aquatic Biota in Surface Waters). Länderfinanzierungsprogramm "Wasser und Boden" (LAWA-Projekt Nr. O 10.03 I+II). DVGW-Forschungsstelle am Engler-Bunte-Institut. 31-12-2004 (I), 30-06-2006 (II). Sponsor: Umweltbundesamt (Federal Environmental Agency), Berlin, Germany.

Mattsson N, Unger T, Ekvall T (2003) Effects of perturbations in a dynamic system – The case of Nordic power production. In Unger, T.: Common energy and climate strategies for the Nordic countries – A model analysis. PhD thesis. Chalmers University of Technology, Göteborg, Sweden.

Metcalf & Eddy (2004). Wastewater Engineering – Treatment and Reuse. International edition 2004, Mc Graw Hill.

Miege C, Choubert JM, Ribeiro L, Eusebe M, Coquery M (2009). Fate of pharmaceuticals and personal care products in wastewater treatment plants – Conception of a database and first results. *Environmental Pollution* 157, 1721-1726.

Miladinovic N (2009a). Personal communication with Natalija Miladinovic, Eawag. October 2009. Data on WO, gasification. Supplemented by an LCA report by Granit Technologies/Ecobilan on WO (Granit Technologies SA – Ecobilan OVH/Incinération – 08.07) Confidential.

Miladinovic N (2009b). Personal communication with Natalija Miladinovic, Eawag. December 2009.

Miladinovic N (2009c). Personal communication with Natalija Miladinovic, Eawag. June 2009. Description of patented Kopf gasification process: "Sustainable Valorisation of Sewage Sludge by the patented KOPF Gasification Process". Kopf Anlagenbau

Miladinovic N (2008a). Personal communication with Natalija Miladinovic, Eawag. June 2008.

Miladinovic N (2007). Personal communication with Natalija Miladinovic, Eawag. October 2007.

Metzger, S (2008). Aktivkohleeinsatz in der KA Neu-Ulm: Entnahmeraten für Arzneimittlrückstände und Kosten. Vortrag bei der gemeinsamen Sitzung der DWA-AGs IG-2.114 und AK-8.1 am 30.01.2008 in Darmstadt. Veröffentlicht in den Tagungsunterlagen.

Moser R (2010). Personal communication with Ruedi Moser, Hunziker Betatech AG. January 2010. Beilage 1-4, Betriebsreglement SVA. SVA HARD VEL. Stadtwerk Winterthur (ARA Hard).

Moser R (2009a). Personal communication with Ruedi Moser, Hunziker Betatech AG. November 2009. Infrastructure data on PAC addition.

Moser R (2009b). Personal communication with Ruedi Moser, Hunziker Betatech AG. July 2009. Infrastructure data on anammox.

Moser R (2008). Personal communication with Ruedi Moser, Hunziker Betatech AG. July 2008. Infrastructure data on ozonation and sand filtration.

Muñoz, I Llorenç Canals M i, Clift R, Doka G (2007). A simple model to include human excretion and waste water treatment in life cycle assessment of food products. CES Working Paper 01/07. ISSN: 1464-8083. Centre for Environmental Strategy, University of Surrey, Guildford (Surrey) GU2 7XH, United Kingdom. <http://www.surrey.ac.uk/CES>

Muñoz (2006). Life Cycle Assessment as a Tool for Green Chemistry: Application to Different Advanced Oxidation Processes for Wastewater Treatment. Thesis. Universitat Autònoma de Barcelona, Bellaterra, Spain. <http://www.psa.es/webeng/projects/cadox/documents/THESIS.pdf>

Nziguheba G, Smolders E (2008). Input of trace elements in agricultural soils via phosphate fertilizers in European countries. *Science of the total environment* 390 (2008) 53-57.

Parkson (2009). Thermo-system « Active solar sludge dryer », commercial report, WEF Innovative Technology Award. Contact: technology@parkson.com

PE (2008). GaBi 4. PE, LDB: GaBi 4. Software-System and Databases for Life Cycle Engineering. Copyright, TM. Stuttgart, Echterdingen. 1992-2008. <http://www.gabi-software.com/>

ProBas (2008). Prozessorientierte Basisdaten für Umweltsmanagement-Instrumente (Probas). Umweltbundesamt und Öko-Institut, Germany. (accessed 30-09-2008) http://www.probas.umweltbundesamt.de/php/volltextsuche.php?&prozessid={7A84A606-B082-4BE0-A42D-9F4A8C31FDF4}&id=1&step=1&search=Aktivkohle&b=1&show_specificum=9

Schwaiger J, Ferling H, Mallow U, Wintermayr H, Negele RD (2004). Toxic effects of the non-steroidal anti-inflammatory drug diclofenac: Part I: histopathological alterations and bioaccumulation in rainbow trout. *Aquatic Toxicology*, Volume 68, Issue 2, 10 June 2004, Pages 141-150

Shaffer M (2001). Waste lands. The threat of toxic fertilizers. Toxics Policy Advocate Calpirg Charitable Trust. The state PIRGs. May 3, 2010. California, USA.: <http://www.pirg.org/toxics/reports/wastelands/>

Siegrist H (2009a). Personal communication with Hansruedi Siegrist, Eawag. Excel sheet "DPU_Auswertung Ozonierung und PAC_sandfiltration.xls" with removal rates from DPU measurements on PAC addition followed by sand filtration. October 2009.

Siegrist H (2009b). Evaluation of anaerobic pretreatment in municipal WWTP instead of primary sedimentation with LCA. Powerpoint presentation, Dübendorf, 3 November 2009, Eawag.

Siegrist H (2008a). Personal communication with Hansruedi Siegrist, Eawag. Copy of slides from presentation by John C. Crittenden, Arizona State University: "Life Cycle Inventory of GAC for CGTF. August 2008.

Siegrist H (2008b). Personal communication with Hansruedi Siegrist, Eawag, at the NEPTUNE Varna meeting, 23-10-2008, Bulgaria.

Siegrist H, Salzgeber D, Eugster J, Joss A (2008). Anammox brings WWTP closer to energy autarky due to increased biogas production and reduced aeration energy for N-removal. *Water Science & Technology*, 57.3, pp. 383 – 388.

Stalter D, Magdeburg A, Weil M, Knacker, T, Oehlmann J (2010a). Toxication or detoxication? In vivo toxicity assessment of ozonation as advanced waste water treatment with rainbow trout. *Water Research* 44 (2010) 439 – 448.

Stalter D, Magdeburg A, Oehlmann J (2010b). Comparative toxicity assessment of ozone and activated carbon treated sewage effluents using *in vivo* test battery. *Research* 44 (2010) 2610 – 2620.

Stalter D (2008). Personal communication with PhD student Daniel Stalter, Goethe University Frankfurt am Main, Germany.

Stranddorf H, Hoffmann L, Schmidt A (2005). Impact categories, normalisation and weighting in LCA. Environmental News no. 78 from the Danish Environmental Protection Agency, Copenhagen

Ternes T A, Joss A (2006). Human Pharmaceuticals, Hormones and Fragrances. The challenge of micropollutants in urban water management. IWA Publishing, London, UK.

Triebkorn R, Casper H, Heyd A, Eikemper R, Köhler H-R, Schwaiger J (2004). Toxic effects of the non-steroidal anti-inflammatory drug diclofenac: Part II. Cytological effects in liver, kidney, gills and intestine of rainbow trout (*Oncorhynchus mykiss*). *Aquatic Toxicology*, Volume 68, Issue 2, 10 June 2004, Pages 151-166.

Weemaes M (2008). Personal communication with Marjoleine Weemaes, Aquafin, Belgium.

Wenzel H, Hauschild M, Alting L (1997). Environmental Assessment of Products, Vol. 1. First edn. Chapman & Hall, London, Great Britain.

Zwickenpflug B (2010a). Personal communication with Benjamin Zwickenpflug, Eawag. February 2010.

Zwickenpflug B, Boehler M, Hollender J, Dorusch F, Siegrist H, Fink G, Ternes, Ante S, Ellerhorst S (2010b). PAC addition to remove micropollutants: promising flow schemes. Platform presentation at the “NEPTUNE and Innwatch End User Conference: Innovative and Sustainable Technologies for Urban and Industrial Wastewater Treatment, 27th January, 2010, Ghent, Belgium. www.eu-neptune.org

Appendix 1: Physical inventory data on ozonation from Hunziker (Moser 2008)

LCA ozonation, 100'000 p.e.													
												Winterthur, 01.07.08	
component	life time category	dimension	copper [kg]	aluminium [kg]	steel [kg]	stainless steel [kg]	galvanized steel [kg]	PVC [kg]	polyethylene [kg]	concrete [m³]	composite material recyclable [kg]	perlite [kg]	Mn [kg]
tank with liquid oxygen	2	V = 21 m³	28	65	4.011	5.224	/	/	/	/	/	1060	/
evaporator	3		/	295	/	/	/	/	/	/	/	/	/
generation of ozone													
container	2	4500 x 3000 x 2591 mm	/	/	2.500	/	/	/	/	/	/	/	/
ozone generator	3	3920 x 1050 x 2210 mm	/	/	/	3.200	/	/	/	/	/	/	/
cooling unit (for ozone generator)													
other	3	1081 x 1329 x 2071 mm	10	/	/	300	110	5	5	/	20	/	/
condenser	3		50	80	/	/	/	/	/	/	/	/	/
register													
35 diffusers (ozone)	3	d = 178 mm	/	/	/	50	/	/	/	/	/	/	/
supporting structure	3	30 m, d = 1.5"	/	/	/	110	/	/	/	/	/	/	/
separating plates													
separating plates	2	d = 3 mm, 5 units	/	/	/	1.575	/	/	/	/	/	/	/
annihilation of residual ozone													
catalyst	3		9	12	/	/	/	/	/	/	/	/	51
blower / casing	3		/	/	700	/	/	/	/	/	/	/	/
pipng													
tank - ozonation	2	21.5 m, DN 30	/	/	/	39	/	/	/	/	/	/	/
ozonation - register	2	45 m, DN 30	/	/	/	81	/	/	/	/	/	/	/
cooling circuit	2	8 m, DN 63	/	/	/	/	/	/	7	/	/	/	/
exhaust air ozone reactor	2	10 m, DN 60	/	/	/	36	/	/	/	/	/	/	/
exhaust air room	2	20 m, DN 150	/	/	/	240	/	/	/	/	/	/	/
sampling pipeline	2	10 m, 2"	/	/	/	52	/	/	/	/	/	/	/
pipe socket	3	4 units à 2.5 m, 2"	/	/	/	52	/	/	/	/	/	/	/
dosing of sodium bisulfite	3	4 m	/	/	/	/	/	5	/	/	/	/	/
ozone dissolved - sensor	3	15 m, 2"	/	/	/	78	/	/	/	/	/	/	/
high pressure pipeline	3	10 m, DN 60, 0.6 - 0.7 mm	/	/	/	38	/	/	/	/	/	/	/
other													
steel sheet in front of oxygen tank	2	5 x 3.5 m d = 20 mm	/	/	2.515	/	/	/	/	/	/	/	/
guide board	2	2 x 2 m	/	/	50	/	/	/	/	/	/	/	/
bottom plate with inspection glass	1	2x1 m	/	/	/	50	/	/	/	/	/	/	/
construction													
concrete	1		/	/	/	/	/	/	/	520	/	/	/
reinforcing steel	1		/	/	53.820	/	/	/	/	/	/	/	/
total			97	452	63.596	11.125	110	10	12	520	20	1.060	51

additional remarks		Winterthur, 01.07.08	
		ozonation	sand filtration
energy consumption	kWh/m ³	0,1	0,03
liquid oxygen	flow: 10 kg O ₃ /h	55 m ³ gas/h (O ₂ +O ₃)	-
sewage quantities	m ³ /a	12.100.000	
	m ³ /a per p.e.	121	
life time categories			
	1 buildings, constructions	30 - 40 years	
	2 pipes and valves	20 - 30 years	
	3 electromechanical equipment	15 - 20 years	

Appendix 2: Physical inventory data on sand filtration from Hunziker (Moser 2008)

LCA sand filtration, 100'000 p.e.									
Winterthur, 01.07.08									
component	life time category	dimension	steel [kg]	stainless steel [kg]	galvanized steel [kg]	polyethylene [kg]	concrete [m ³]	siliceous sand [m ³]	anthracite [m ³]
pipng									
pipng	2	60m, DN 300	/	1.366	/	/	/	/	/
pipng	2	15m, DN 200	/	229	/	/	/	/	/
influent flap	2	6 * 80 kg	/	480	/	/	/	/	/
effluent (muddy water) flap	2	6 * 300 kg	/	1.800	/	/	/	/	/
pipes	2	45 m, DN 100	/	/	/	98	/	/	/
pipes	2	12 m, DN 200	/	/	190	/	/	/	/
pipes	2	45 m, DN 300	/	/	1.075	/	/	/	/
other									
blower	3	2 * 700 kg	1.400	/	/	/	/	/	/
valves	2	12 * 60 kg	1.200	/	/	/	/	/	/
valves	2	12 * 25 kg	300	/	/	/	/	/	/
pumps	3	2 * 250 kg	500	/	/	/	/	/	/
sand									
siliceous sand	1	6 * 22.5 m2, 0.7m	/	/	/	/	/	95	/
anthracite	1	6 * 22.5 m2, 0.8m	/	/	/	/	/	/	108
construction									
concrete	1		/	/	/	/	1.280	/	/
reinforcing steel	1		132.520	/	/	/	/	/	/
total			135.920	3.875	1.265	98	1.280	95	108

Regarding energy consumption, sewage quantities and life time categories see Appendix 1.

Appendix 3: Physical inventory data on PAC addition from Hunziker (Moser 2009a)

LCA PAC, 100'000 p.e.															
component	life time category	dimension	copper [kg]	aluminium [kg]	steel [kg]	stainless steel [kg]	galvanized steel [kg]	Iron [kg/a]	PVC [kg]	polyethylene [kg]	concrete [m³]	composite material recyclable [kg]	siliceous sand [m³]	anthracite [m³]	perlite [kg]
Additives															
PAC		115 t/a	/	/	/	/	/	/	/	/	/	/	/	/	/
Flocculant		11.4 t/a	/	/	/	/	/	1.482	/	/	/	/	/	/	/
Flocculant addition (FHM)		2.3 t/a	/	/	/	/	/	/	/	/	/	/	/	/	/
PAC Installation															
Stapeling container	3	V = 70 m³ (16 t) x 2	/	/	7.618	/	/	/	/	/	/	/	/	/	/
Squeezing valve (pneumatic)	3	DN 200, 0.56m, 35 kg	/	/	/	/	70	/	/	/	/	/	/	/	/
Input pipe	2	DN 200, 15m, 15.83 kg/m	/	/	/	/	475	/	/	/	/	/	/	/	/
Overl. Underpressure flap in the stapeling container ceiling	1		/	/	/	/	1	/	/	/	/	/	/	/	/
Powerful air filtration (inkl engine)	3		0	/	/	3	/	/	1	/	/	/	/	/	/
Controlsensor for overfilling (microwave)	3		0	/	2	/	/	/	/	/	/	/	/	/	/
Star feeder lock (including engine)	3	0.5 m³/h	0	/	73	/	/	/	/	/	/	/	/	/	/
Screw conveyor	3	L = 2500 mm	0	/	44	/	/	/	/	/	/	/	/	/	/
Side channel blower	3	230 Nm³/h, 350mbar	16	/	144	/	/	/	/	/	/	/	/	/	/
Suspensionpreparation-Step (including dosingstation for dried material)			/	/	/	/	/	/	/	/	/	/	/	/	/
Dosingstation for dried material (Trichter)	2	V = 135 dm³ x 2 dosingcap. 150 dm³/h	/	/	/	/	10	/	/	/	/	/	/	/	/
Connectionpipes	2	2'40 m, DN 110, 1.8 kg/m	/	/	/	/	/	/	144	/	/	/	/	/	/
Fast running stirring device	3		3	/	48	/	/	/	/	/	/	/	/	/	/
Peristaltic dosingpump	3	30kg per pump	6	/	54	/	/	/	/	/	/	/	/	/	/
Controlsensor for overfilling (microwave)	3		0	/	1	/	/	/	/	/	/	/	/	/	/
Mixing container	2	V = 0.46 m³ x 2 H: 0.850m, D: 0.925m	/	/	/	/	/	/	/	68	/	/	/	/	/
Controlbox	3	800 x 2000 x 400 mm	30	/	270	/	/	/	/	/	/	/	/	/	/
Flocculant Installation															
Stapeling container flocculant	2	V = 10 m³	/	/	/	/	/	/	/	350	/	/	/	/	/
Stapeling container flocculant addition	2	V = 5 m³	/	/	/	/	/	/	/	130	/	/	/	/	/
Fast running stirring device flocculant addition	3		1	/	13	/	/	/	/	/	/	/	/	/	/
Dosingstation flocculant addition	3		1	/	10	/	/	/	/	/	/	/	/	/	/
Input pipe	2	DN 20, 40m, 1.2 kg/m	/	/	48	/	/	/	/	/	/	/	/	/	/
Valves	2	DN 20, 10 pieces, 0.2 kg/piece	/	/	2	/	/	/	/	/	/	/	/	/	/
Dosingpump	3	5kg per pump	3	/	27	/	/	/	/	/	/	/	/	/	/
piping															
Concrete piping	3	50 m, DN 700	/	/	/	/	/	/	/	/	0.5	/	/	/	/
Connectionpipe into Flokkingreactor	2	DN 100, 4m, 9.27kg/m	/	/	/	/	37	/	/	/	/	/	/	/	/
Sedimentation - Excess Sludge	2	DN 200, 120m, 15.6kg/m	/	/	/	1872	/	/	/	/	/	/	/	/	/
Connctionpipe between the reactors	2	DN 200, 10m, 15.6kg/m	/	/	/	156	/	/	/	/	/	/	/	/	/
pumps															
Recirculationpump out of sedimentation area	3	2 * 220 kg, DN200	44	/	396	/	/	/	/	/	/	/	/	/	/
Stirring device															
Mixer (flygt)	3	10kW; including fixation	18	/	333	/	/	/	/	/	/	/	/	/	/
Horizontal stirrer	3	4kW; including fixation x2 (Contact Area)	40	/	760	/	/	/	/	/	/	/	/	/	/

additional remarks		Winterthur, 13.11.09	
		PAC	sand filtration
energy consumption	kW h/m³	0,06	0,03
PAC	flow: 315 kg PAC/d		-
sewage quantities	m³/a	12.100.000	
	m³/a per p.e.	121	
life time categories			
1	buildings, constructions	30 - 40 years	
2	pipes and valves	20 - 30 years	
3	electromechanical equipment	15 - 20 years	

Appendix 4: Physical inventory data on anammox from Hunziker (Moser 2009b)

LCA Anammox, 100'000 p.e.											Winterthur, 10.07.2009
component	life time category	dimension	copper [kg]	aluminium [kg]	steel [kg]	stainless steel [kg]	galvanized steel [kg]	polypropylen [kg]	EPDM ethylene propylene diene M-class rubber [kg]	glass [kg]	concrete [m³]
pipng											
air main	2	DN 250, 53m, 41.44 kg/m						2.196			
air distribution	2	DN 250, 2 * 28m, 20 kg/m				1.120					
flaps influent	2	1 * 34 kg, DN 125			34						
flaps air	2	4 * 76 kg, DN 250			304						
pipes influent	2	DN 125, 2 * 60m, 12.73kg/m						1.528			
pipes effluent	2	DN 125, 1 * 70m, 12.73kg/m				891					
pipes excess sludge	2	DN 100, 1 * 60m, 9.27kg/m				556					
pipes sprinkler	2	DN 50, 4*16m, 5.1kg/m						326			
other											
blow er	3	2 * 977 kg			1.954						
motor blow er	3	2 * 335 kg	67		603						
air distribution	3	DN 75, 60 * 2.5m, 0.89 kg/m						134			
aerators	3	60 * 4 * 1.5 kg							360		
mixer	3	2 * 120 kg				240					
valves	3	6 * 17 kg, DN 100			102						
valves	3	14 * 21 kg, DN 125			294						
valves air	3	5 * 77 kg, DN 250			385						
pumps excess sludge	3	2 * 180 kg, DN100	36		324						
pumps clear water	3	2 * 180 kg, DN100	36		324						
pump feeding	3	1 * 180 kg, DN 100	18		162						
control box	3	100 kg	10		90						
construction											
concrete	1										320
reinforcing steel	1				33.120						
insulation	1	57 m³ foamglas, 165 kg/m³								9.405	
total			167		37.696	2.807	4.050	134	360	9.405	320

additional remarks Anammox			Winterthur, 10.07.09
		Anammox	
energy consumption	kWh/m³ sewage quantities total	0,008	
	kWh/m³ sewage quantities only Anammox	1,1	
sewage quantities total	m³/a	12.100.000	
	m³/a per p.e.	121	
sewage quantities only Anammox	m³/a	80.000	
	m³/a per p.e.	0,8	
life time categories			
1	buildings, constructions	30 - 40 years	
2	pipes and valves	20 - 30 years	
3	electromechanical equipment	15 - 20 years	

