CREEA - Compiling and Refining Environmental and Economic Accounts

Task 3.3:

Water quality accounts for physical, chemical and thermal pollution

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The main goal of CREEA is to refine and elaborate economic and environmental accounting principles as discussed in the London Group and consolidated in the future SEEA 2012, to test them in practical data gathering, to troubleshoot and refine approaches, and show added value of having such harmonized data available via case studies. This will be done in priority areas mentioned in the call, i.e. waste and resources, water, forest and climate change / Kyoto accounting. In this, the project will include work and experiences from major previous projects focused on developing harmonized data sets for integrated economic and environmental accounting (most notably EXIOPOL, FORWAST and a series of EUROSTAT projects in Environmental Accounting). Most data gathered in CREEA will be consolidated in the form of Environmentally Extended Supply and Use tables (EE SUT) and update and expand the EXIOPOL database. In this way, CREEA will produce a global Multi-Regional EE SUT with a unique detail of 130 sectors and products, 30 emissions, 80 resources, and 43 countries plus a rest of world. A unique contribution of CREEA is that also SUT in physical terms will be created. Partners are:

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Summary

The main objective of task 3.3 is to help standardize water quality accounts. For this purpose, a review of the approaches in setting water quality standards and an inventory of ambient water quality standards are provided. An example of integrating water quality in national accounts is provided through the experimental water quality accounts that have been developed for the Netherlands. The quantification of thermal and nutrient (nitrogen and phosphorus) emission to the water system is further discussed. Furthermore, discussion on how to go from emissions to impact using LCIA and the comparison of LCIA and grey WF methodologies in environmental assessment of nutrient emission is provided. Finally a brief discussion is presented on how thermal pollution and grey water footprint can be made comparable and how water pollution can be made comparable to water consumption through the water footprint concept.

1. Introduction

One of the aims of environmental accounting is to describe the pressures the economy exerts on the environment in the form of physical flow accounts such as energy accounts or air emission accounts. In case of water, this is expressed in the form of water emission accounts that quantify emissions by economic activity. These pressures eventually result in environmental impacts that have an effect on the quality of water resources. Water quality accounts provide a description of water resources in quantitative and qualitative terms for a country as a whole or at a sub-national level, in such a way that different types of water resources (rivers, lakes, etc.) can be compared, for instance in terms of volume or surface area. Water quality accounting is a relatively undeveloped area of environmental accounting. The SEEAW (System of Environmental-Economic Accounting for Water) (UN, 2012a) contains a chapter on water quality accounts, but considers these accounts experimental as few internationally accepted best practices have emerged so far. Although there are various statistics on water quality[1], water quality accounts i.e. the integration of such data with economic and social statistics, would allow for a comprehensive understanding of the interaction between the economy and the environment.

In this deliverable we discuss research that was undertaken in the combined area described above of emission and water quality accounting.

The structure is as follows. Section 2 discusses existing water quality standards. Section 3 reports on experimental water quality accounts that have been developed for the Netherlands. Section 4 discusses several new approaches that have been developed in the area of emission accounts. While emission accounts traditionally focus on physical and chemical pollution, results are presented of quantifying thermal pollution following an accounting approach. Section 4 also discusses the grey water footprint and its extension to phosphorous. Section 5 discusses how to go from emissions to impact using LCIA. Section 6 concludes.

[1] http://www.compendiumvoordeleefomgeving.nl/

2. Ambient water quality standards

Water quality accounts have little meaning if not accompanied by (context-dependent) water quality standards. Water quality standards have historically been concerned with the protection of human health. As a result, earlier water standards focuses on the bacteriological characteristics of surface waters that could be used as raw water supplies (MacDonald, 1994). However, with the expansion of knowledge of human toxicology and widespread problems of water pollution better quality standards which address the chemical attributes and designated uses of the water body become necessary.

Although ambient water quality standards often exist in national or state legislation or have to be formulated by catchment and/or water body in the framework of national legislation or regional agreement (like in the European Water Framework Directive), they vary from country to country and are often incomplete. The lack of standardization of how ambient water quality standards are established, makes it impossible to compare water quality accounts for different countries. In this context, we attempted to make an inventory of existing ambient water quality standards. We first summarize the approaches in setting water quality standards, and then make an inventory of the existing ambient water quality standards.

2.1 Approaches in setting water quality standards

Various countries have implemented different methods to develop water quality standards. Most of these methods have been developed using some variation of the theoretical toxicological approach, which is an effect-based approach that relies on published toxicity data from the literature (MacDonald, 1994). An extended summary of the approaches used in setting water quality standard is provided in MacDonald (1994) and Yillia (2012). Here, we have provided a brief summary of some of the approaches applied in selected countries.