Appendix 6: Removal rates for micropollutants regarding ozonation of secondary effluent (based on Hollender (2009))

ng/L	1.6gO3/m3WW - 1 sample				1.9gO3/m3WW - 1 sample				2.8gO3/m3WW - 1 sample				3.2gO3/m3WW - 3 samples			
	Conv RR	Oz RR	SF RR	Oz+SF RR	Conv RR	Oz RR	SF RR	Oz+SF RR	Conv RR	Oz RR	SF RR	Oz+SF RR	Conv RR	Oz RR	SF RR	Oz+SF RR
Atenolol	0,25	0,32	0,00	0,17	0,48	0,54	0,00	0,54	0,39	0,90	0,24	0,93	0,43	0,80	0,33	0,90
Bezafibrat	0,50	0,38	0,00	0,23	0,62	0,71	0,00	0,43	0,92	0,82	0,34	0,88	0,83	0,62	0,81	0,87
Carbamazepin	0,18	0,97	0,00	0,94	0,18	0,99	0,00	0,96	0,00	1,00	0,00	1,00	0,07	1,00	n/a	1,00
Clarithromycin	0,25	0,83	0,00	0,21	0,45	1,00	0,00	0,90	0,59	0,98	0,00	0,98	0,49	0,96	0,96	0,99
Clindamycin	0,89	n/a	n/a	n/a	0,32	0,56	0,84	0,93	0,44	0,98	0,00	0,98	0,34	0,95	n/a	0,95
Clofibrinsäure	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0,00	0,66	0,00	0,55
Diatrizoate	0,04	0,00	0,00	0,00	n/a	n/a	n/a	n/a	n/a	0,31	0,15	0,41	0,61	0,00	0,08	0,00
Diclofenac	0,15	0,97	0,00	0,31	0,30	0,99	n/a	0,99	0,31	1,00	n/a	1,00	0,16	1,00	n/a	1,00
Erythromycin	0,77	0,00	0,74	0,00	n/a	n/a	n/a	n/a	0,41	0,41	n/a	0,41	0,20	0,80	n/a	0,80
Ibuprofen	0,98	n/a	n/a	n/a	0,97	0,88	n/a	0,88	0,99	n/a	n/a	n/a	0,99	0,00	0,87	0,82
Iohexol	0,70	0,00	0,74	n/a	n/a	n/a	n/a	n/a	0,97	0,00	0,19	0,07	0,71	0,00	0,00	0,00
Iopamidol	0,00	0,46	0,13	0,53	0,15	0,13	0,00	0,00	0,13	0,00	0,03	0,00	0,00	0,24	0,06	0,35
Iopromid	0,00	0,56	0,00	0,46	0,69	0,00	0,15	0,15	0,83	0,19	0,09	0,26	0,21	0,26	0,00	0,29
Metoprolol	0,02	0,41	0,00	0,12	0,31	0,49	0,12	0,55	0,20	0,94	0,00	0,92	0,28	0,88	0,18	0,92
Naproxen	0,56	0,97	0,00	0,59	0,48	0,98	n/a	0,98	0,58	0,98	0,00	0,98	0,57	0,99	n/a	0,98
NDMA	0,72	-1,24	0,46	-0,22	0,94	-2,00	0,29	-1,12	-0,09	-3,08	0,27	-2,00	0,85	-1,71	0,50	-0,54
Primidon	0,00	0,60	0,00	0,50	0,82	0,00	0,44	0,00	0,05	0,55	0,00	0,55	0,22	0,62	0,10	0,66
Propranolol	0,09	0,95	0,00	0,55	0,00	0,97	n/a	0,97	0,36	0,92	0,00	0,66	0,38	0,90	n/a	0,90
Roxithromycin	0,93	n/a	n/a	n/a	0,38	0,76	n/a	0,76	n/a	0,79	0,00	0,79	0,52	0,82	n/a	0,82
Sotalol	0,06	0,96	0,00	0,47	0,00	0,97	n/a	0,98	0,27	0,98	n/a	0,99	0,26	0,98	n/a	0,99
Sulfamethoxazol	0,31	0,81	0,00	0,46	0,29	0,80	0,38	0,88	0,03	0,99	n/a	0,99	0,29	0,95	0,03	0,96
Trimethoprim	0,43	0,96	0,00	0,56	0,00	0,98	n/a	0,98	0,30	0,98	n/a	0,98	0,11	0,98	n/a	0,98
ng/L	3.3gO3/m3WW - 2 samples				3.6gO3/m3WW - 1 sample				3.7gO3/m3WW - 1 sample				5.3gO3/m3WW - 1 sample			
	Conv RR	Oz RR	SF RR	Oz+SF RR	Conv RR	Oz RR	SF RR	Oz+SF RR	Conv RR	Oz RR	SF RR	Oz+SF RR	Conv RR	Oz RR	SF RR	Oz+SF RR
Atenolol	0,37	0,95	0,27	0,96	0,53	0,79	0,17	0,83	0,27	1,00	n/a	1,00	0,26	1,00	n/a	1,00
Bezafibrat	0,87	0,86	n/a	0,87	0,71	0,81	0,00	0,78	0,86	0,90	n/a	0,90	0,80	0,89	n/a	0,90
Carbamazepin	0,14	1,00	n/a	1,00	0,00	1,00	n/a	1,00	0,00	1,00	n/a	1,00	0,00	1,00	n/a	1,00
Clarithromycin	0,54	0,99	0,99	0,99	0,40	1,00	n/a	1,00	0,67	0,99	n/a	0,99	0,44	0,99	n/a	0,99
Clindamycin	0,59	0,95	n/a	0,95	0,40	0,94	n/a	0,94	0,44	0,92	n/a	0,92	0,00	0,96	n/a	0,96
Clofibrinsäure	0,30	0,68	0,25	0,63	n/a	0,87	n/a	0,45	0,44	0,92	n/a	n/a	0,36	0,86	n/a	n/a
Diatrizoate	n/a	n/a	0,25	n/a	0,00	0,50	0,80	0,90	n/a	n/a	0,00	n/a	n/a	n/a	n/a	n/a
Diclofenac	0,32	1,00	n/a	1,00	0,18	1,00	n/a	1,00	0,23	1,00	n/a	1,00	0,35	0,99	n/a	0,99
Erythromycin	0,29	0,78	n/a	0,78	n/a	n/a	n/a	n/a	0,13	0,79	n/a	0,79	n/a	n/a	n/a	n/a
Ibuprofen	0,99	n/a	n/a	n/a	0,98	0,87	n/a	0,87	0,99	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Iohexol	0,92	0,39	0,67	0,80	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Iopamidol	0,62	n/a	n/a	n/a	n/a	0,63	n/a	0,63	0,00	0,59	0,00	0,57	0,92	n/a	0,02	n/a
Iopromid	0,85	0,35	0,21	0,48	0,00	0,20	0,00	0,20	0,79	0,43	0,21	0,54	0,87	0,49	0,17	0,58
Metoprolol	0,25	0,95	0,36	0,96	0,19	0,86	0,00	0,85	0,20	0,98	n/a	0,98	0,33	0,97	n/a	0,97
Naproxen	0,63	0,98	n/a	0,98	0,57	0,98	n/a	0,98	0,57	0,98	n/a	0,98	0,57	0,97	n/a	0,97
NDMA	0,75	-2,04	0,62	-0,03	0,18	-1,57	0,65	0,09	0,89	-4,71	0,47	-2,00	0,20	-0,04	0,77	0,76
Primidon	0,19	0,59	0,18	0,66	0,22	0,64	0,00	0,50	0,05	0,82	0,00	0,81	0,55	0,91	n/a	0,91
Propranolol	0,00	0,89	0,85	0,94	0,09	0,98	n/a	0,98	0,00	0,94	0,00	0,60	0,00	0,97	n/a	0,97
Roxithromycin	n/a	n/a	n/a	n/a	0,67	0,80	n/a	0,80	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Sotalol	0,04	0,98	n/a	0,99	0,00	0,98	n/a	0,99	0,00	0,98	n/a	0,99	0,10	0,98	n/a	0,98
Sulfamethoxazol	0,27	0,98	n/a	0,98	0,44	0,99	n/a	0,99	0,34	0,98	n/a	0,98	0,42	0,96	n/a	0,97
Trimethoprim	0,36	0,98	n/a	0,98	0,28	0,98	n/a	0,98	0,16	0,98	n/a	0,98	0,18	0,97	n/a	0,97

Conv.: Conventional RR: Removal rate Oz: Ozonation SF: Sand filtration WW: Waste water

Appendix 7: EDIP97 fresh water ecotoxicity characterization factors (CFs) for emissions to fresh water

CAS No.	Substance name	Fresh water ecotoxicity CF (m ³ /kg)*
81103-11-9	Clarithromycin	3.23E+06
18323-44-9	Clindamycin	1.17E+05
114-07-8	Erythromycin	5.00E+06
80214-83-1	Roxithromycin	3.56E+05
723-46-6	Sulfamethoxazole	1.69E+06
738-70-5	Trimethoprim	1.25E+03
41859-67-0	Bezafibrate	4.35E+05
882-09-7	Clofibric acid	4.07E+04
15307-86-5	Diclofenac	1.00E+04
15687-27-1	Ibuprofen	5.21E+03
22204-53-1	Naproxen	5.18E+03
298-46-4	Carbamazepin	4.00E+05
125-33-7	Primidone	6.94E+02
60166-93-0	Iopamidol	2.65E+00
73334-07-3	Iopromide	1.00E+01
66108-95-0	Iohexol	1.36E-01
117-96-4	Diatrizoate	9.09E+01
29122-68-7	Atenolol	2.99E+03
51384-51-1	Metoprolol	1.32E+04
3930-20-9	Sotalol	3.33E+03
525-66-6	Propranolol	2.00E+07
62-75-9	N-nitrosodimethylamin (NDMA)	2.50E+04

*

Data quality high; based on at least three measured (standard) chronic values from three trophic levels (algae, crustacean, fish)

Data quality fair; only two measured chronic effect datapoints from two trophic levels and/or based on at least three acute data points from three trophic levels (one (Q)SAR[#] data point included in a few cases)

Data quality low; more than one effect data point based on (Q)SARs[#]

(Quantitative) Structure Activity Relationships. ECOSAR is used for effect data (<http://www.epa.gov/oppt/newchems/tools/21ecosar.htm>)

Data references used for the calculation of CFs

Al-Ahmad A, Daschner FD, Kümmerer K (1999). Biodegradability of Cefotiam, Ciprofloxacin, Meropenem, Penicillin G, and Sulfamethoxazole and Inhibition of Waste Water Bacteria. Arch. Environ. Contam. Toxicol. 37, 158–163

Alexy R, Kümpel T, Kümmerer K (2004). Assessment of degradation of 18 antibiotics in the Closed Bottle Test. Chemosphere 57 (2004) 505–512

Blaise C, Gagné F, Eullaffroy P, Férard J-F (2006). Ecotoxicity of selected pharmaceuticals of urban origin discharged to the Saint-Lawrence river (Québec, Canada): a review. Braz. J. Aquat. Sci. Technol., 2006, 10(2):29-51

Brain RA, Johnson DJ, Richards SM, Sanderson H, Sibley PK, Solomon KR (2004). EFFECTS OF 25 PHARMACEUTICAL COMPOUNDS TO LEMNA GIBBA USING A SEVEN-DAY STATIC-RENEWAL TEST. Environmental Toxicology and Chemistry, Vol. 23 (2) 371-382

- Calleja MC, Persoone G, Geladi P (1994). Comparative Acute Toxicity of the First 50 Multicentre Evaluation of In Vitro Cytotoxicity Chemicals to Aquatic Non-Vertebrates. *Arch. Environ. Contam. Toxicol.* 26, 69-78
- Caminada D, Escher C, Fent K (2006). Cytotoxicity of pharmaceuticals found in aquatic systems: Comparison of PLHC-1 and RTG-2 fish cell lines. *Aquatic Toxicology* 79 (2006) 114–123.
- Carlsson C, Johansson A-K, Alvan G, Bergman K, Kühler T (2006). Are pharmaceuticals potent environmental pollutants? Part I: Environmental risk assessments of selected active pharmaceutical ingredients. *Science of the Total Environment* 364 (2006) 67– 87
- Chèvre N, Maillard E, Loepfe C, Becker-van Slooten K (2008). Determination of water quality standards for chemical mixtures; Extension of a methodology developed for herbicides to a group of insecticides and a group of pharmaceuticals. *Ecotoxicology and Environmental Safety* 71 (2008) 740– 748
- Cleuvers M (2003). Aquatic ecotoxicity of pharmaceuticals including the assessment of combination effects. *Toxicology Letters* 142 (2003) 185 -194
- Cleuvers M (2005). Initial risk assessment for three β -blockers found in the aquatic environment. *Chemosphere* 59 (2005) 199–205
- Dzialowski EM, Turner PK, Brooks BW (2006). Physiological and Reproductive Effects of Beta Adrenergic Receptor Antagonists in *Daphnia magna*. *Arch. Environ. Contam. Toxicol.* 50, 503–510
- Eguchi K, Nagase H, Ozawa M, Endoh YS, Goto K, Hirata K, Miyamoto K, Yoshimura H (2004). Evaluation of antimicrobial agents for veterinary use in the ecotoxicity test using microalgae. *Chemosphere* 57 (2004) 1733–1738
- Emblidge JP, DeLorenzo ME (2006). Preliminary risk assessment of the lipid-regulating pharmaceutical clofibrac acid, for three estuarine species. *Environmental Research* 100 (2006) 216–226
- Enick OV, Moore MM (2007). Assessing the assessments: Pharmaceuticals in the environment. *Environmental Impact Assessment Review* 27 (2007) 707-729. Referring to Marking et al. 1988
- EPIwin suite: United States Environmental Protection Agency Office of Pollution Prevention and Toxics (OPPTS) 1200 Pennsylvania Avenue, NW. Mail Code 7401M Washington, DC 20460 Telephone: (202) 564-3810. Web (EPIwin Suite v4.0): <http://www.epa.gov/oppt/exposure/pubs/episuitedi.htm> (2008 - 2009)
- Escher B, Bramaz N, Richter M, Lienert J (2006). Comparative Ecotoxicological Hazard Assessment of Beta-Blockers and Their Human Metabolites Using a Mode-of-Action-Based Test Battery and a QSAR Approach. *Environ. Sci. Technol.* 2006, 40, 7402-7408
- Ferrari B, Mons R, Vollat B, Fraysse B, Paxeus N, Lo Giudice R, Pollio A, Garric J (2004). Environmental risk assessment of six human pharmaceuticals: Are the current environmental risk assessment procedures sufficient for the protection of the aquatic environment? *Environmental Toxicology and Chemistry*, Vol. 23, No. 5, pp. 1344–1354
- FIFA (2010). Draft code of practice for fertilizer description & labeling. Fertilizer Industry Federation of Australia. 10-03-2010: http://www.fifa.asn.au/default.asp?V_DOC_ID=1148

Fraysse B, Garric J (2005). Prediction and experimental validation of acute toxicity of beta-blockers in *Ceriodaphnia dubia*. *Environmental Toxicology and Chemistry*, Vol. 24, No. 10, pp. 2470–2476

Göbel A, McArdell CS, Joss A, Siegrist H, Giger W (2007). Fate of sulfonamides, macrolides, and trimethoprim in different wastewater treatment technologies. *Science of the Total Environment* 372 (2007) 361–371

Grung M, Källqvist, Sakshaug S, Skurtveit S, Thomas KV (2008). Environmental assessment of Norwegian priority pharmaceuticals based on the EMEA guideline. *Ecotoxicology and Environmental Safety* 71 (2008) 328–340

Halling-Sørensen B, Holten Lützhøft H.-C., Andersen H. R., Ingerslev F. (2000). Environmental risk assessment of antibiotics: comparison of mecillinam, trimethoprim and ciprofloxacin. *J. Antimicrob. Chemother.*, Aug 2000; 46: 53 - 58.

Halling-Sorensen B, Nors Nielsen S, Lanzky PF, Ingerslev F, Holten Lützhøft H-C, Jørgensen SE (1998). Occurrence, fate and effects of pharmaceutical substances in the environment- A review. *Chemosphere*, Vol. 36, No.2, pp. 357-393

Henschel K-P, Wenzel A, Diedrich M, Fliedner A (1997). Environmental Hazard Assessment of Pharmaceuticals. *REGULATORY TOXICOLOGY AND PHARMACOLOGY* 25, 220–225

Hernando MD, Mezcuca M, Fernández-Alba AR, Barceló D (2006). Environmental risk assessment of pharmaceutical residues in wastewater effluents, surface waters and sediments. *Talanta* 69 (2006) 334–342

Hernando MD, Petrovic M, Fernández-Alba AR, Barceló D (2004). Analysis by liquid chromatography-electrospray ionization tandem mass spectrometry and acute toxicity evaluation for β -blockers and lipid-regulating agents in wastewater samples. *Journal of Chromatography A*, 1046 (2004) 133–140

Holten Lützhøft H-C, Halling-Sørensen B, Jørgensen SE (1999). Algal Toxicity of Antibacterial Agents Applied in Danish Fish Farming. *Arch. Environ. Contam. Toxicol.* 36, 1–6 (1999)

Huggett DB, Brooks BW, Peterson B, Foran CM, Schlenk D (2002). Toxicity of Select Beta Adrenergic Receptor-Blocking Pharmaceuticals (B-Blockers) on Aquatic Organisms. *Arch. Environ. Contam. Toxicol.* 43, 229–235

Isidori M, Lavorgna M, Nardelli A, Parrella A, Previtiera L, Rubino M (2005). Ecotoxicity of naproxen and its phototransformation products. *Science of the Total Environment* 348 (2005) 93– 101

Isidori M, Lavorgna M, Nardelli A, Pascarella L, Parrella A (2005). Toxic and genotoxic evaluation of six antibiotics on non-target organisms. *Science of the Total Environment* 346 (2005) 87-98

Isidori M, Nardelli A, Pascarella L, Rubino M, Parrella A (2007). Toxic and genotoxic impact of fibrates and their photoproducts on non-target organisms. *Environment International* 33 (2007) 635–641

Jjemba PK (2006). Excretion and ecotoxicity of pharmaceutical and personal care products in the environment. *Ecotoxicology and Environmental Safety* 63 (2006) 113–130

Jos A, Repetto G, Rios JC, Hazen MJ, Molero ML, del Peso A, Salguero M., Fernández-Freire P, Pérez-Martín JM, Cameán A (2003). Ecotoxicological evaluation of carbamazepine using six different model systems with eighteen endpoints. *Toxicology in Vitro* 17 (2003) 525–532.

- Joss A, Zabczynski S, Göbel A, Hoffmann B, Löffler D, McArdell CS, Ternes TA, Thomsen A, Siegrist H (2006). Biological degradation of pharmaceuticals in municipal wastewater treatment: Proposing a classification scheme. *Water Research* 40 (2006) 1686-1696.
- Kim Y, Choi K, Jung J, Park S, Kim P-G, Park J (2007). Aquatic toxicity of acetaminophen, carbamazepine, cimetidine, diltiazem and six major sulfonamides, and their potential ecological risks in Korea. *Environment International* 33 (2007) 370–375
- Kunkel U, Radke M (2008). Biodegradation of Acidic Pharmaceuticals in Bed Sediments: Insight from a Laboratory Experiment. *Environ. Sci. Technol.* 2008, 42, 7273–7279
- Lilius H, Isomaa B, Holmström T (1994). A comparison of the toxicity of 50 reference chemicals to freshly isolated rainbow trout hepatocytes and *Daphnia magna*. *Aquatic Toxicology* 30 (1994) 47-60
- Maurer M, Escher BI, Richle P, Schaffner C, Alder AC (2007). Comparative physiology, pharmacology and toxicology of β -blockers: Mammals versus fish. *Aquatic Toxicology* 82 (2007) 145–162
- Nalecz-Jawecki G, Persoone G (2006). Toxicity of Selected Pharmaceuticals to the Anostracan Crustacean *Thamnocephalus platyurus* - Comparison of Sublethal and Lethal Effect Levels with the 1h Rapidtoxkit and the 24h Thamnotoxkit Microbiotests. *Environ Sci & Pollut Res* 13 (1) 22 – 27
- Nikolaou A, Meric S, Fatta D (2007). Occurrence patterns of pharmaceuticals in water and wastewater environments. *Anal Bioanal Chem* (2007) 387:1225–1234
- Nunes B, Carvalho F, Guilhermino L (2005). Acute toxicity of widely used pharmaceuticals in aquatic species: *Gambusia holbrooki*, *Artemia parthenogenetica* and *Tetraselmis chuii*. *Ecotoxicology and Environmental Safety* 61 (2005) 413–419
- Owen SF, Giltrow E, Huggett DB, Hutchinson TH, Saye JA, Winter MJ, Sumpter JP (2007). Comparative physiology, pharmacology and toxicology of β -blockers: Mammals versus fish. *Aquatic Toxicology* 82 (2007) 145–162.
- Pascoe D, Karntanut W, Müller CT (2003). Do pharmaceuticals affect freshwater invertebrates? A study with the cnidarian *Hydra vulgaris*. *Chemosphere* 51 (2003) 521–528
- PharmaEcoBase. Compiled data on pharmaceutical compounds. ERAPharm. Authors: Marc BABUT, Christophe BIASINI, Arnaud COGOLUÈGNES, Benoît FERRARI, Nicolas RAIDELET, Olivier SMEDTS, Dai-Minh VO. CEMAGREF. 'REMPHARMAWATER' project. <http://pharmaecobase.lyon.cemagref.fr/> (Accessed 2008 – 2009)
- PhysProp Database. First published 1993. Syracuse Research Corporation 301 Plainfield Road, Suite 350 Syracuse, New York 13212-2510 Telephone (315) 452-8400 Fax: (315) 452-8440 Web (PhysProp): http://www.syrres.com/what-we-do/databaseforms_frame.aspx?id=386 (Accessed 2008 – 2009)
- Pomati F, Netting AG, Calamari D, Neilan BA (2004). Effects of erythromycin, tetracycline and ibuprofen on the growth of *Synechocystis* sp. and *Lemna minor*. *Aquatic Toxicology* 67 (2004) 387–396

Quinn B, Gagne F, Blaise C (2008). An investigation into the acute and chronic toxicity of eleven pharmaceuticals (and their solvents) found in wastewater effluent on cnidarian, *Hydra attenuata*. *Science of the Total Environment* 389 (2008) 306-314

Richards SM, Cole SE (2006). A toxicity and hazard assessment of fourteen pharmaceuticals to *Xenopus laevis* larvae. *Ecotoxicology* (2006) 15:647–656

Siemens J, Huschek G, Siebe C, Kaupenjohann M (2008). Concentrations and mobility of human pharmaceuticals in the world's largest wastewater irrigation system, Mexico City-Mezquital Valley. *Water Research* 42 (2008) 2124 – 2134

Stanley JK, Ramirez AJ, Mottaleb M, Chambliss K, Brooks BW (2006). Enantiospecific toxicity of the β -blocker propranolol to *Daphnia magna* and *Pimephales promelas*. *Environmental Toxicology and Chemistry*, Vol. 25, No. 7, pp. 1780–1786

Steger-Hartmann T, Länge R, Schweinfurth H (1999). Environmental Risk Assessment for the Widely Used Iodinated X-Ray Contrast Agent Iopromide (Ultravist). *Ecotoxicology and Environmental Safety* 42, 274-281

Ternes T A, Joss A (2006). *Human Pharmaceuticals, Hormones and Fragrances. The challenge of micropollutants in urban water management.* IWA Publishing, London, UK.

Vieno N, Tuhkanen T, Kronberg L (2007). Elimination of pharmaceuticals in sewage treatment plants in Finland. *Water Research* 41 (2007) 1001 – 1012

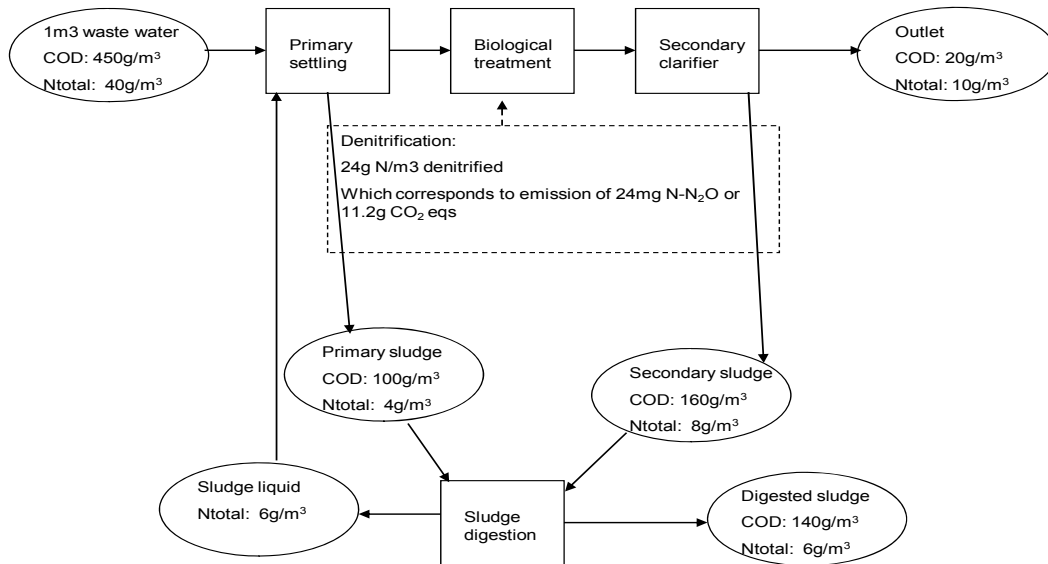
Villegas-Navarro A, Rosas-L E, Reyes JL (2003). The heart of *Daphnia magna*: effects of four cardioactive drugs. *Comparative Biochemistry and Physiology Part C* 136 (2003) 127–134

Winter MJ, Lillicrap AD, Caunter JE, Schaffner C, Alder AC, Ramil M, Ternes TA, Giltrowd E, Sumpter JP, Hutchinson TH (2008). Defining the chronic impacts of atenolol on embryo-larval development and reproduction in the fathead minnow (*Pimephales promelas*). *Aquatic Toxicology* 86 (2008) 361–369

Yamashita N, Yasojima M, Nakada N, Miyajima K, Komori K, Suzuki Y, Tanaka H (2006). Effects of antibacterial agents, levofloxacin and clarithromycin, on aquatic organisms. *Water Science and Technology* 53(11), 65-72.

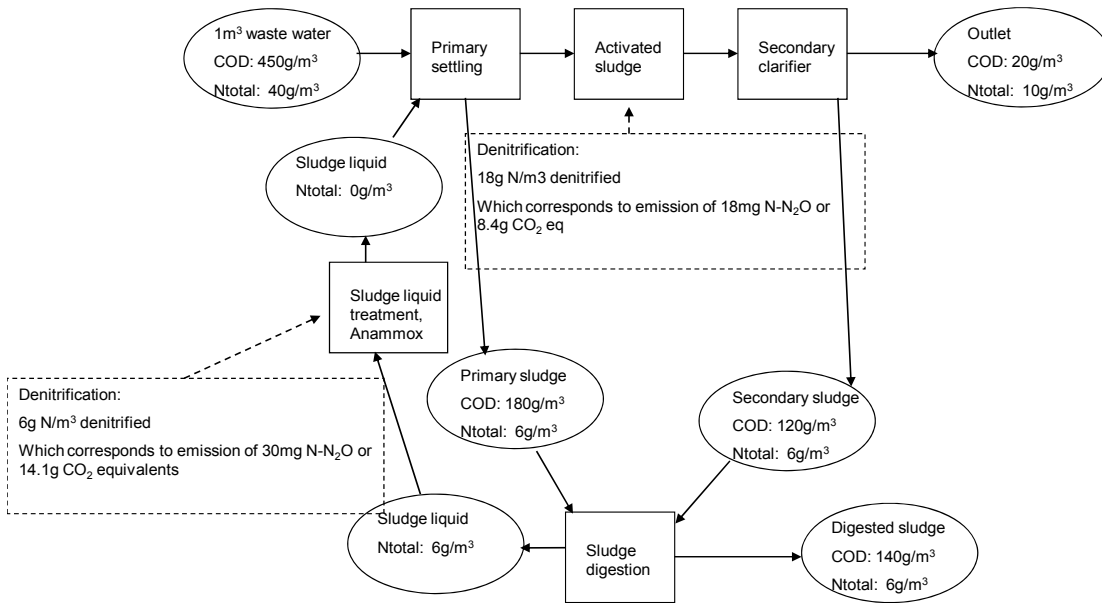
Appendix 8: Mass balance data for anammox (Miladinovic 2008a)

Conventional treatment, sludge liquid back into the process



- For all calculations:
- 468.75kg CO₂ per kg N-N₂O
 - Plant size, 100 000E
 - Qaver., 25 000m³/d

Separate sludge liquid treatment with Anammox process



Conventional treatment, sludge liquid back into the process versus Anammox; energy

Oxygen consumption and electrical energy for aeration (including separate sludge liquid treatment with anammox)	kWh/m3	
	Conventional process	Anammox
~for COD degradation	0.16	0.12
~nitrification/denitrification	0.088	0.088
Electrical energy for pumping and mixing	0.08	0.08
COD of methane and electrical energy from biogas	0.152	0.204
Net energy consumption	0.176	0.084

Appendix 9: Physical and functional inventory for on-site sludge incineration

A9.1 Description of the process

The incineration process is the reference scenario among the sludge treatment alternatives. The digested sludge is mechanically dewatered (from 4%DM → ~30%DM) before incineration. Additional fuel is needed for the combustion.

The sketch of the incineration process is shown in Figure 4.6 (Section 4). For further details about the process, please refer to Deliverable 1.3 Strategies for a safe and sustainable sludge management Bagnuolo et al. (2009).

The inventory data for the incineration scenario is based on data from ARA Hard Sludge Incineration Plant (SIP) in Winterthur, Switzerland supplied by NEPTUNE partner Eawag (described at the end of this appendix, A9.5) and NEPTUNE partner Hunziker (Moser 2010).

A9.2 Mass balance

The mass balance considered for the environmental assessment of the incineration process is summarized in the table below, from the inventory list:

	Flow	Quantity	Comment	Notation
Input	Sludge	1 tDM	After dewatering (30% DM)	SLU
Outputs	Slag	515 kg	Bottom ash and fly ash	SOL
	Liquid residue	12 250 L	Returned to WWTP	WAT
	Gas	9 250 Nm ³	Total exhaust gas production	AIR

Table A9.1 : Flows for the gasification process

The establishment of the mass balance is necessary to obtain transfer coefficients. For each compound present in the sludge, transfer coefficients will determine how many percent a given compound in the sludge goes to the outputs: solid residue, liquid residue and gas. Then a typical sludge composition can be used to allow the comparison between sludge treatment processes.

Sludge: The sludge composition is partially known for N and P (9% in weight) and for some trace heavy metals (Cd, Cr, Cu, Hg, Ni, Pb, Zn but not As). The rest of the composition will be estimated from typical sludge composition from anaerobic digestion when used for the modeling.

Solid residue: The composition of the solid residue from the incineration process is assumed to be slag ash. Its composition is known for P, N and the heavy metals Cd, Cr, Cu, Hg, Ni, Pb, Sb and Zn but not As. If it is bottom ash only, it will be disposed of in a slag compartment of a sanitary landfill. The fly ash has to be disposed of in a residual material landfill facility, both with process-specific burdens associated.

Liquid residue: The composition of the liquid residue is known for P, N, Cl and for the heavy metals Cd, Cr, Cu, Hg, Ni, Pb and Zn. It will be returned to the wastewater treatment plant to be treated. Regarding the metals the treatment of this residue is modelled in a conventional wastewater treatment model.

Gas: The emissions are known for the gases NO₂, CO, SO₂, HCl, HF and NH₃ and for the heavy

metals Cd, Hg, Pb and Zn.

From the data, the mass balance is completely known for the compounds **P, Cd, Hg, and Zn** (with detection in the measurements on the air emissions). For the compounds **Cr, Cu Pb and Ni**, the mass balance is considered to be completely known except for the air emissions, where no heavy metal can be detected, and no threshold value is indicated. Thus the air emissions for those heavy metals are set to 0. However, for the comparison with the gasification and high temperature pyrolysis processes, the missing air emissions are set equal, for a fair comparison.

A9.3 Functional inventory: Transfer coefficients

It is important to estimate the transfer coefficients so the analysis of the process can be done focusing both on the emissions and on the process inventory (energy, infrastructure and disposal).

<i>Incineration</i>	Exhaust gas	Liquid residue	Ash
Arsenic	0	45.08%	54.92%
Cadmium	0.19%	0.78%	99.03%
Chromium	0	2.13%	97.87%
Copper	0	0.26%	99.74%
Mercury	38.19%	56.18%	5.47%
Nickel	0	4.02%	95.98%
Lead	0	1.42%	98.58%
Zinc	0.23%	0.09%	99.68%
Antimony	0	35.19%	64.81%

Table A9.2: Transfer coefficients from incineration inventory list

Considering the importance of heavy metal emissions on the environmental profile of the process (especially Hg, but also the others); another set of transfer coefficients is used for the comparison with the gasification and high temperature pyrolysis processes, as follows:

<i>Incineration</i>	Exhaust gas	Liquid residue	Ash
Arsenic	0,00%	45,08%	54,92%
Cadmium	0,19%	0,78%	99,03%
Chromium*	0,11%	2,13%	97,76%
Copper*	0,01%	0,26%	99,73%
Mercury	38,19%	56,33%	5,48%
Nickel*	0,13%	4,01%	95,86%
Lead*	0,05%	1,42%	98,53%
Zinc	0,23%	0,09%	99,68%
Antimony	0,00%	35,19%	64,81%

* air emissions estimated from gasification

A9.3 : Transfer coefficients for incineration with air emissions estimated from gasification

A9.4 Physical inventory

Infrastructure

The data for the construction inventory list are unknown for the Winterthur sludge incineration plant. Instead, the infrastructure of a Swiss municipal incinerator is taken in GaBi 4.3. It has a

capacity of 100,000 tons/year and an expected lifetime of 40 years. It includes the production of all the materials in the inventory list and their corresponding disposal.

« CH: municipal waste incineration plant »

One piece of infrastructure (the SIP) will be used to treat a large amount of sludge. Consequently, in order to relate the « quantity » of infrastructure needed for the treatment of 1 t DM, or 3,3 t of waste because the sludge has a dry matter content of 30%. Consequently, 8,25.10⁻⁷ « pieces of infrastructure » are needed per t DM of sludge treated by incineration.

However, in order to model the infrastructure in a similar way in all the processes, i.e. including also the disposal and recycling of infrastructure material, especially copper materials, the process « CH: municipal waste incineration plant » has been adapted to:

« CH: municipal waste incineration plant (Neptune, with recycling) ».

The following recycling processes have been added:

Reinforcing steel	« RER: Iron scrap, at plant (inverted) » (80%) « CH: disposal, building, reinforcement steel, to final disposal » (20%)
Steel	« CH: disposal, building, bulk iron (excluding reinforcement), to sorting plant »
Concrete	« CH: gravel, unspecified, at mine (inverted) » (80%) « CH: disposal, building, reinforced concrete, to sorting plant » (20%)
Gravel	« CH: gravel, unspecified, at mine (inverted) » (80%) « CH: disposal, building, reinforced concrete, to sorting plant » (20%)

Disposal of solid waste

Slag compartment: It is used for the disposal of the solid residue. The capacity of this compartment (part of a sanitary landfill and same logistics) is 375,000 m³ for a waste with an average density of 1500 kg/m³ (density assumed). The mass of the solid residue is 515 kg/tDM. The use phase is 30 years. Consequently the number of « pieces of slag compartment » needed are 3,05e-8 pieces per tDM. The process-specific burdens are related to the amount of waste treated.

The processes for the disposal of solid waste are:

Slag compartment « CH: slag compartment »
(GaBi 4.3) « CH: process-specific burdens, slag compartment »

Chemicals: The physical inventory of the chemicals consumed during the treatment of 1t DM of sludge in the SIP lists the quantity of each chemical consumed during the overall process but without details on their utility (which compounds they aim to reduce for instance) and neither on the precise treatment step they are used in. However, the qualitative utility of each chemical can be found out using knowledge about the incineration process.

Chemical	Amount (kg/kg DM)	Addition al water (kg/kg)	Utility
Flocculant (copolymers of	0.22	45.56	Dewatering

acrylamide) <i>0.48% solution</i>			
Quarry sand for filtration	0.028		Filtration
Calcium chloride (CaCl ₂)	0.00065		Heavy metal removal in wet scrubber production of gypsum (with Ca ²⁺ ions) if NaOH used Water-soluble neutralization product
TMT15/Na 3T (sodium 2,4,6-trimercapto-s- triazine) <i>15% solution</i>	0.16	0.88	Organic sulphur compound Improve heavy metal precipitation in wet scrubber especially HgS and CdS
Sodium persulfate (Na ₂ S ₂ O ₈)	0.00064		Na ₂ S (inorganic sulphides) to improve precipitation of heavy metals, esp. HgS & CdS. Water-soluble neutralization product
Sodium hydroxide (NaOH)	0.034		Neutralization agent for SO ₂ in alkaline scrubbing and in wastewater from scrubber and pH adjustment
NH ₃ <i>45% ammonia</i>	5.07	6.20	Reduction of NO _x to N ₂ (SNCR=Selective Non Catalytic Reduction)
Hydrochloric acid (HCl)	0.0020		<i>Assumption: for pH control in wastewater from the wet scrubber</i>
Sulfuric acid (H ₂ SO ₄)	0.00120		<i>Assumption: for pH control in wastewater from the wet scrubber</i>
Salt for softening (NaCl)	0.00059		Water-soluble neutralization product

Table A9.4: Chemical utility in incineration

The chemical production is modelled with the following processes in GaBi 4.3:

Chemical	Process (GaBi 4.3)
Flocculant	GLO: chemicals inorganic, at plant
Quarry sand for filtration	CH: sand, at mine
Calcium chloride (CaCl ₂)	RER: calcium chloride, CaCl ₂ , at plant
TMT15/Na 3T	GLO: chemicals organic, at plant
Sodium persulfate (Na ₂ S ₂ O ₈)	GLO: sodium persulfate, at plant
Sodium hydroxide (NaOH)	RER: sodium hydroxide, 50% in H ₂ O, production mix, at plant
NH ₃	CH: ammonia, liquid, at regional storehouse
Hydrochloric acid (HCl)	RER: hydrochloric acid, 30% in H ₂ O, at plant
Sulfuric acid (H ₂ SO ₄)	RER: sulphuric acid, liquid, at plant
Salt for softening (NaCl)	RER: sodium chloride, powder, at plant

Table A9.5: Chemical processes in GaBi 4.3 model

Water consumption: The water consumption is the water needed to dilute the solutions of chemicals, for TMT15, NH₃ and the flocculant.

« CH: tap water, at user »

Transport: The slag is transported to final disposal by lorry inside Switzerland.

Transport	Cargo	Distance	Quantity
« CH: transport, lorry >28t, fleet average »	Slag: 515 kg	100 km	51,5 tkm

Energy: The plant needs externally supplied fuel and electricity to run the incineration process, where it produces heat. Part of the electricity comes from the digesters of the wastewater treatment plant, part of the electricity comes from the power grid. Fuel oil and biogas from the wastewater treatment plant are used as fuels.

The processes used in GaBi 4.3 are:

Electricity consumption	« CH: electricity, at cogen with ignition biogas engine, allocation exergy » « CH: electricity, at cogen 500kWe lean burn, allocation exergy »
Fuel consumption	« CH: biogas, from sewage sludge, at storage » « CH: light fuel oil, at regional storage »
Heat production	« CH: heat, at cogen 500kWe lean burn, allocation exergy (inverted, neptune) »

Lean burn is assumed to be the source of energy used commonly and thus replaced by the heat production of the process.

Energetical inventory list:

Electricity consumption (from digester)	113 kWh/tDM	- 330 kWh/tDM
Electricity consumption (external supply)	221 kWh/tDM	
Consumption of fuel oil: 6.8 kg/tDM	(Energy content) ~ 80 kWh/tDM	~ - 820 kWh/tDM
Consumption of biogas: 115.7 kg/tDM	(Energy content) ~ 740 kWh/tDM	
<i>Heat production ???</i>	<i>193 kW or kWh ???</i>	<i>+ ???</i>

Characteristics of light fuel oil: Heat value = 43 MJ/kg [PE-GaBi 4.3 2006]

Characteristics of Biogas: Heat value = 23MJ/kg [PE-GaBi 4.3 2006]

Table A9.6: Energy inventory list

Heat: The heat production has to be estimated because the inventory list on incineration is not complete and contains incorrect information.