EU's approach: The Water Framework Directive (Directive 2000/60/EC and its amendment Directive 2008/105/EC) sets out environmental quality standards concerning the presence in surface water of certain pollutants and substances or groups of substances identified as priority on account of the substantial risk they pose to or via the aquatic environment. The priority substances are defined by Directive 2000/60/EC (the Water Framework Directive) which establishes a list of priority substances and substances which are classed as hazardous.

The environmental objectives of the Directive are defined in Article 4. Article 4.1 defines the general objectives which include: prevention of deterioration of the status of all surface and groundwater bodies; and protection, enhancement and restoration of all bodies of surface and groundwater with the aim of achieving Good Status by 2015. The environmental quality standards are limits to the degree of concentration, i.e. the quantity in water of the substances concerned must not exceed certain thresholds. The quality standards are differentiated for inland surface waters (rivers and lakes) and other surface waters (transitional, coastal and territorial waters) (EP, 2000, 2008).

The member states are free to determine for themselves how they will meet the standards but are required to quantify what the Directive means by Good Status through an intercalibration exercise. They are also required to specify detailed values defining the status for each water body. The Intercalibration exercise is aimed at ensuring that the boundaries for Good Status given by each country's biological methods are consistent with the Directive's descriptions of Good Status.

Australia and New Zealand's approach: The Australian & New Zealand Environment and Conservation Council (ANZECC) have developed water quality guidelines for marine and freshwater systems (ANZECC guidelines) for both countries. The main objective is to provide an authoritative guide for setting water quality objectives that can be tailored to local environmental conditions in both Australia and New Zealand. The ANZECC guideline values are regarded as guideline trigger values that can be modified into regional, local or site-specific guidelines by taking into account factors such as the variability of the particular ecosystem or environment, soil type, rainfall and level of exposure to contaminants (ANZECC/ARMCANZ, 2000).

The first steps followed in setting the guideline values include collecting all available and technical information for a defined water body. Then, the environmental values that are to be protected in a particular water body and the spatial designation of the environmental values are identified. The environmental values recognized in the guideline include: aquatic ecosystems, primary industries (irrigation and general water uses, stock drinking water, aquaculture and human consumption of aquatic foods), recreation and aesthetics, drinking water, industrial water, and cultural and spiritual values. Once the environmental values for a water body have been identified, the level of environmental quality or water quality necessary to maintain each value is determined and the relevant water quality guidelines (a numerical concentration limit or narrative statement recommended to support and maintain a designated water use) for measuring performance are selected. Based on these guidelines, water quality objectives that must be met to maintain the environmental values are set (ANZECC/ARMCANZ, 2000).

India's approach: In India, the water quality standard is set based on the intended use of the water body. The Central Pollution Control Board (CPCB) of India has developed a concept of "designated best use". According to which, out of several uses a particular water body is put to, the use which demands highest quality of water is called its "designated best use", and accordingly the water body is designated. CPCB has identified five "designated best uses" such as drinking, outdoor bathing, propagation of wildlife and fisheries, and irrigation and industrial cooling. For each of these five "designated best uses", water quality requirements in terms of few chemical characteristics, known as primary water quality criteria are identified (CPCB, 2008). The water quality parameters considered are pH, Temperature, Turbidity chlorides, SO4 NO3, BOD, DO, TDS, coliform.

South Africa's approach: The derivation of water quality criteria is based on the best available scientific and technical information in the form of numerical and qualitative that describe its potential effects on the health of species representative of major trophic groups occurring in aquatic ecosystems and the fitness of water for other uses. The rationale for this is that if the most sensitive species within representative trophic groups are protected, then other species within the trophic group will also be protected. The criteria used in the South African Water Quality Guidelines were derived by assuming continuous and long term exposure (life-long exposure) to water of a given quality and incorporate a margin of safety. The South African Water Quality Guidelines consists of guidelines for domestic, recreational, industrial and agricultural water uses, guidelines for the protection of aquatic ecosystems as well as guidelines for the protection of the health and integrity of aquatic ecosystems and guidelines for the protection of the marine environment (South African Government, 1996).