From the inventory list (Section A9.5), the thermal energy production from incineration is written to be 193 kWh/tDM, called as heat production. Considering that the energy of the external fuels (fuel oil and biogas) added to the process is 820 kWh/tDM, this means that the incineration process is not even efficient to get the heat from burning a high calorific value fuel.

Fortunately, another LCA on sludge incineration (included in a wet oxidation report) is available from Granit (Miladinovic 2009a). This report is for confidential use in the NEPTUNE project but

some data can be used as a comparison for the incineration process. External fuel is added as well, with an energy content of 1440 kWh/tDM. The heat produced in the incinerator is 900 kWh/tDM, which is 62,5% of the energy content of the external fuel. For comparison, in the Winterthur SIP, only 25% of this heat was recovered.

In fact, description of the ARA SIP Beilage (Moser 2010) shows power data instead of energy data:

Wärmenutzung durch Heisswassererzeugung und Speicher Umformerleistung	Heat exchanger to warm water from 75°C to 130°C	500 kW
Alternative Wärmenutzung durch Turbogenerator	Heat production	193 kW
Durchsatz entwässerter Schlamm DM 32%	Capacity	1120 kgDM/h

Table A9.7 : Power data for the sludge incineration plant

In order to get an energy production, the time during which the SIP is running is needed. Considering that 3140 tDM of sludge are treated annually (Inventory list Section A9.5) and that the capacity is 1120 kgDM/h (Moser 2010), the SIP is running 2805 hours over a year (8550 hours) at full capacity and not the rest of the time.

A more realistic estimation of the heat production is:

Type of heat use	Power	Energy	Energy gain
Heat production	193 kW	170 kWh/tDM	+ 620 kWh / tDM
Heat exchanger	500 kW	450 kWh/tDM	

Table A9.8: Heat production for incineration

Electricity consumption

Electricity consumption	Dewatering	- 20 kWh/tDM	External electricity supply
	Incineration	- 110 kWh/tDM	Digester gas use
	Incineration	- 220 kWh/tDM	External electricity supply
Total electricity consumption	- 350 kWh/tDM		

Table A9.9: Electricity consumption for incineration

The energy consumption for dewatering (about 1 kWh/ ton water released) is an estimations based on data from Batstone (2006) and Chau et al. (2005).

Energy efficiency: The energy gain from the incineration process itself starting with 30% DM sludge (620kWh/tDM) is not higher than the sludge energy content (360kWh/tDM at 30%) plus the fuel energy content (820kWh/tDM). This is a simple check.

Energy balance

<i>Incineration</i>	Electricity flows (kWh/tDM)
Electricity balance	- 350 kWh/tDM

Table A9.10: Electricity balance for incineration

<i>Incineration</i>	Heat flows (kWh/tDM)
Heat consumption – as fuel	- 820
Heat production – heat exchanger	+ 450
Heat production	+ 170
Heat / fuel balance	- 200 kWh/tDM

Table A9.11: Heat balance for incineration

A9.5 Raw data for functional inventory (Miladinovic 2007).**Dewatering and Incineration, Sludge Incineration Plant (SIP) Hard, Winterthur**

Composition of the digested sludge			<i>g/kg</i>
	Nitrogen (total)		60,4
	Phosphorus (P ₂ O ₅)		74,18
	Calcium		64,02
	Magnesium		7,63
	Kalium		5,1
	Heavy Metals	Limit. values	<i>mg/kg</i>
	Cadmium	5	1,92
	Chromium	500	32,33
	Copper	600	425
	Nickel	80	25,0
	Mercury	5	1,06
	Zink	2000	1164
	Molybdenum	20	<10
	Lead		85,2
	Cobalt	60	11,35
	AOX	500	276

Total amount of treated sludge 3 142 t DM

Fuel consumption

	Total	Per tDM
Fuel oil	21.35t	
Bio gas (from the plant)	363.38t	
Total, calculated as fuel oil	384.73t	122kg

Power consumption

External supply	221kWh
Internal supply (self-production)	113kWh
Total	334kWh

Thermal energy production

193kwh

Ashes produced, disposed 8km further by trucks	1 619t	0.52t
		<u>mg/kg</u>
Phosphorus (total)		59700
Nitrogen (total)		3,6
Antimony		29
Chlorine		2700
Lead		165
Cadmium		3
Chromium		109
Copper		907
Nickel		56,7
Mercury		0,07
Zink		2550
Water discharge (returned to WWTP)	38 489m ³	12.25m ³
		<u>mg/kg</u>
Phosphorus (total)		3,9
Nitrogen (total)		1,8
Chlorine		16000
Lead		0,1
Cadmium		<0.1
Chromium		0,1
Copper		0,1
Nickel		0,1
Mercury		0,03
Zink		0,1
Chemicals consumption		
Flocculant (copolymers of acrylamide, 0.48% solution)	45 775kg	0.070kg
Quarry sand for filtration	87t	27.7kg
Calcium chloride	2 050kg	0.65kg
TMT15/Na 3T (sodium 2,4,6-trimercapto-s-triazine)	1 040kg	0.05kg
Sodium persulfate	2 000kg	0.64kg
Sodium hydroxide	106 137kg	33.78kg
NH ₃	11 269kg	1.62kg
Hydrochloric acid	6 310 kg	2kg
Sulfuric acid	3 758kg	1.2kg
Salt for softening (NaCl)	1 850kg	0.59kg
Total exhaust gas production	29 067	9 251m ³
	552m ³	<u>(mg/Nm³)</u>
NO ₂		33,49

CO	11
SO ₂	4,74
O ₂	11,33
HCl	3,6
HF	<1.1
NH ₃	1,9
Pb	<0.01
Zn	0,33
Cd	<0.001
Hg	0,027

Appendix 10: Physical and functional inventory for wet oxidation of sludge

A10.1 Description of the process

Wet oxidation (also called wet air oxidation or critical oxidation) is an oxidative process where the digested sludge is partly oxidized in the reactor, and the effluent streams are gas emissions, liquid effluent and mineral residue. This process has been applied in the Orbe unit in Switzerland for urban digested sludge and a capacity of 2,000 t DM per year. The WO reactor operates at 280°C, with pure oxygen as an oxidant and with a retention time of 45min.

The digested sludge need to be slightly dewatered (from 4%DM → 10%DM) compare to the other processes (incineration, gasification and high temperature pyrolysis)where the sludge is at least dewatered to 30% DM. Because the dewatering step (and the energy and infrastructure associated) are outside the system boundaries, the benefit of a lower dewatering will be credited to the WO process as an energy gain of 10 kWh/tDM.

The sketch of the WO process is shown in Figures A10.1 below. For further details about the process, please refer to Deliverable 1.3 (Bagnuolo et al. 2009) and the report by Granit (2007).

The inventory data for the WO scenario is based on data from NEPTUNE partner Eawag (described at the end of this appendix, A10.5) including the Granit report (Granit 2007).

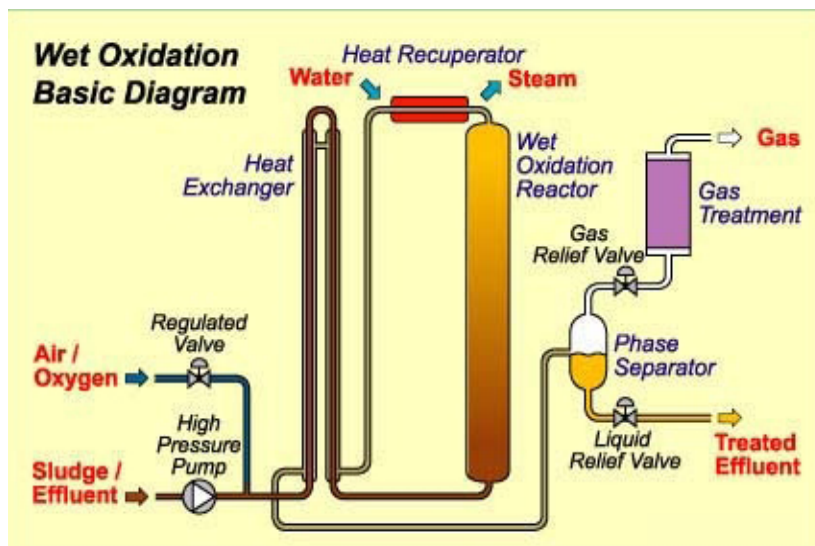


Figure A10.1: Wet oxidation process (Granit 2007)

A 10.2 Mass balance

	Flow	Quantity	Comment	Notation
Input	Sludge	1 tDM	After dewatering (10 % DM)	SLU
Outputs	Mineral residue	750 kg	Produced during cleaning of liquid	GRA
	Solid residue	0 kg	Produced in gas filter	RES
	Liquid rich with simple organic compounds	9 600 L	Available for digestion and returned to WWTP	WAT
	Gas	400 kg	From the reactor	AIR

Table A10.1: Flows for wet oxidation

The establishment of the mass balance for the process wet oxidation is necessary to obtain transfer coefficients. For each compound present in the sludge, transfer coefficients will determine how many percent a given compound in the sludge goes the outputs: mineral granulate, rich liquid and gas.

Sludge: The sludge composition is partially known (20% in weight mainly for N, P and Ca). Some heavy metals concentrations have been measured too (Cr, Cu, Co, Mo, Ni, Pb, Zn). The rest of the composition will be estimated from typical sludge composition from anaerobic digestion when used for the modeling.

Mineral residue: Its composition is known only for trace compounds (less than 1% in weight). For example, the concentrations of the heavy metals Cd, Cu, Ni, Hg, Pb, Zn are available and those concentrations are among the criteria for landfill disposal of hazardous or non hazardous waste. Moreover, from the documentation (Granit 2007), it is said that the mineral residue can be used for road constructions or for brick production, which means that it could be disposed of in a inert material landfill (Switzerland, Doka (2007a) or landfill of EU Category 3 (EU 2005).

Liquid: The composition of the liquid, rich in simple organic compounds condensate is known for the compounds AOX, As, Pb, Cd, Cr, Cu, Ni, Hg, Ag, Zn (given in mg/L). It will be returned to the wastewater treatment plant to be treated.

Off-gas: The composition of the off-gas from the energetical gas utilisation unit is known for the compounds As, Pb, Cd, Cr, Cu, Ni, Hg, Sb, Co, Mn, V and organic compounds as C. Those compounds will be emitted directly to air, through the process « Gasification » in the modelling, which includes the cogeneration system in it.

A10.3 Functional inventory: Transfer coefficients

The 7 fully tracked compounds are shown in Table A 10.2. Only the air emissions of zinc to air are unknown but they are assumed to be equal to 0. Moreover, it is not the main contributor to the environmental impact among the heavy metals, whereas the mercury emission is an important parameter.

WO	Gas	Liquid	Granulate
Cd	0,0272%	0,66%	99,31%
Cr	0,0041%	0,11%	99,88%
Cu	0,0005%	0,10%	99,90%
Hg	2,2128%	0,00%	97,79%
Ni	0,0176%	0,27%	99,71%
Pb	0,0033%	0,14%	99,86%
Zn	0,0000%	0,07%	99,93%

Table A10.2: Transfer coefficients for wet oxidation

It is important to estimate the transfer coefficients so the analysis of the process can be done focusing both on the emissions and on the process inventory (energy, infrastructure and disposal).

A10.4 Physical inventory

Infrastructure:

The lifetime of the WO plant is 20 years and its capacity is 2000tDM/year. The infrastructure materials and disposal are estimated to be the same as the gasification plant (15 years with a capacity of 1500tDM/year).

Expected life time of plant (years)	20
Capacity (ton DM/year)	2000
Concrete (kg)*	50000
Steel (kg)*	70000
Aluminium*	536
Copper*	321

* Proxy based on data from gasification (see Appendix 12)

Table A 10.3: Construction inventory list

A sub-plan called « Infrastructure (WO), Granit 1pc » has been created and is shown in Figure 5.11 in Section 5.3.2. It includes the production of all the materials in the inventory list and their corresponding disposal.

The recycling of infrastructure material is:

- « CH: Gravel, unspecified, at mine (inverted) » (80% of concrete)
- « FER: Iron scrap, at plant (inverted) » (80% of steel)

The rest of the infrastructure is disposed of as:

- « CH: disposal, building, reinforced concrete, to sorting plant » (20% of concrete)
- « CH: disposal, building, reinforced steel, to sorting plant » (20% of steel)
- « CH: disposal, aluminium, 0% water, to sanitary landfill » (100% of aluminium)
- « CH: disposal, copper, 0% water, to municipal incineration » (100% of copper)

One piece of infrastructure (the WO plant here) will be used to treat a large amount of sludge. Consequently, in order to relate the « quantity » of infrastructure needed for the treatment of 1 t DM, the expected lifetime of the plant and its capacity are needed. The expected lifetime is 20 years and the capacity of the plant is 2000 t DM year. Consequently, 2.5E-5 « pieces of infrastructure » are needed per t DM of sludge treated by WO.

Disposal of solid waste

Inert material landfill facility: The inert material landfill facility modelled in Switzerland has a capacity of 450 000 m³, and is used to landfill 750 kg/tDM of granulate residue with a density of 1067 kg/m³. The number of pieces of infrastructure per tDM is: 1.56E-6. The process-specific burdens associated are related to the amount of waste treated (750kg/tDM).

- « CH: inert material landfill facility »

« CH: process-specific burdens, inert material landfill »

Chemicals: HNO₃ at 60% is used in quantity of 0.002m³/tDM. Because it is a solution at 60% by weight and has a density of about 1.4 in mass, the quantity of nitric acid needed is 1.7 kg/tDM.

« DE: Nitric acid (60%) PE »

Oxygen consumption: 0.87 t of liquid oxygen is consumed in the WO reactor per t DM of sludge treated. The plan modelled in GaBi is "Oxygen (WO)" and shown in Figure 5.12 is Section 5.3.2.

However, the LCA from Granit states that the oxygen consumption is only 0.40 t/tDM (Miladinovic 2009a). If the impact profile of oxygen consumption is consequent in the results, this uncertainty on the exact amount of oxygen consumed could be further assessed.

Water consumption: 0.4 m³ of water is consumed per t DM. The process used in GaBi 4.3 is:

« CH: tap water, at user »

Transport: The mineral residue is transported to final disposal and the oxygen consumed is transported. There are two types of transport average lorry and distance:

Transport	Cargo	Distance	Quantity
« RER: transport, lorry >16t, fleet average »	Oxygen liquid: 870 kg	300 km	261 tkm
« CH: transport, lorry >28t, fleet average »	Mineral residue: 750 kg	100 km	75 tkm

Table A10.4: Transport physical inventory for wet oxidation

Energy balance:

From the inventory list, the following energetical data are given for the WO of dried sludge:

Electrical energy consumption: 350 kWh/tDM

Methane production gain in the reactor: 15-30%

Energy gain from digestion: 19,2 kWh/m³ of liquid

The input to the WO process is dewatered sludge with 10-15% dry matter. The dewatering is done mechanically. No heat nor electricity is directly produced during the process WO. On the other hand, the liquid rich in organic compounds is returned to the digester where it increases the biogas production from 15 to 30%.

Electricity consumption

Electricity consumption kWh/tDM	Granit (Section A10.5)	LCA Granit (Miladinovic 2009a)	Final NEPTUNE estimation
Dewatering	10	40	10
WO	350	180	180
Dewatering+WO	360	220	190
Scenario WO (mixed sludge before digestion)	-	340	-

Table A10.5: Electricity consumption for wet oxidation

The electricity consumption given in the inventory list (Section A10.5) might contain a mistake, or a misunderstanding. The electrical energy consumption stated is 350 kWh/tDM (and the inventory list is supposed to deal with the WO process itself, after digestion and dewatering of sludge). However,

the LCA made by Granit on its process (Miladinovic 2009a) reveals that the total electrical energy consumption (including thickening, digestion, dewatering and WO) is 340 kWh/tDM and that might correspond to the value of the inventory list. Therefore, the final estimation for the electrical energy consumption is 180kWh/tDM. The consumption for dewatering is as for sludge incineration based on the estimation about 1 kWh/ ton water released (lower end as dewatering is only up to 10-15% DM).

Energy gain: The energy gain is 19.2 kWh/m³ of liquid, and because there is 9.6 m³ of liquid/tDM, the final energy gain is:

+ 180 kWh/tDM

The energy gain from digestion due to the returned liquid is also calculated in the data from the LCA of the Granit company (Miladinovic 2009a). It is 20Nm³/tDM of methane, (heat value=44MJ/kg [PE-GaBi 4.3 2006]) which is equivalent of an energy gain of 160 kWh/tDM. The result from the inventory list is consequently acceptable.

The electricity production is 1/3 of the energy gain, and the heat production is 2/3.

The processes used in GaBi are:

« CH: electricity, at cogen 500kWe lean burn, allocation exergy »

« CH: heat, at cogen 500kWe lean burn, allocation exergy (inverted, neptune) »

Lean burn is assumed to be the source of energy used commonly and thus replaced by the energy production of the process.

Energy balance on the sludge

WO	Energy flows (kWh/tDM)
Electricity consumption	- 190
Electricity production	+ 60
Electricity balance	- 130 kWh/tDM

Table A10.6 : Electricity balance for wet oxidation

WO	Energy flows (kWh/tDM)
Fuel consumption	0
Heat production	+ 120
Heat / fuel balance	+ 120 kWh/tDM

Table A10.7: Heat balance for wet oxidation

The full GaBi inventory model for WO of sludge is shown in Figure 5.13 in Section 5.3.2.

A10.5 Raw data for functional inventory, inventory data Granit (Miladinovic 2009a).

Composition of the digested sludge

Nitrogen (total)

g/kg

48

Phosphorus (P ₂ O ₅)	97,5
Calcium	46,6
Magnesium	2
Kalium	1
Heavy Metals	<u>µg/kg</u>
Cadmium	<0.5
Chromium	45
Copper	202
Nickel	24,0
Mercury	
Zink	632
Molybdenum	7
Lead	21
Cobalt	8
AOX	
Total amount of treated sludge per year	2000tDM/year
Electrical energy consumption	Per tDM 350kWh
Oxygen consumption	0.87t
Thermal energy production	0
Water consumption	
technical water (WWTP outlet)	0.1m ³
network water	0.3m ³
Total	<u>0.4m³</u>
Solid residue produced during gas cleaning	0kg
Catalyst used for gas cleaning	
The heterogeneous catalyst used is in a form of honeycomb (as in a car); there are 9 elements of a volume of 9x 150x150x150	
Composition:	
Gas release to the atmosphere	0.4t
<u>Composition</u>	<u>mg/dm³</u>
CO	0,2262
NOx (as mgNO ₂)	0,0083
HF	0,001
NH ₃	0,0002
HCl	0,0013
SOx as SO ₂	0,0008
VOC (Volatile Organic Carbon)	0,0163
dioxines and furannes	8,50E-16
As	<0.0000048
Sb	<0.0000031

Cd	<0.000019
Cr	<0.000017
Co	<0.000011
Cu	<0.0000110
Mn	<0.000497
Hg	<0.000055
Ni	<0.000030
Pb	<0.000011
Tl	<0.000035
V	<0.000098

Mineral granulate produced during cleaning of liquid 0.75t

<u>Composition</u>	<u>mg/kg</u>
Cd	1
Pb	47,6
Cu	335
Ni	24,5
Zn	942
Hg	0,35
TOC	2,1

Liquid rich with simple organic compounds available for digestion, returned to WWTP and increase methane production 15-30% 9.6t

<u>Composition</u>	<u>mg/l</u>
BOD5	7800
COD	15080
DOC	61
Ptot.	69
Ntot.	3100

	<u>µg/kg</u>
Pb	<0.1
Cd	<0.01
Cr	<0.1
Cu	0,26
Ni	<0.1
Co	<0.1
Zn	0,46
Mo	0,62

Energy gain from digestion 19.2kwh/m³

Chemicals consumption m³/t
HNO₃ 60% 0,002

Construction inventory list; amount of:

stainless steel: kg
reactor 4500

"recovery" heat exchanger	500
"cooling" heat exchanger	300
drain tank	100
saparation tank	100
HP piping	100
LP piping	300
storage tank for liquid	2000
storage tank for sludge	3000
HP pump	500
other pumps	300
valves, sensors, other	500
Total	12200
Iron:	kg
press filter	1500
	m²
Filtering tissue (some synthetic material)	24
	kg
Concrete	15000
Aluminium	100
	100
Other (joints in graphite, membranes for pumps, teflon, plastics...)	
	m³
Isolation rock wool	15
expected life time of the plant	20 years

Appendix 11: Physical and functional inventory for high temperature pyrolysis (HTP) of sludge

A11.1 Description of the process

The high temperature pyrolysis (HTP) transforms the sludge into syngas and solid residues. It is similar to the process gasification (also called middle temperature pyrolysis). The digested sludge is mechanically dewatered the same way as for the incineration scenario (from 4%DM → ~30%DM). It is then dried (from 30%DM → 70-85% DM) with the excess heat from the process.

The sketch of the http process is shown in Figure 4.9 (Section 4.3). For further details about the process, please refer to Deliverable 1.3 (Bagnuolo et al. 2009) and the report by Granit (2007).

The inventory data for the HTP scenario is based on data from NEPTUNE partners Eawag and Pyromex (described at the end of this appendix, A11.5).

A11.2 Mass balance

The mass balance considered for the environmental assessment of the high temperature pyrolysis process is summarized with the following flows in the table below:

	Flow	Quantity	Comment	Notation
Input :	Sludge	1 tDM	After drying (70-85 % DM)	SLU
Outputs:	Inert solid residue	250 kg	Produced in the pyrolysis reactor	INE
	Solid residue	8 kg	Produced in the gas cooling pipes	RES
	Gas	10 000 Nm ³	From energetical gas utilisation unit, assumed to be identical as in incineration	AIR

Table A11.1: Flows for high temperature pyrolysis

Liquid residue: It is assumed that there is no liquid residue because the temperatures are too high (>1200°C). The water consumed during the process in the gas washer is 20 L/day, with 7000 tDM treated per year: 1L water / tDM.

The establishment of the mass balance is necessary to obtain transfer coefficients. For each compound present in the sludge, transfer coefficients will determine how many percent a given compound in the sludge goes to the outputs: inert residue (residue), solid residue (ash) and gas (air).

Sludge: The sludge composition is partially known for N and P (13% in weight) and for some trace heavy metals (Cd, Cr, Cu, Ni, Pb, Zn but not Hg neither As). The rest of the composition will be estimated from typical sludge composition from anaerobic digestion when used for the modelling.

Inert residue: The composition of the inert solid residue from the reactor is known for heavy metals such as Cd, Pb, Cu, Ni, Zn, Cr and Mn and PAH. Its properties make it a useful material for road construction (according to the inventory data, see Section A11.5), but as it is not done currently; it will be disposed of in an inert material landfill.

Solid residue: The composition of the solid residue from the gas cooling pipes is known for heavy metals such as Cd, Pb, Cu, Ni, Zn, Cr and Mn and PAH. Because it is a residue from gas cleaning,

it is not considered as a non hazardous waste in the EU landfill regulations, so it will be disposed of in a residual material landfill facility, with process-specific burdens associated and cement solidification.

Gas: The quantity and composition of gas emissions are unknown but as the sludge and solid residues composition is known, the gas emissions can be estimated with the mass balance for the metals: Cd, Cu, Pb, Zn.

The composition of the syngas in total heavy metals ([As, Co, Cr, Cu, Mn, Ni, Pb, Sb, Sn, V] = 0,07 mg/Nm³) is known. After combustion in the gas utilisation unit, it will go through a scrubber for heavy metals removal. Because the volume of syngas produced is not known, this information cannot be used for the mass balance.

For the uncertain transfer coefficients, an analogy with gasification can be done, especially for mercury (Hg) because it is an influential compound on the final results.

A11.3 Functional inventory: Transfer coefficients

1st estimation: the missing percentage of each compounds are assumed to be emitted as gas:

Pyrolysis	Method: repartition of the missing compounds		
	Air	Residue	Ashes
Cd	26,27%	71,88%	1,86%
Cr	-	87,40%	12,60%
Cu	27,99%	70,12%	1,89%
Hg	100,00%		
Ni	-	90,06%	9,94%
Pb	41,73%	49,38%	8,89%
Zn	80,53%	9,63%	9,84%

Table A11.2 : Transfer coefficients for HTP, maximum air emissions

→ *This assumption is taking logics to extremes and thus is not representative of the reality. Such high emissions of Hg in the air lead to an extremely bad environmental impact.*

2nd estimation: the transfer coefficient for gas in HTP are the transfer coefficient for gas and liquid in the gasification process:

HT Pyrolysis	Method analogy with gasification		
	Air	Residue	Ashes
Cd	0,18%	97,3%	2,5%
Cr	0,16%	87,3%	12,6%
Cu	0,09%	97,3%	2,6%
Hg	0,16%	-	-
Ni	0,25%	89,8%	9,9%
Pb	0,07%	84,7%	15,2%
Zn	0,13%	49,4%	50,5%

Table A11.3 : Transfer coefficients for HTP, analogy with gasification

3rd estimation: the transfer coefficient for gas in HTP are the transfer coefficients for gas and liquid in the gasification process, except for mercury and cadmium:

Another option is to estimate the transfer coefficient for mercury, by comparing the given air emissions of mercury from the inventory list (0.002 mg/Nm³) to the average composition of sludge

(European average) used as input to all sludge inertization scenarios, i.e. mercury 0.6mg/kgDM,(see Table 5.12B in Section 5.5). Furthermore, the transfer coefficient for Cd is as the only possible one based on the data in Section A11.5.

HT Pyrolysis Method: Analogy with Gasification and European average

	Air	Residue	Ashes
Cd	0,50%	97,0%	2,5%
Cr	0,16%	87,3%	12,6%
Cu	0,09%	97,3%	2,6%
Hg	3,33%	?	?
Ni	0,25%	89,8%	9,9%
Pb	0,07%	84,5%	15,2%
Zn	0,13%	49,4%	50,5%

Table A11.4 : Transfer coefficients for HTP, analogy with gasification except Hg and Cd

The transfer coefficients in Table A11.4 is considered the best proxy and used in the modelling.

A11.4 Physical inventory

Infrastructure: The data for the construction inventory list come from the pyrolysis pilot plant in Switzerland (Section A11.5). The expected lifetime is 20 years.

Construction inventory list	
Expected life time of the plant	20 years
Capacity	7 000 t DM / y
Concrete (m3)	50
Stainless steel (kg)	7 000
Steel (kg)	25 000
Aluminium (kg)	2500
Copper (kg)	1500

Table A11.5 : Construction inventory list for HTP

A sub-plan called « Infrastructure (high T°C pyrolysis), Switzerland 1 pc » has been created and is shown in Figure 5.14 in Section 5.3.3. It includes the **production** of all the materials in the inventory list and their corresponding **disposal**. 80% of concrete and iron are recycled; copper is recycled up to 90%, aluminum is disposed of. The recycling rates come from the general building material recycling rates in Doka (2007d).

The processes for the recycling of infrastructure material are:

- « CH: Gravel, unspecified, at mine (inverted) » (80% of concrete)
- « FER: Iron scrap, at plant (inverted) » (80% of steel)
- « RER: copper, at regional storage (neptune, inverted) » (90% of copper)

The processes for the disposal of infrastructure material are:

- « CH: disposal, building, reinforced concrete, to sorting plant » (20% of concrete)
- « CH: disposal, building, reinforced steel, to sorting plant » (20% of steel)
- « CH: disposal, building, bulk iron (excluding reinforcement), to sorting plant » (100% of stainless steel)
- « CH: disposal, aluminium, 0% water, to sanitary landfill » (100% of aluminium)
- « CH: disposal, copper, 0% water, to municipal incineration » (10% of copper)

One piece of infrastructure (the pyrolysis plant) will be used to treat a large amount of sludge. Consequently, in order to relate the « quantity » of infrastructure needed for the treatment of 1 t DM, the expected lifetime of the plant and its capacity are needed. The expected lifetime is 20 years and the capacity of the plant is 7000 t DM year. Consequently, $7.14E-6$ « pieces of infrastructure » are needed per t DM of sludge treated by pyrolysis.

Disposal of solid waste

Inert material landfill facility: It is used for the inert material solid residue. The capacity of this landfill is $450,000 \text{ m}^3$, the mass of the inert residue is 250 kg/tDM and its density is 1067 kg/m^3 . Consequently the number of « pieces of inert material landfill facility » needed are $5.21E-7$ pieces per tDM. The process-specific burdens are related to the amount of waste treated.

Residual material landfill facility: It is used for the residue from the gas cooling pipes and from gas treatment more generally. The capacity of this landfill is $300,000 \text{ m}^3$ for a waste with an average density of 1600 kg/m^3 (density assumed). The waste is solidified with cement and water in proportions: waste-cement-water = 50-20-30 % in weight. Because the mass of residue is 8 kg (about 16 kg solidified) $3.33E-8$ « pieces of residual material landfill facility » are needed per tDM. The process-specific burdens are related to the amount of waste treated (16 kg here).

The processes for the disposal of solid waste are:

Inert material landfill facility	« CH: inert material landfill facility » « CH: process-specific burdens, inert material landfill »
Residual material landfill facility	« CH: residual material landfill facility » « CH: process-specific burdens, residual material landfill » « CH: cement, unspecified, at plant » « CH: disposal, cement, hydrated, 0% water, to residual material landfill »

Chemicals: There is no chemical consumption in the inventory list, but it is drawn in the process that the gas cleaning is done in a scrubber (acid and base) which means that chemicals are used, but their composition and quantity is uncertain. Because there is a gas treatment (scrubber) after the gas utilisation unit, it is assumed that the same chemicals are used in the same quantities as for the gas treatment of incineration. (This is particularly interesting for the comparison incineration vs high temperature pyrolysis where the air emissions and the energy balance are the determining parameters).

Water consumption: The water consumption in the gas washer is 20 L / day, and assuming that the pyrolysis plant is running all year at its capacity of 7000 tDM / year, makes a water consumption of 1 L / t DM.

Transport: The mineral residues are transported to Swiss landfills. There are two types of transport average lorry and distance:

Transport	Cargo	Distance	Quantity
« CH: transport, lorry >28t, fleet average »	Inert residue: 250 kg	100 km	25 tkm
« CH: transport, lorry >28t, fleet average »	Solid residue: 8 kg	100 km	$0,8 \text{ tkm}$

Table A11.6: Transport inventory list

Energy balance:

The plant needs externally supplied electricity to run the pyrolysis process, where it produces gas. The energetical gas utilisation unit makes heat and electricity out of it, in an energy ratio assumed to be around 30% electricity and 70% heat. Considering that 960 kWh of electricity are produced (which is the maximum theoretical amount), the amount of heat available is 2 200 kWh.

Energy production

	Energy in kWh/tDM	
Theoretical electricity production	960	Used internally and externally
Theoretical heat production	~2 200	Consumed internally, heat loss or used externally
Total energy gains	3 160 kWh/tDM	
Energy content in the sludge at 80% DM	3 490 kWh/tDM	Using Schwanecke's formula, for complete oxidation

Table A11.7: Theoretical energy gains for HTP

Because the theoretical electricity gain has been used and that there is no measurement of the 'real' electricity gain, the HTP process is advantaged in this energy balance. It can be seen that the theoretical total energy gains are really close to the energy content of the sludge at 80%DM. For a better environmental assessment of this process, electricity production is needed.

Energy consumption

For more details about energy consumption for dewatering/drying, please refer to Appendix 12.

Option	Process	Electricity consumption (kWh/tDM)		Heat production (kWh/tDM)
Solar	Dewatering	-20	-420	+ 2 200 Used externally (assumed)
	Solar drying	-80		
	HTP	-320		
Heat	Dewatering	-20	-340	+ 2 200 Used internally and externally (assumed)
	Heat drying	None		
	HTP	-320		

Table A11.8 : Electricity consumption for HTP (solar or heat drying)

The processes used in GaBi 4.3 are:

- « CH: electricity, at cogen 500kWe lean burn, allocation exergy »
- « CH: heat, at cogen 500kWe lean burn, allocation exergy »

Lean burn is assumed to be the source of energy (used as default in NEPTUNE).

Energy balance

HTP	Electricity flows (kWh/tDM)	
	Heat drying	Solar drying
Electricity consumption	- 340	- 420
Electricity production	+ 960	+ 960
Electricity balance	+ 620	+ 540

Table A11.9: Electricity balance for HTP (solar or heat drying)

HTP	Heat flows (kWh/tDM)	
	Heat drying	Solar drying
Heat consumption as fuel / heat	- 1 510	0
Heat production	+ 2 200	+ 2 200
Heat balance	+ 690	+ 2 200

Table A11.10: Heat balance for HTP (solar or heat drying)

The full model including heat drying is shown in Figure 5.16 (Section 5.3.3) and the one including solar drying is shown in Figure A11.1.

Sludge pyrolysis, waste-specific, WW specific (Switzerland) sc5 solar drying

p

GaBi 4 process plan: Reference quantities
The names of the basic processes are shown.

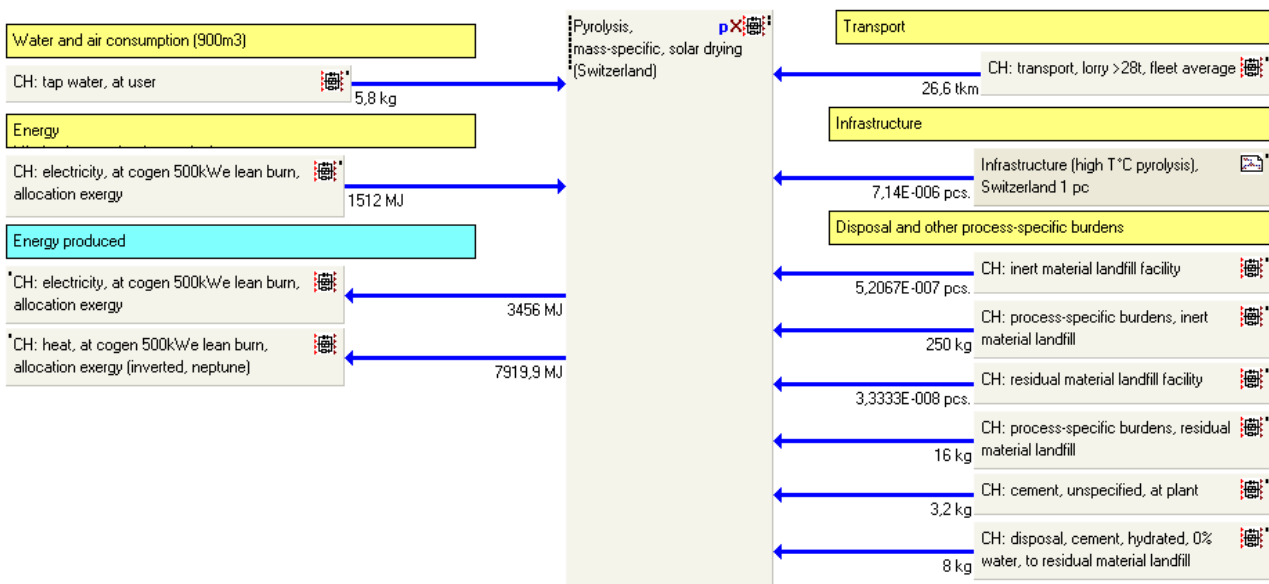


Figure A11.1: HTP model – solar drying (Miladinovic 2009b).

A11.5 Raw data for functional inventory (Miladinovic 2009b).**SLUDGE ULTRA HIGH TEMPERATRURE PYROLYSIS**

Composition of the digested sludge	<u>g/kg</u>
Nitrogen (total)	97
Total phosphorus (P)	28
Total organic matter	620

Heavy Metals	<u>µg/g</u>
Cadmium	4
Chromium	31,3
Copper	169
Nickel	17,7
Mercury	
Zink	610
Molybdenum	
Lead	40,5
Cobalt	
AOX	

Total amount of treated sludge per year	7000tDM/year
Electrical energy consumption	Per tDM 320kWh
Oxygen consumption	0
Energy gain (standard gas engine)	960kWh
Water consumption	
<u>gas washer</u>	<u>20 ltr/day</u>
Total	

Solid residue produced during gas cleaning	0kg
Gas release to the atmosphere	0

Inert solid residue which comes out of the reactor 250 kg/t

<u>Composition</u>	<u>mg/kg</u>
Cd	11,5
Pb	80
Cu	474
Ni	87

Zn	235
Cr	111
Mn	862
PAH's (total)	0,34
<u>Possibilities for application</u>	
land filling (Ptot.)	6-9%
road construction	
fertilizer	
Solid residue in the gas cooling pipes	8kg/t
<u>Composition</u>	<u>mg/kg</u>
Cd	9,3
Pb	450
Cu	400
Ni	300
Zn	7500
Cr	500
Mn	300
PAH's (total)	490
Chemicals consumption	m ³ /t
	0
Construction inventory list; amount of:	
concrete (m3)	50
stainless steel (kg)	7000
steel (kg)	25000
copper (kg)	1500
aluminum (kg)	2500
any other material important for the environment	
approximate life time of the plant (years)	20
Syngas composition (not released, used for electricity production)	
	mg/Nm ³
Chlorine (as HCl)	1
Fluor (as HF)	0,03
SO ₂	20
NO ₂	135
Hg	0,002
Cd, Tl	0,002
Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V, Sn (total)	0,07

Appendix 12: Physical and functional inventory for middle temperature pyrolysis (gasification) of sludge

A12.1 Description of the process

The gasification process transforms the sludge into gaseous compounds regarding the organic molecules whereas the inorganic constituents remain as solid residues. It creates liquid residue too. The digested sludge is mechanically dewatered the same way as for the incineration scenario (from 4%DM → ~30%DM). It is then as for HTP dried (from 30%DM → 70-85% DM) with either heat drying or solar drying.

The sketch of the gasification process is shown in Figure A11.1. For further details about the process, please refer to Deliverable 1.3 (Bagnuolo et al. 2009) and the description of the patented Kopf gasification process (Miladinovic 2009c).

The inventory data for the gasification scenario is based on data from NEPTUNE partners Eawag (described at the end of this appendix, A12.5) including the description of the patented Kopf gasification process (Miladinovic 2009c).

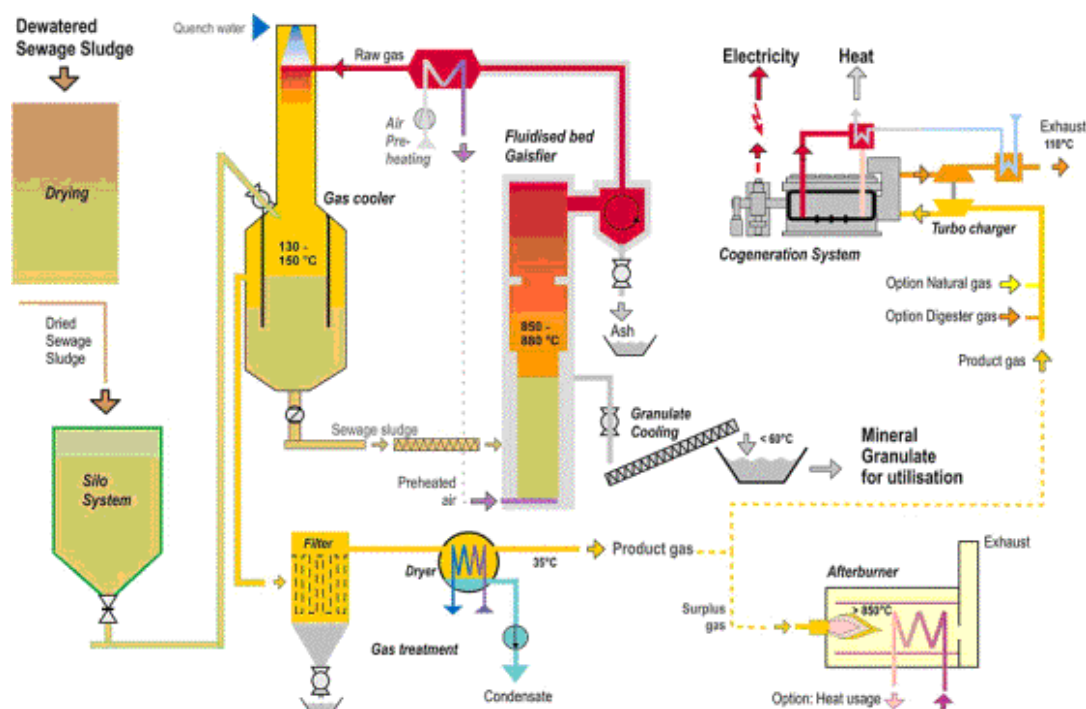


Figure A12.1: Gasification process Kopf Balingen (Miladinovic 2009c)

A12.2 Mass balance

	Flow	Quantity	Comment	Notation
Input	Sludge	1 tDM	After drying (70-85 % DM)	SLU
Outputs	Mineral granulate	500 kg	Produced in gasification reactor	GRA
	Solid residue	1 kg	Produced in gas filter	RES
	Condensate after gas	350 L	Returned to WWTP	WAT
	Off-gas	1 765 Nm ³	From energetical gas utilisation unit	AIR

Table A12.1: Flows for gasification

The establishment of the mass balance for the process gasification is necessary to obtain transfer coefficients. For each compound present in the sludge, transfer coefficients will determine how many percent a given compound in the sludge goes the outputs: mineral granulate, solid residue, condensate and off-gas.

The complete mass balance cannot be established for a compound if it is not tracked in more than 1 flow. Because the composition of the solid residue produced in gas filter is unknown, the mass balance can be established only for the compounds that have been measured in the other 4 flows.

Sludge: The sludge composition is partially known (14% in weight). The rest of the composition will be estimated from typical sludge composition from anaerobic digestion when used for the modelling.

Mineral granulate: Its composition is known only for trace compounds (less than 1% in weight). For example, the concentration of the heavy metals As, Pb, Cd, Cr, Cu, Ni, Hg, Tl, Zn is available and those concentrations are among the criteria for landfill disposal of hazardous or non hazardous waste. Moreover, from the Kopf documentation (Miladinovic 2009c), it is said that the mineral granulate is « inert, non-leachable, grainy and dry », which means that it could be disposed of in an inert material landfill (Switzerland, Doka (2007a) or landfill of EU Category 3 (EU 2005).

Solid residue: The composition of the solid residue from the gas quencher and filter is unknown. It represents a small fraction of the total flows (1 kg per t DM) and contains 2% organic matter. This is an important information for its disposal because the EU and Swiss landfill criteria uses 5% organic carbon limit values to consider a waste as inorganic. Considering that this residue comes from gas treatment and might be hazardous, it will be disposed of in a residual material landfill (Switzerland) or a landfill of Category 1 (EU) for « stable non reactive hazardous waste, inorganic or mineral composition ». Therefore, the composition of this residue has not to be estimated in more details. However, it might contain a non-negligible part of the heavy metals present in the sludge.

Condensate: The composition of the condensate is known for the compounds AOX, As, Pb, Cd, Cr, Cu, Ni, Hg, Ag, Zn (given in mg/L). It will be returned to the wastewater treatment plant to be treated. The treatment of this residue in a conventionnal wastewater treatment plant (2nd iteration, as for the sludge incineration process) will be modelled.

Off-gas: The composition of the off-gas from the energetical gas utilisation unit is known for the

compounds As, Pb, Cd, Cr, Cu, Ni, Hg, Sb, Co, Mn, V and organic compounds as C. Those compounds will be emitted directly to air, through the process « Gasification » in the modelling, which includes the cogeneration system in it.

A12.3 Functional inventory: Transfer coefficients

Not including the gas filter residue transfer coefficients for eight metals can be estimated, see Table A12.2.

	Air	Granulate	WWTP
As	0,20%	99,44%	0,36%
Cd	0,01%	98,89%	1,11%
Cr	0,11%	99,84%	0,05%
Cu	0,01%	99,91%	0,07%
Hg	34,41%	63,52%	2,07%
Ni	0,13%	99,75%	0,12%
Pb	0,04%	99,68%	0,28%
Zn	0,00%	99,87%	0,13%

Table A12.2: Transfer coefficients for gasification excluding gas filter residue

If the data on Cd, Hg and Pb for the gas filter residue is taken into account the transfer coefficients in table A12.3 result.

	Air	Gas filter residue	Granulate	WWTP
As	0,20%		99,44%	0,36%
Cd	0,00%	84,20%	15,63%	0,18%
Cr	0,11%		99,84%	0,05%
Cu	0,01%		99,91%	0,07%
Hg	0,15%	99,55%	0,28%	0,01%
Ni	0,13%		99,75%	0,12%
Pb	0,01%	79,10%	20,83%	0,06%
Zn	0,00%		99,87%	0,13%

missing flows are in the gas filter

Table A12.3: Transfer coefficients for gasification including gas filter residue

The transfer coefficients in Table A12.3 is used in the modelling

A12.4 Physical inventory

Infrastructure: The modelling of the gasification plant has been done with the data from the construction inventory list. It is important to notice that the amount of aluminium and copper in the gasification plant has been estimated from the high temperature pyrolysis plant, where the amount of aluminium and copper was available (2500 kg of Al and 1500 kg of Cu). Because the capacity of the high temperature pyrolysis plant is 7000 tDM per year and the one of the gasification plant is 1500 tDM per year, the amount of aluminium and copper needed for the gasification plant is 20% of the amounts for the high temperature pyrolysis plant.

Construction inventory list	
expected life time of the plant	15 years
capacity	170 kg DM / h
concrete (kg)	50 000
steel (kg)	70 000
Aluminium (kg)	536 *
Copper (kg)	321 *
* estimated from high temperature pyrolysis infrastructure	

tion

A sub-plan called « Infrastructure (gasification), Balingen » has been created and is shown in Figure 5.16 (Section 5.3.4) . It includes the production of all the materials in the inventory list and their corresponding disposal.

The recycling of infrastructure material is:

- « CH: Gravel, unspecified, at mine (inverted) » (80% of concrete)
- « FER: Iron scrap, at plant (inverted) » (80% of steel)
- « RER: copper, at regional storage (neptune, inverted) » (90% of copper)

The rest of the infrastructure is disposed of as:

- « CH: disposal, building, reinforced concrete, to sorting plant » (20% of concrete)
- « CH: disposal, building, reinforced steel, to sorting plant » (20% of steel)
- « CH: disposal, aluminium, 0% water, to sanitary landfill » (100% of aluminium)
- « CH: disposal, copper, 0% water, to municipal incineration » (10% of copper)

One piece of infrastructure (the gasification plant here) will be used to treat a large amount of sludge. Consequently, in order to relate the « quantity » of infrastructure needed for the treatment of 1 t DM, the expected lifetime of the plant and its capacity are needed.

The expected lifetime is 15 years and the capacity of the plant is 170 kgDM/h, which if assuming that the plant is working all year long, gives a capacity of 1500 t DM year. Consequently, $4.4E-5$ « pieces of infrastructure » are needed per t DM of sludge treated by gasification.

Disposal of solid waste

Inert material landfill facility: The inert material landfill facility modelled in Switzerland has a capacity of 450 000 m³, and is used to landfill 500 kg/tDM of granulate residue with a density of 1067 kg/m³. The number of pieces of infrastructure per tDM is therefore $1.04E-6$. The process-specific burdens associated are related to the amount of waste treated (500kg/tDM).

- « CH: inert material landfill facility »
- « CH: process-specific burdens, inert material landfill »

Residual material landfill facility: The residual material landfill facility modelled in Switzerland has a capacity of 300 000 m³ and an average waste density of 1600 kg/m³. The residue from gas treatment is landfilled with solidifying cement in a proportion waste-cement-water of 50-20-30 % Doka (2007a). Thus, 2kg of waste/tDM (1kg of residue + 1 kg of cement/water) and its density is assumed to be 1600 kg/m³ too. The number of pieces of infrastructure per tDM is: $4.17E-9$. The process-specific burdens associated are related to the amount of waste treated (2kg/tDM).

- « CH: residual material landfill facility »
- « CH: process-specific burdens, residual material landfill »
- « CH: cement, unspecified, at plant »
- « CH: disposal, cement, hydrated, 0% water, to residual material landfill »

Chemicals: There is no direct chemical consumption for the gasification process (at least none mentioned in the inventory list).

Air consumption: 900 m³ of air is consumed in the gasification reactor per t DM of sludge treated.

Water consumption: 300L of tap water is used for gas cleaning per t DM of sludge treated.

Transport: The mineral residue is transported to Swiss landfill. There are two types of transport average lorry and distance:

Transport	Cargo	Distance	Quantity
« CH: transport, lorry >28t, fleet average »	Inert residue: 500 kg	100 km	50 tkm
« CH: transport, lorry >28t, fleet average »	Solid residue: 1 kg	100 km	0,1 tkm

Table A12.5: Transport inventory list

Energy balance:

The input to the gasification process is dried sludge with 70 to 85% dry matter. In Balingen plant, the drying is made with solar energy with photoelectric cells. From a commercial report (Parkson 2009), the additional electricity consumption for this drying is estimated to be 80kWh/tDM. However, if no solar drying is available, the heat produced in the process (1000kWh/tDM) is assumed to be internally used as part of the energy use for drying the dewatered sludge. The consistency of this hypothesis is validated by the thermodynamic (theoretical) energy needed to evaporate the amount of water needed to reach a 80% dried sludge.

Options for total energy consumption (including sludge drying):

Solar drying leads to an electricity consumption of 80kWh/tDM. The heat produced in the process is used externally.

When heat drying is used all the heat produced in the process is used internally. Because the energy needed is higher (based on thermodynamic calculation) additional energy is used (1440-1000=440kWh/tDM) to finish to drying of the sludge. Oil is chosen as external fuel (such as in incineration) because using oil is a common way of supplying energy for heat drying (Metcalf & Eddy 2004). The heat value of fuel oil is 43 MJ/kg [PE-GaBi 4.3 2006] and because 440 kWh are needed, it makes 36.8 kg of fuel oil.

The total electricity consumption of gasification and dewatering is 120 kWh. If heat drying is used, there is fuel consumption, whereas if solar drying is used, there is an additional electricity consumption of 80kWh/tDM.

Regarding the electricity consumption for dewatering it is as for sludge incineration based on the estimated about 1 kWh/ ton water released.

The two options (heat or solar drying) will be modelled in GaBi.

Option	Process	Electricity consumption (kWh/tDM)		Heat production (kWh/tDM)	Oil consumption (kWh/tDM)
Solar	Dewatering	-20	-200	+ 1000 Used externally	None
	Solar drying	-80			
	Gasification	-100			
Heat	Dewatering	-20	-120	None - Used internally	-440
	Heat drying	None			
	Gasification	-100			

Table A12.6 : Electricity consumption for gasification (solar or heat drying)

The processes used in GaBi 4.3 are:

- « CH: electricity, at cogen 500kWe lean burn, allocation exergy »
- « CH: heat, at cogen 500kWe lean burn, allocation exergy »

« CH: heat, at cogen 500kWe lean burn, allocation exergy (inverted, neptune) »
 « CH: light fuel oil, at regional storage »

Lean burn is used as default in NEPTUNE.

Energy balance

<i>Gasification</i>	Electricity flows (kWh/tDM)	
	Heat drying	Solar drying*
Electricity consumption	- 120	- 200
Electricity production	+ 500	+ 500
Electricity balance	+ 380	+ 300

Table A12.7: Electricity balance for gasification (solar or heat drying)

<i>Gasification</i>	Heat flows (kWh/tDM)	
	Heat drying	Solar drying*
Heat production	+ 1 000	+ 1 000
Heat consumption as fuel/heat	- 1 440	0
Heat balance	- 440	+ 1 000

* The solar drying scenario is the real-case situation used at the plant in Balingen.

Table A12.8: Heat balance for gasification (solar or heat drying)

The full model including heat drying is shown in Figure 5.18 (Section 5.3.4) and the one including solar drying is shown in Figure A12.2.

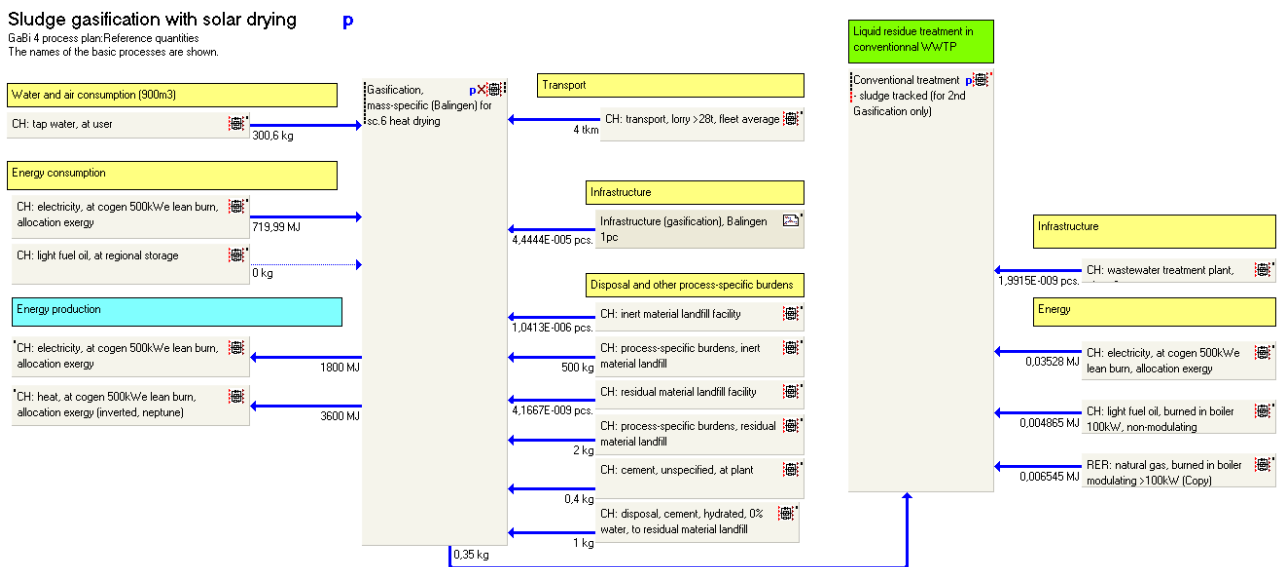


Figure A12.2: Gasification model in GaBi 4.3, solar drying

A12.5 Raw data for functional inventory (Miladinovic 2009a).

Sludge gasification plant, Balingen

Composition of the digested sludge	<u><i>g/kg</i></u>
Nitrogen (total)	
Phosphorus (P ₂ O ₅)	60-70
Potassium as K ₂ O	8-120
Medicinal drugs and antibiotics	0-1
Flame retardants	0.1-0.5
Polychlorinated dioxines/furanes	6-1500
AOX	100-300
Cadmium	1-3
Chloride	500-3000
Sulphur	5-13
Mercury	0.1-7.5
Lead	60-180

Total ammount of treated sludge per year

Capacity	170kgDM/h	Per tDM
----------	-----------	---------

Fuel consumption
Natural gas

Electrical energy consumption

External supply	0kWh
Internal supply (self-production)	100kWh
Total	100kWh

Electrical energy production 500kWh

Heat production 1000kWh

Water consumption
For gas cooling 300L

Mineral granulate produces in gasification reactor 500kg

<u>Composition</u>	<u><i>mg/kg</i></u>
Extractable liophilic compounds	74
Total hydrocarbons	1-6
Benzene	<0.1
Toluene	<0.1
Meta-para-Xylol	<0.1
Orto-xylol	<0.1
Ethyl-benzene	<0.1
Naphthalene	0,07
other PAH according to EPA	<0.01
TOC	1400

<u>heavy metals</u>	
Arsenic	5.85-11.4
Lead	42.5-57.5
Cadmium	0.25-1
Chromium, total	72.5-198
Copper	350-1330
Nikel	32.5-80
Mercury	0.013-0.03
Thallium	0.50-0.75
Zinc	695-1180
Cyanides, total	0.09-0.4
Polychlorinated Dioxins and Furanes	0,05

	<u>µg/kg</u>
Dichlormethane	<100
1,1,1-trichlorethan	<10
trichlorethen	<10
tetrachlorethen	<10
tetrachlormethan	<10
1,2-cis-dichlorethene	<200

Air consumption

Gasification reactor	900m ³
Total	900m ³

Solid residue produced in gas filter

1kg

Composition

2% organic matter

Condensate after gas drying (returned to WWTP)

350L

Composition

	<u>mg/L</u>
adsorbable organic halogen compounds (AOX)	1,5
Arsenic	0,045
Lead	0,2
Cadmium	<0.01
Chromium	<0.1
Copper	0,9
Nikel	<0.1
Mercury	0,001
Silver	0,9
Zinc	1,7

Chemicals consumption

Off gas from energetical gas utilisation unit

1 765m³
(mg/Nm³)

NO ₂	
CO	
SO ₂	
O ₂	
HCl	
HF	
NH ₃	
Pb	0,0055
Cr	0,043
Cu	0,024
Ni	0,02
As	0,0048
Sb	<0.00001
Co	0,034
Mn	0,048
V	0,01
Sn	<0.00001
Cd	<0.00001
Hg	0,0033
Organic compounds as C	<3

Construction inventory list; amount of:

expected life time of the plant	15 years
concrete (kg)	50.000
steel (kg)	70.000
Aluminium (kg)	2500
Copper (kg)	1500

Appendix 13: Transfer coefficients for mercury to air regarding sludge inertization methods.

Sensitivity analysis on mercury emissions to air

The impact categories “Human Toxicity Water” and “Human Toxicity Soil” have dominating important impact potentials regarding the different sludge inertization processes investigated due mainly to emissions of mercury to air. Because it is a sensible parameter, its influence is important to take into account, particularly regarding the way those emissions have been estimated.

For each sludge inertization process, the mass balance has been estimated from the inventory list (Appendix 9-12), where, ideally, the composition of the sludge and of the different outputs (air, solid residue, liquid residue) was known, especially for the 7 heavy metals investigated. However, not only a particular transfer coefficient can be found but a variation range, using the inventory list as well as the European average on mixed sludge composition, as shown in Table 5.12B in Section 5.2 (shown without ranges).

The mercury content of the sludge and of the air emissions is known for the incineration process (see Appendix 9). Moreover, the mercury input is 1.06 mg-Hg/kgDM (which is within the European range of 0.1-1.7 mg-Hg/kgDM based on Weemaes (2008), see Section 5.5). If the typical (average) value of 0.6 mg-Hg/kgDM (see Table 5.12B) was taken, keeping the same air emissions, the transfer coefficient of mercury to air would be increased to 68%. In that case, the environmental impact profile of on-site incineration worsen, which increases the gap between this process and the others.

For the process high temperature pyrolysis, there was no value for mercury content of the sludge but there was for air emissions. It was chosen to take the European average value of 0.6 mg-Hg/kgDM, and to estimate the range of transfer coefficients with the European range.

The mercury content of the sludge was given as a range for the gasification process, therefore, the average was taken for the estimation of the transfer coefficients. The extreme values of the transfer coefficients are estimated in the same way as for HTP.

For wet oxidation, all values were given and worst and best case scenarii are estimated too for mercury emissions to air.

The summary of the processes investigated for the sensitivity analysis is shown in table A13.1

	Mercury in sludge		Mercury emitted to air	Transfer coefficient of mercury to air	Variation of transfer coefficient
	mg/kgDM		mg/kgDM		
On-site incineration (sc.1)	1,06			38,2%	
worst case scenario	0,60	*	0,405	67,5%	24%-68%
best case scenario	1,70			23,8%	
Wet oxidation (sc.2)	0,76		0,006	2,2%	
worst case scenario	0,10	*	0,012	11,9%	0,5 - 12%
best case scenario	1,70		0,006	0,4%	
High temperature pyrolysis (sc.3-5)	0,60	*		3,30%	
worst case scenario	0,10		0,020	20,0%	1 - 20%
best case scenario	1,70			1,2%	
Gasification (sc.4-6)	3,70			0,16%	
worst case scenario	0,10		0,006	5,8%	0,1 - 6%
average scenario	0,60	*		1,0%	

* = using the European average 0,6mg-Hg/kgDM

Range 0,1 – 1,7 mg-Hg/kgDM

Table A13.1: Sensitivity analysis of mercury emissions to air

Implementing these ranges in the impact profiles for the sludge inertization methods does not change the overall picture, i.e. that on-site incineration is the less environmentally sustainable inertization method among those compared here. As the transfer coefficient is considered most uncertain for HTP it may be mentioned that substituting the 3.3% transfer coefficient used for HTP with 20% (max) “only” increases the net environmental impact by a factor 3.5.

Appendix 14: Physical and functional inventory for sludge triage

A14.1 Description of the process

Sludge triage consists in treating differently primary and secondary sludge, instead of treating them together as mixed sludge. Secondary sludge is assumed to be richer in nutrients than primary sludge (Deliverable 1.3: Bagnuolo et al. 2009). Thus, application of treated secondary sludge on agricultural land may be more environmentally profitable.

The reference scenario A1 (4A) is the treatment of mixed sludge by incineration (sludge incineration or co-incineration). The alternative processes are the treatment of primary sludge by incineration, and the treatment of secondary sludge by:

- B (4B): short aerobic thermophilic treatment with intermittent feed (5 days; 45°C)
- C (4C): ultrasound disintegration + anaerobic digestion
- D (4D): thermal disintegration + anaerobic digestion

The sketches of the sludge triage scenarios are shown below in Section 5.4 (Figure 6.16 and Figure 5.17). The flow names used below (e.g. Flow G) is the ones in the Minini flow sheets included in Deliverable 1.3 (Bagnuolo et al. 2009, Figure 7.1 and Figure 7.2) and shown below as Figure A14.1 and A14.2

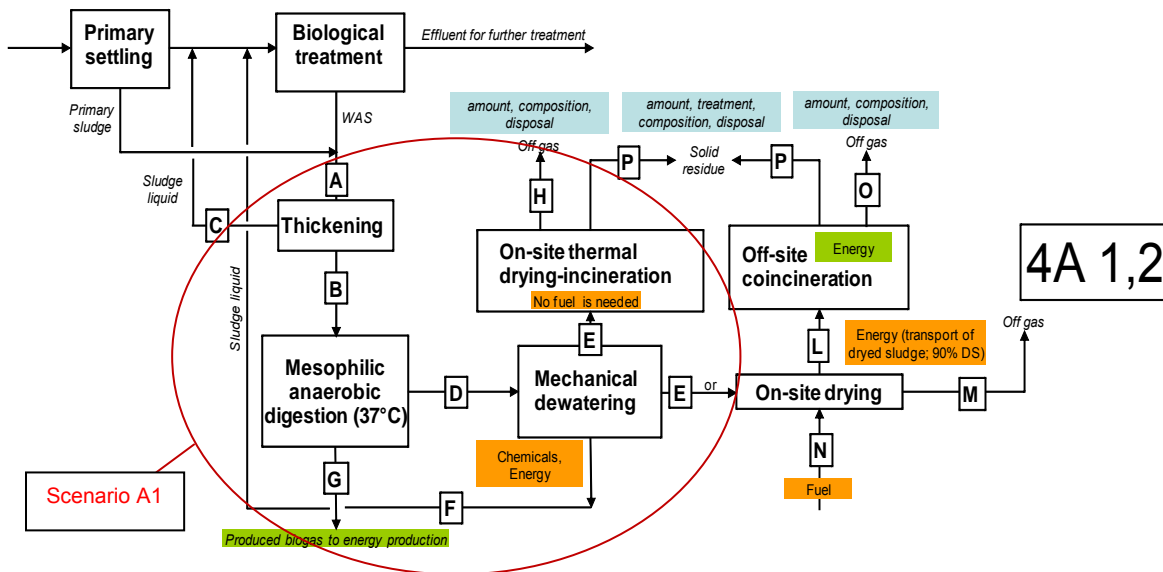


Figure A14.1: Conventional sludge treatment – no triage (Bagnuolo et al. 2009)

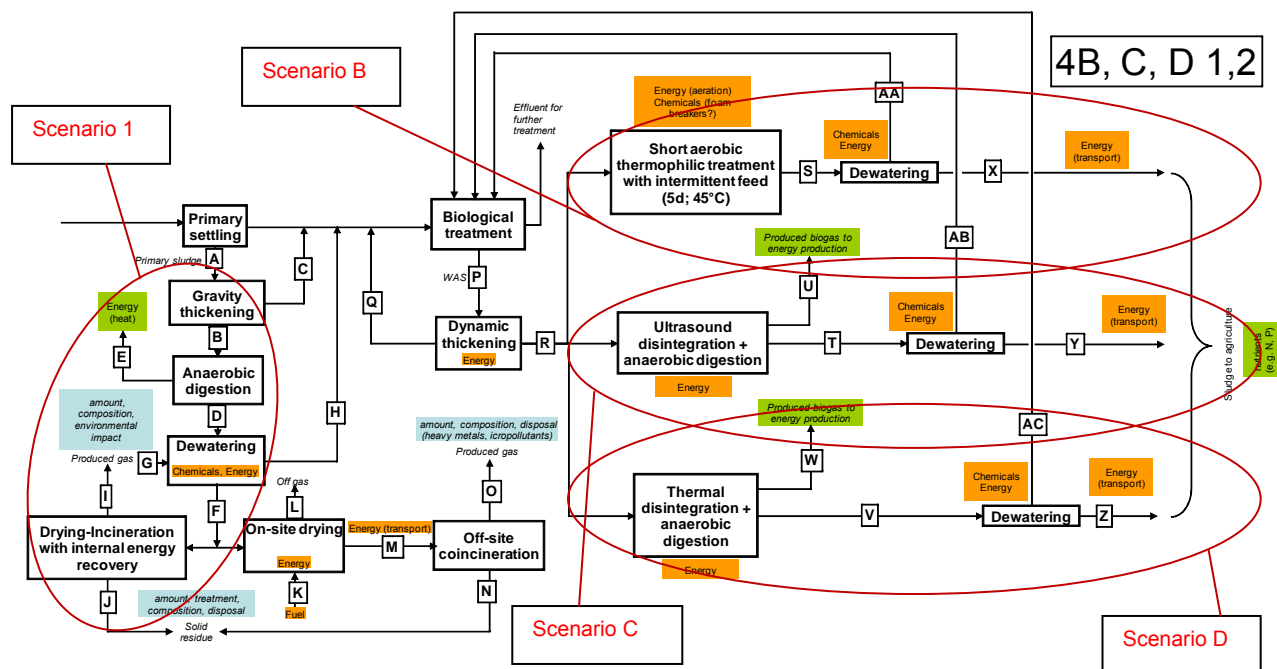


Figure A14.2: Sludge triage (Bagnuolo et al. 2009)

A14.2 Mass balance

Full inventory lists are not available for the 3 processes of secondary treatment, only data on energy production, energy consumption and nutrient content of the products but no data on e.g. heavy metals. The heavy metals in secondary sludge (Table 5.12A in Section 5.5) will therefore all be assumed to end on soil when treated secondary sludge is applied on land (modified model based on model by Doka (2007b) in a simplified version developed by Munoz et al. (2007)) whereas the heavy metals of the primary sludge (Table 5.12A in Section 5.5) will be modelled by the on-site incineration (see Appendix 9).

The mass balance for the reference scenario sludge incineration has already been done for the comparison of the sludge inertization processes (Appendix 9).

For the sludge triage scenario, the mass balance is divided into two flows for primary and secondary sludge. First of all, the ratio between primary and secondary sludge is needed in order to always compare the treatment of 1tDM of total sludge. The results from Minnini mass balances (Deliverable 1.3: Bagnuolo et al. 2009, including excel sheet from Giuseppe Mininni) have a proportion of 56% of primary sludge and 44% of secondary sludge in weight (DM basis). Other calculated proportions are shown in table A13.1.

Furthermore, it has been observed that in the mass balances by Minini (Deliverable 1.3: Bagnuolo et al. 2009, including excel sheet from Giuseppe Mininni) total P means really total P (and ortho P is included) whereas, tot N means only N without N-NH4 when it has been measured. This fact has been checked with the mass balance.

Proportion of primary and secondary sludge

Primary sludge/Mixed sludge (dry solid basis)	56,41%
Secondary sludge/Mixed sludge (dry solid basis)	43,59%
Primary sludge/Mixed sludge (VS basis)	58,24%
Secondary sludge/Mixed sludge (VS basis)	41,76%
Primary sludge/Mixed sludge (volume basis)	24,45%
Secondary sludge/Mixed sludge (volume basis)	75,55%

Relative % DM content compare to mixed sludge	For 4% DM original mixed sludge
%DM primary = 2,24 * %DM mixed	Primary = 9,0 %DM
%DM secondary = 0,58 * %DM mixed	Secondary = 2,2 %DM

Table A14.1: Calculated proportions based on data from Bagnuolo et al. (2009), including excel sheet from Giuseppe Mininni

The following scenarios will be LCA modelled by use of the GaBi tool:

1. Mixed sludge treated by anaerobic digestion followed by on-site incineration (scenario A1)
2. Primary sludge (56% of DM) treated by anaerobic digestion followed by on-site incineration combined with secondary sludge (44% of DM) treated by aerobic thermophilic treatment followed by final disposal on agricultural land (sludge triage scenario 1+B)
3. Primary sludge (56% of DM) treated by anaerobic digestion followed by on-site incineration combined with secondary sludge (44% of DM) treated by ultrasound disintegration followed by final disposal on agricultural land (sludge triage scenario 1+C)
4. Primary sludge (56% of DM) treated by anaerobic digestion followed by on-site incineration combined with secondary sludge (44% of DM) treated by thermophilic disintegration followed by final disposal on agricultural land (sludge triage scenario 1+D)
5. Mixed sludge treated by anaerobic digestion followed by disposal on agricultural land (bonus scenario)

A14.3 Functional inventory

In all scenarios the partitioning of 56% dry matter primary sludge and 44% dry matter secondary sludge are used. Furthermore, the heavy metal concentrations in the different sludge types are based on the figures in Table 5.12A (Section 5.5) and the naming of the flows refer to the flow names in Figure A13.1 and Figure A13.2.

Regarding substitution of fertilizer by spreading the treated sludge on agricultural land tot-P and tot-N are used in the balances.

Mixed sludge (scenario A1 – no triage)

Flow A in scenario 4A: Mixed sludge before thickening (kg/m³WW):

Sludge total	Water	Dry matter	Dry matter (%)
16.24	15.95	0.29	1.80%

Flow G in scenario 4A (Figure A14.1): Biogas from digestion of mixed sludge (Nm³/m³WW)

	Biogas	CH ₄	CO ₂	H ₂ O
Amount [L/m ³ WW]	61.7	33.5	23.8	4.4
Amount [Nm ³ /tDM]	212.8	115.5	82.1	15.2
Amount [kg/tDM]	235	76		
Energy [kWh/tDM]	1500	930		

Biogas: M_{biogas}=26.9g/mol ; Heat value: 23MJ/kg [PE-GaBi 4.3 2006] natural gas 44MJ/kg [PE-GaBi 4.3 2006]

Assuming that the biogas is burned in a co-generator with a yield of 1/3 electricity and 2/3 heat, the energy produced is as follows:

Heat (kWh/tDM)	Electricity (kWh/tDM)
1000	500

Data on energy consumption is missing in the documentation by Bagnuolo et al. (2009, including excel sheet from Giuseppe Mininni). However, in the Granit report (on WO and incineration but including digestion separately) (Miladinovic 2009a) similar energy yields as those by Giuseppe Mininni are reported and supplemented by energy consumptions. These consumptions are therefore used here:

	kWh/tDM	
Electricity consumption	-70	
Heat consumption	-320	40Nm ³ of natural gas 26,3 kg of natural gas
Electricity production (1/3 at cogeneration)	520	230Nm ³ of biogas (65%CH ₄)
Heat production (2/3 at cogeneration)	1040	230Nm ³ of biogas (65%CH ₄)
Total	1170	

Biogas: 65%CH₄; 35%CO₂ , M_{biogas}=25,8g/mol ; Heat value: 23MJ/kg [PE-GaBi 4.3 2006]

The following energy data and associated EcoInvent processes are therefore used in the modelling of anaerobic digestion:

Electricity consumed	70 kWh/tDM	CH: electricity, at cogen 500kWe lean burn, allocation exergy
Gas consumed	26,3 kg gas/tDM	CH: natural gas, from low pressure network (<0.1 bar), at service station
Electricity produced	500 kWh/tDM	CH: electricity, at cogen 500kWe lean burn, allocation exergy (inverted, triage)
Heat produced	1000 kWh/tDM	CH: heat, at cogen 500kWe lean burn, allocation exergy (inverted, triage)

On-site incineration is modelled as described in Appendix 9.

The GaBi plan showing the physical inventory of mixed sludge digested followed by on-site incineration is shown in Figure 5.21 in Section 5.4

Direct disposal on agricultural land of mixed sludge after digestion is included as a bonus scenario.

Primary sludge – digested and on-site incinerated (scenario 1 – part of triage)

Flow A (Figure A14.2): Primary sludge before thickening (kg/m³WW)

Sludge total	Water	Dry matter	Dry matter (%)
3.97	3.81	0.16	4.00%

Flow E (Figure A14.2): Biogas from primary sludge digestion (L/m³WW)

	Biogas	CH ₄	CO ₂	H ₂ O
Amount [L/m ³ WW]	49.5	25.6	20.8	3.1
Amount [Nm ³ /tDM]	170.7	88.3	71.7	10.7
Amount [kg/tDM]	195.5	58		
Energy [kWh/tDM]	1250	-710		

Biogas: $M_{\text{biogas}}=27,9\text{g/mol}$; Heat value: 23MJ/kg [PE-GaBi 4.3 2006] natural gas: 44MJ/kg

Assuming that the biogas is burned in a co-generator with a yield of 1/3 electricity and 2/3 heat, the energy produced is as follows:

Heat production (kWh/tDM)	Electricity production (kWh/tDM)
833	417

Secondary sludge (part of scenario B, C and D)

Flow P (Figure A14.2): Secondary sludge before thickening (kg/m³WW)

Sludge total	Water	Dry matter	Dry matter (%)
12.27	12.15	0.12	1.00%

In total (primary + secondary) there is 0.28 kgDM/m³WW.

Scenario B: Aerobic thermophilic treatment (no mesophilic anaerobic digestion after)

Flow X (Figure A14.2): Dewatered secondary sludge after thermophilic aerobic treatment

Total – wet sludge	Dry Matter	Water
	28%	72%
1.57 t	0.44 t	1.13 t

The mass of sludge to transport and apply on land is thus 1.57t.

The substitution of fertilizers is estimated with the N and P content of the sludge [in kg/kgDM]:

Total N	N (w/o NH ₄)	N-NH ₄	Total P	ortho-P
1.63E-2	1.57E-2	6.96E-4	1.03E-2	0.00E+0

P₂O₅ substituted: 2.36E-2 kg/kgDM

RER: ammonium nitrate phosphate, as P₂O₅, at regional storehouse

N substituted: 1.63E-2 kg/kgDM

RER: ammonium nitrate phosphate, as N, at regional storehouse

There is no energy production from the secondary sludge treatment because there is no anaerobic digestion.

The GaBi plan showing the physical inventory of scenario B combined with scenario 1 (sludge triage scenario 1+B) is shown in Figure 5.22 in Section 5.4

Scenario C: Ultrasound treatment followed by anaerobic digestion

Flow Y (Figure A14.2): Dewatered secondary sludge after ultrasound disintegration and anaerobic digestion

Total – wet sludge	Dry Matter	Water
	26%	74%
1.69 t	0.44 t	1.25 t

The mass of sludge to transport and apply on land is thus 1.69t.

The substitution of fertilizers is estimated with the N and P content of the sludge (in kg/kgDM):

Total N	N (w/o NH ₄)	N-NH ₄	Total P	ortho-P
6.71E-3	5.30E-3	1.41E-3	4.14E-3	1.03E-3

P₂O₅ substituted: 9.39E-3 kg/kgDM

RER: ammonium nitrate phosphate, as P₂O₅, at regional storehouse

N substituted: 6.71E-3 kg/kgDM

RER: ammonium nitrate phosphate, as N, at regional storehouse

The energy production from the anaerobic digestion of the secondary sludge can be estimated from the biogas composition Flow U (Figure A14.2):

Biogas from anaerobic digestion of secondary sludge after ultrasound disintegration:

	Biogas	CH ₄	CO ₂	H ₂ O
Amount [L/m ³ WW]	22.1	13.4	7.2	1.5
Amount [Nm ³ /tDM]	79	47.9	25.7	5.4
Amount [kg/tDM]	82			
Energy [kWh/tDM]	525			

Biogas: $M_{\text{biogas}}=25.3\text{g/mol}$; Heat value: 23MJ/kg [PE-GaBi 4.3 2006]

Assuming that the biogas is burned in a co-generator with a yield of 1/3 electricity and 2/3 heat, the energy produced by the secondary sludge anaerobic digestion is as follows:

Electricity consumed	70 kWh/tDM	CH: electricity, at cogen 500kWe lean burn, allocation exergy
Gas consumed	26.3 kg gas/tDM	CH: natural gas, from low pressure network (<0.1 bar), at service station
Electricity produced	175 kWh/tDM	CH: electricity, at cogen 500kWe lean burn, allocation exergy (inverted, triage)
Heat produced	350 kWh/tDM	CH: heat, at cogen 500kWe lean burn, allocation exergy (inverted, triage)

Energy consumption for the ultrasound pre-treatment: 5.4 kWh/tDM (negligible). Deliverable 1.3; Mannheim plant (Bagnuolo et al. 2009).

Energy consumption of the ultrasound disintegration: 125 kWh/tDM (Batstone 2006)

Final energy consumption (as power) is therefore assumed to be 125 kWh/tDM.

The GaBi plan showing the physical inventory of scenario C combined with scenario 1 (sludge triage scenario 1+C) is shown in Figure 5.23 in Section 5.4

Scenario D: Thermal disintegration followed by anaerobic digestion

Flow Z (Figure A14.2): Dewatered secondary sludge after thermal disintegration and anaerobic digestion

Total – wet sludge	Dry Matter	Water
	25%	75%
1.76 t	0.44 t	1.32 t

The mass of sludge to transport and apply on land is thus 1.76t.

The substitution of fertilizers is estimated with the N and P content of the sludge (in kg/kgDM):

Total N	N (w/o NH ₄)	N-NH ₄	Total P	ortho-P
3.80E-3	2.16E-3	1.64E-3	4.14E-3	1.03E-3

P ₂ O ₅ substituted: 9.39E-3 kg/kgDM	RER: ammonium nitrate phosphate, as P ₂ O ₅ , at regional storehouse
N substituted: 3.80E-3 kg/kgDM	RER: ammonium nitrate phosphate, as N, at regional storehouse

The energy production from the anaerobic digestion of the secondary sludge can be estimated with the biogas composition Flow W: Biogas from anaerobic digestion of secondary sludge after thermal disintegration.

	Biogas	CH ₄	CO ₂	H ₂ O
Amount [L/m ³ WW]	26.7	17.3	8.8	1.7
Amount [Nm ³ /tDM]	100	61.8	31.4	6.1
Amount [kg/tDM]	88.9			
Energy [kWh/tDM]	640			

Biogas: M_{biogas}=22.7g/mol ; Heat value: 23MJ/kg [PE-GaBi 4.3 2006]

Assuming that the biogas is burned in a co-generator with a yield of 1/3 electricity and 2/3 heat, the energy produced by the secondary sludge anaerobic digestion is as follows:

Electricity consumed	70 kWh/tDM	CH: electricity, at cogen 500kWe lean burn, allocation exergy
Gas consumed	26.3 kg gas/tDM	CH: natural gas, from low pressure network (<0.1 bar), at service station
Electricity produced	210 kWh/tDM	CH: electricity, at cogen 500kWe lean burn, allocation exergy (inverted, triage)
Heat produced	430 kWh/tDM	CH: heat, at cogen 500kWe lean burn, allocation exergy (inverted, triage)

Based on data from (Batstone, 2006) energy consumption by thermophilic treatment is about 400 kWh/tDM.

Final energy consumption (as power) is therefore assumed to be 400 kWh/tDM.

The GaBi plan showing the physical inventory of scenario D combined with scenario 1 (sludge triage scenario 1+D) is shown in Figure 5.24 in Section 5.4

A14.4 Physical inventory

No infrastructure data on tanks, pipes etc. regarding aerobic thermophilic treatment, ultrasound disintegration, anaerobic digestion and thermal disintegration have been identified. Anyway, the infrastructure materials used for the different process steps compared are assumed not to be that different as having a significant impact on the results. As regards comparison to on-site incineration the infrastructure of the special processes are most probably negligible. Regarding the anaerobic digester (only excluded in scenario B) the favorising of all other special processes (scenario C and D) should be kept in mind when comparing environmental sustainability among the scenarios. For incineration the infrastructure (and the full model for the incineration part) is based on the model described in Section 5.3.1.

A14.5 Raw data for functional inventory

The NEPTUNE raw data for sludge triage are to be found Deliverable 1.3 (Bagnuolo et al. 2009, including excel sheet from Giuseppe Mininni) and in Batstone (2006).

Appendix 15: Sensitivity analysis on PAC and electricity production

In order to assess the uncertainties present in the LCA models or the reliability of the results it is necessary to carry out sensitivity analysis. This is mainly done because uncertainty sources in LCA are notoriously difficult to quantify in a meaningful way with traditional statistical methods (such as standard deviation) because of the magnitude of the uncertainties and the subjective character of them in some cases. Anyway, in most cases data are missing making it impossible to calculate uncertainty in a traditional statistical way.

Sensitivity analysis is carried out by varying in principle all parameters but typically only the assumed most significant parameters within their relevant range and observing the consequences on the overall result. Here, we will look at two significant parameters:

- Activated carbon
- Electricity

The production of pulverized activated carbon (PAC) plays a dominating role for the induced impact regarding PAC addition in cluster 1 and is the main reason for the inferior impact profile of PAC addition as compared to ozonation.

The production of electricity plays generally a significant role in the induced impacts of the technologies included in the NEPTUNE LCA assessments.

A15.1 Activated carbon

As shown in Figure 8.7 and Figure 8.8 (Section 8.1.3), the main contributor to induced impact in the PAC addition scenarios is the ancillary product, namely activated carbon.

During the NEPTUNE project it was originally assumed that the activated carbon was recuperated from the effluent and regenerated with a 10% loss. However, during the Varna NEPTUNE meeting, it was concluded that the used powdered activated carbon is difficult to collect and can't be regenerated because it can't withstand the heat during regeneration without combusting.

In NEPTUNE it has been chosen to model activated carbon regeneration/production according to the Muñoz 2 model (Muñoz (2006)). However, there are other sources giving different ways to model activated carbon production. These include:

- Muñoz 1 (included in Muñoz (2006) together with the Muñoz 2 model)
- CGTF: two scenarios: one foreground model and one aggregated in SimaPro (Siegrist 2008a)
- ProBas (two scenarios: one foreground model and one aggregated (ProBas 2008))

Therefore, to carry out the sensitivity analysis of activated carbon modeling, we will model the process based on the different sources listed above as well as taking into account recycling and lack of recycling of the activated carbon. Processes in which the AC is recycled will be referred to as GAC regeneration while processes in which no recycling occurs will be referred to as GAC production. It has not been possible to find useful data on PAC so data for granular activated carbon (GAC) is used as a substitute. The impact of this substitution on the results is assumed to be of minor importance as the main difference between the two processes is perceived to be the

pulverization of the carbon, a process expected to be insignificant compared to the others involved – especially activation.

The modeling of PAC production/regeneration has been done in GaBi based on the following Ecolnvent processes:

- Regeneration of 1 kg:
 - 3 MJ - RER: hard coal, burned in industrial furnace 1-10MW
 - 10.5 MJ - RER: natural gas, burned in industrial furnace >100kW
 - 0.6 kg - RER: steam, for chemical processes, at plant
 - 0.108 MJ - UCTE: electricity, natural gas, at power plant
 - 1.299 tkm - RER: transport, lorry >16t, fleet average
- Production of 100g:
 - 1.2 kg - CH: water, deionised, at plant
 - 0.576 MJ - UCTE: electricity, natural gas, at power plant
 - 1.32 MJ - RER: natural gas, burned in industrial furnace >100kW
 - 6.08 MJ - RER: hard coal, burned in industrial furnace 1-10MW
 - 0.1 kg - UCTE: hard coal mix, at regional storage

To begin with, the energy balance in absolute values for the different ways to model the process is presented below. These models are non-aggregated, meaning that they specify the inputs necessary to obtain a given quantity of activated carbon. Aggregated options resulting from modeling non-aggregated processes are also available and will be shown below in the accompanying figures. However, aggregated models are considered unreliable in this connection because they have been aggregated using a different methodology as the one used in this study and are most probably less comprehensive.

Table A15.1 The energy balance for GAC according to different models

Cases		Munoz 1	Munoz 2	CGTF	ProBas	
Energy balance For 1 kg GAC	Production	Natural gas (MJ)	196	13.2	31.13	4.1
		Steam (kg)	3			
		Electricity (MJ)	0.076	5.76	5.94	0.19
		Hard coal (kg)	2	3	3	1.35
		Other fuels (MJ)				0.46
	Regeneration	Natural gas (MJ)	108	10.5		
		Steam (kg)	0.3	0.6		
		Hard coal (kg)	1.1	1.1		
		Electricity (MJ)	0.001	0.03		
	No recycling	Primary energy MJ	266	118	137	46
	With recycling	Primary energy MJ	169	58		

As can be seen in table A15.1 the first three models are all within one order of magnitude from each other and are therefore considered the most reliable estimates. It has been chosen to ignore the ProBas case (last column) because it is based on an online database with very limited documentation.

PAC regeneration: To begin with, we will look at the different options available to model activated carbon regeneration (this implies AC recycling). These options are illustrated in Figure A15.1 and show the range of impact potentials generated by AC regeneration.

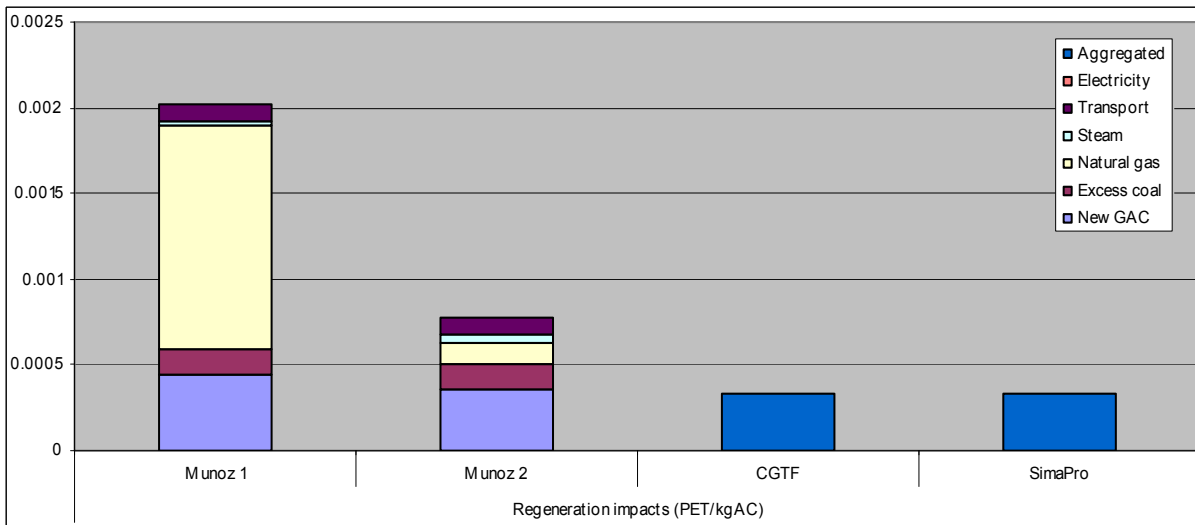


Figure A15.1 Sensitivity analysis: AC regeneration - showing the potential impact from 1 kg activated carbon including regeneration

Figure A15.2 below shows the different profiles that would be obtained for the case of the addition of 20g/m³ of PAC to biology. This specific PAC dose is displayed because it is the PAC case in which induced and avoided impacts are closest in absolute value, hence most likely to be affected by changes in modeling.

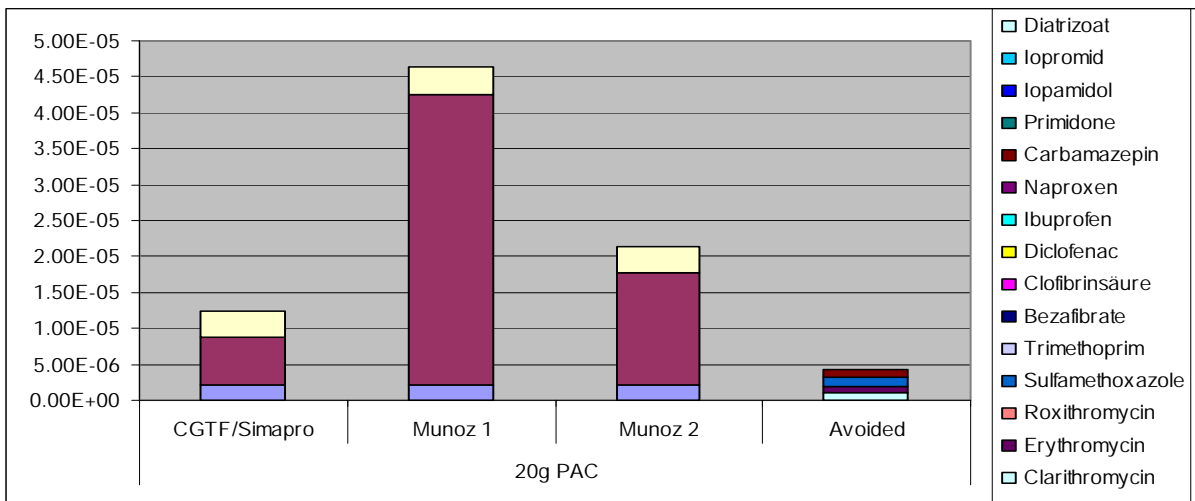


Figure A15.2 Sensitivity analysis: AC regeneration – showing the effect of using different models on the induced part (CGTF/SimaPro, Munoz 1 and Munoz 2) as compared to the avoided part for PAC dose 20 g/m³ (unit on y-axis PET/m³ and for the bars on induced impacts blue is infrastructure, red is ancillary (PAC) and yellow is energy as in Section 8.1.3)

As we can see in figure A15.2, the range of values changes rather significantly from roughly half to more than twice the initially value (Munoz 2). PAC regeneration is therefore perceived as being highly variable depending on the modeling scenario. This implies that the uncertainty is relatively large but no matter the model the induced impact is higher than the avoided.

Fortunately, it has been established that no recycling of the AC is possible. Therefore, we should focus on the case in which virgin AC is used.

PAC production: As just mentioned, modeling AC with no recycling is the most correct way regarding the NEPTUNE cases. The range of potential impacts of the process resulting from different modeling scenarios is presented in Figure A15.3.

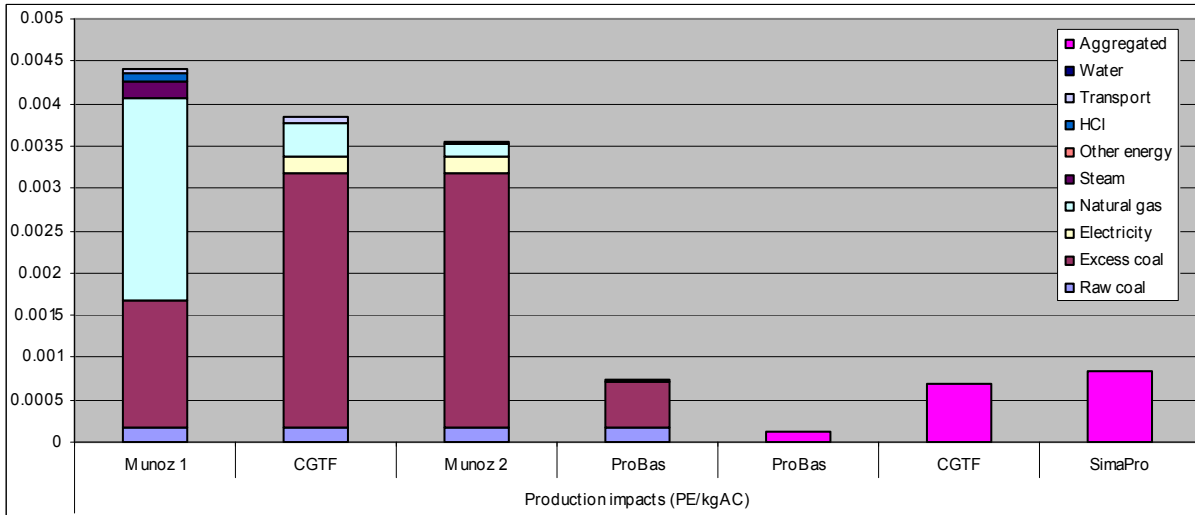


Figure A15.3 Sensitivity analysis: AC production - showing the potential impact from 1 kg activated carbon (no regeneration)

Figure A15.3 confirms that the first three modeling scenarios are rather consistent as displayed in the energy balance. The other scenarios are once again considered unreliable due to insufficient documentation and aggregation. Figure A15.4 below shows the range obtained in the case of PAC addition to biology.

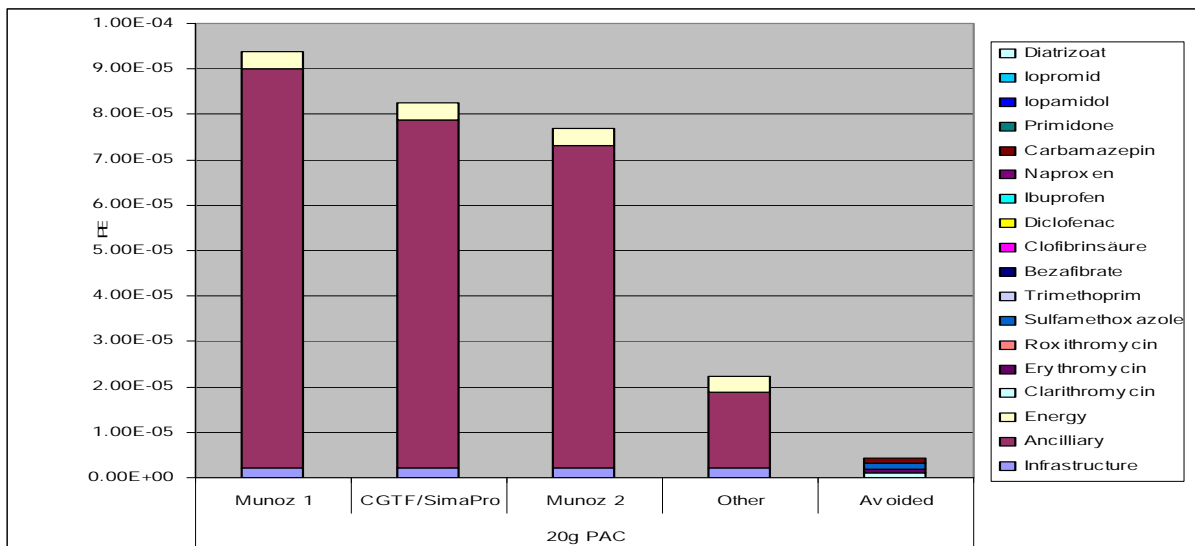


Figure A15.4 Sensitivity analysis: AC production – showing the effect of using different models on the induced part (CGTF/SimaPro, Munoz 1, Munoz 2 and “other” referring to ProBas) as compared to the avoided part for PAC dose 20 g/m3 (unit on y-axis PET/m3)

As we can see from the figure A15.4 (taking into account the first three modeling scenarios only), the range of values for PAC production varies with a maximum of about 20%. Compared to figure A15.2 this implies that the data is actually more stable in this case. Therefore, the overall conclusion that the induced impacts are significantly larger than the avoided ones seems robust.

As a result, it becomes clear that recycling is actually very beneficial when using AC. However, this is not the case in NEPTUNE and the result is that the induced impact is expected to be at the level of 200 times higher than if AC (i.e. PAC) were recycled. Anyway, using a model on AC production without recycling makes the results less sensitive to modeling differences.

Finally, the major part of the high impact associated with fossil-based AC is related to non-biogenic global warming emissions resulting from fossil fuel combustion (coal). Therefore, it may be interesting in future iterations (if better data becomes available) to model AC production from organic material such as wood or nutshells to see if the lower burdens associated may make the technology more attractive.

A15.1 Electricity

Electricity is a parameter in the NEPTUNE LCA models that has been identified as having a more or less generally major contribution to the induced environmental impact potentials. In addition, given the fact that the LCAs carried out in this study are considered consequential, the choice of electricity modeling scenario is particularly important.

Marginal vs. average technology: According to Mattson et al. (2003), when carrying out consequential LCAs, one should model marginal technologies. These are the technologies expected to account for the changes in demand generated by a given process.

In our case, this concept is only significant only for the electricity production technologies because all other processes including production and transport are expected to remain the same in the foreseeable future. Nevertheless, transport might change as regulations might start requiring a certain percentage of fuels to be of biogenic origin and transport to be at least partly based on electricity. This last point is not covered in this study.

Also, electricity in the physical inventory is expected to have a significant impact in the ancillary and energy categories only, as infrastructure processes rely very little on electricity. Instead, big infrastructure contributors are steels and concretes, relying mainly on direct combustion heat and transport. As a result, only the impacts associated with ancillary products and process energy will change due to the different electricity modeling scenarios.

Finally, it is important to point out that the concept of marginal technologies has been approved at the NEPTUNE 2008 Varna meeting.

Electricity production: Figure A15.5 shows the range of potential impacts from electricity production obtained through different modeling scenarios. These scenarios include:

- Marginal technologies
 - CH: electricity, at cogen 500kWe lean burn, allocation exergy
 - UCTE: electricity, natural gas, at power plant
- Average technologies
 - RER: electricity, medium voltage, production RER, at grid
 - CH: electricity, medium voltage, production CH, at grid

- UCTE: electricity, medium voltage, production UCTE, at grid
- PL: electricity, medium voltage, production PL, at grid

Electricity production by natural gas combustion was chosen as the marginal technology for Switzerland based on Heijungs and Ekvall (2009) and Mattson et al. (2003) which states that this should be the best initial choice for the case of Nordic countries. Indeed, as modeled in EcoInvent, Switzerland has an electricity grid mix very similar to the Nordic countries with the dominant technologies being hydropower followed by nuclear power and in third position, natural gas. It is therefore expected that natural gas will be the technology accounting for changes in electricity demand.

Furthermore, natural gas power plants are increasingly viewed as being among the most environmentally friendly and likely (in economic and other terms) purveyors of electricity (Mattson et al. 2003). Based on this fact and the aforementioned recommendation by Heijungs and Ekvall (2009) and Mattson et al. (2003), natural gas was also chosen as the marginal technology on the European level. This has the additional advantage that both the European and Swiss electricity production by natural gas have a very similar environmental profile as shown in Figure A15.5, thereby “normalizing” all models with similar environmental profiles from electricity production.

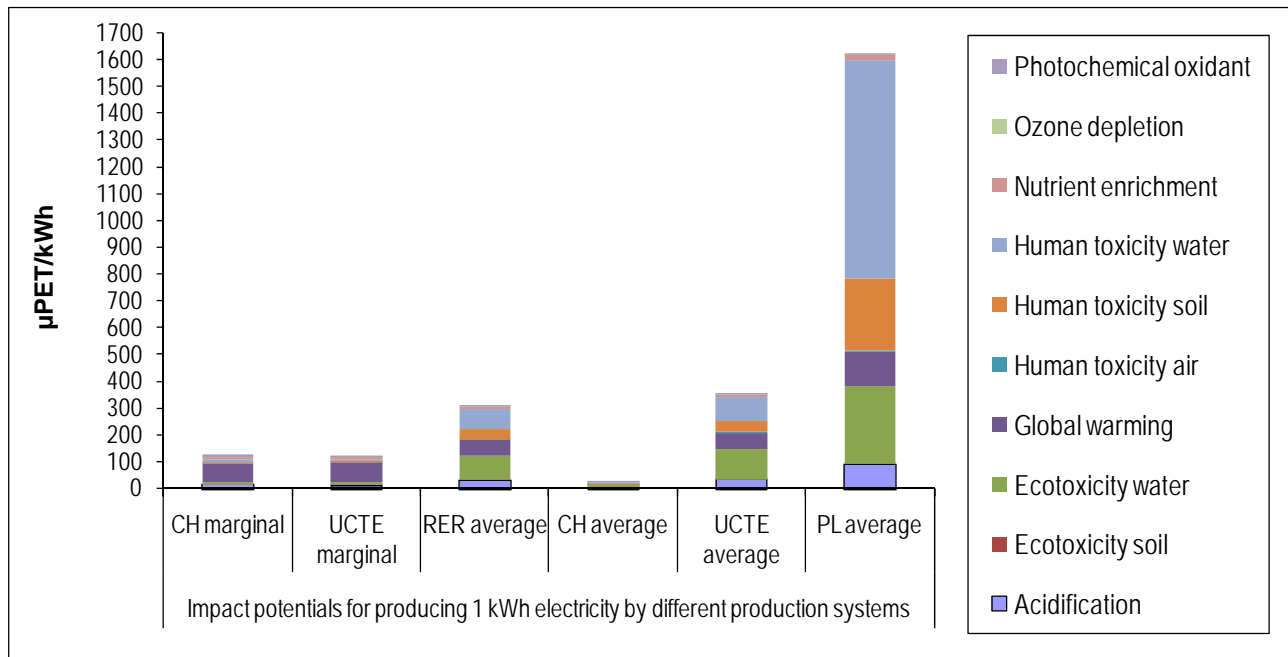


Figure A15.5 Normalised and weighted (WF=1) impact potentials for producing electricity by different systems (natural gas regarding marginal and grid mix regarding average) in Switzerland (CH), Europe (UCTE, RER) and Poland (PL).

As shown in figure A15.5 the range of impacts associated with electricity is rather large when considering both marginal and average technologies, i.e. the extreme is Polish grid average as compared to grid average from Switzerland – a factor 77 in difference. However, when considering only the two marginal technologies based on natural gas, i.e. Swish (CH) as compared to European (UCTE), the difference is only about 3%. Therefore, the marginal technology modeling is considered to have a lower uncertainty and expected to not influence the overall conclusions based on the models significantly.