USEPA's approach: The United States Environmental Protection Agency (USEPA) has developed a formal protocol for deriving generic, numerical water quality criteria for the protection of aquatic life and their uses (Stephan et al. 1985). Using this approach, information is compiled on the physical and chemical properties of the substance under consideration, on its toxicity to aquatic plants and animals, on its bioaccumulation in aquatic organisms, and on its potential effects on consumers of aquatic biota. The formalized protocol includes specific procedures for calculating final acute (FAV), final chronic (FCV), final plant (FPV), and final residue values (FRV) from the available data, provided that the minimum data requirements have been met (MacDonald, 1994)

In the US, water quality standards are risk-based requirements which set site-specific allowable pollutant levels for individual water bodies, such as rivers, lakes, streams and wetlands. States can set water quality standards autonomously by designating uses for the water body (e.g., recreation, water supply, aquatic life, agriculture) and applying water quality criteria to protect the designated uses in addition to issuing an antidegradation policy to maintain and protect existing uses and high quality waters (Yillia, 2012).

2.2 Water quality standards

Countries throughout the world have developed water quality standards for a wide range of pollutants and for different uses of the water body (e.g., drinking water supply, recreation, aquatic life, agriculture, industrial use). Sometimes, instead of single concentration level, different concentrations are set as the quality standard for different assessment levels (e.g., from very good to severely polluted situation).

It is practically impossible to present water quality standards for all countries of the world for different pollutants and different uses of the water body. An extensive compilation of water quality standards can be found in MacDonald et al. (2000). Summary of drinking water quality standards and guidelines is also provided in Carr and Neary (2008). The Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) also provide an inventory of water quality standards for a number of countries (GESAMP, 2013). We have adopted this inventory and provided a summary in Appendix 2.2.

3. Dutch water quality accounts[2]

Statistics Netherlands has a long tradition in water accounting. The NAMWA (National Accounting Matrix including Water Accounts) (Van Rossum et al, 2010) consists of three types of accounts: water use, emissions to water, and regional water accounts. Water use includes abstraction of ground and surface water, (tap) water use and (tap) water use intensity. Emissions to water include heavy metals and nutrients. The regional water accounts show the regional differences in water use and emissions for the different river basins. Recently a water balance has been developed (Graveland and Baas, 2012). Eventually, water quality accounts could be an addition to the Dutch water accounts.

This feasibility study was undertaken in the context of the CREEA (Compiling and Refining Environmental and Economic Accounts) project, which has a work package on water accounts. It entailed a stock-taking of possible data sources, an articulation of methodology, and compilation of pilot accounts. The results presented in this chapter should be considered experimental, but will hopefully facilitate a dialogue with policy makers and interested parties.

In order to compile accounts that may be a relevant source of information for policy makers, we have decided to take the Water Framework Directive (WFD) as a point of departure for this research. This has the additional advantage that it provides an answer to various contentious issues in water quality accounting, most importantly the definition of quality classes. The scope of the research was restricted to surface water.

The outline of this section is as follows. In Section 3.1 we will provide background information about the WFD and water quality accounts in relation to environmental accounting. Section 3.2 discusses sources and methods, followed by experimental results in Section 3.3. Section 3.4 provides the conclusions.

3.1 Context

Water Framework Directive

The European WFD was adopted in October 2000 and entered into force in December of that year (EP, 2000). The WFD is an important and leading environmental policy directive in Europe. European waters must meet good quality requirements by the year 2015. The WFD includes surface water (marine, brackish and sweet) and groundwater. EU member

[2] This section was published as Chapter 11 in Statistics Netherlands (2012).

states are required to send in reports on water quality once every three years (EEA, 2010).

For surface water the WFD divides areas into 'river basin districts' and 'sub river basin districts'. In the Netherlands four river basin districts are identified: Ems, Rhine, Scheldt and Meuse. All Dutch river basin districts are part of international river districts, which means that the policies for the river basin districts also need to be coordinated on an international level. In the Netherlands, the Rhine district is divided into four sub river basin districts.

A key element of the WFD is the identification of water bodies (EC, 2003). The WFD classifies water bodies based on their 'status' and 'type'. The status indicates the degree to which water bodies have been modified: natural, artificial or strongly modified. Most Dutch water bodies fall under the classification of 'artificial' or 'strongly modified'. Water 'type' describes the water debit (rate of flow), the kind of water (river, lake, coastal or transitional) and soil type (sand, clay, etc.). Based on these characteristics a classification scheme of 50 different water types has been developed for the Netherlands (Elbersen et al., 2003). See for an example Figure 3.1.1. A river can be divided in several WFD water bodies because it flows through multiple water manager districts. Another reason can be the water status or type. For example a section of a river that is modified into a canal can be a separate water body. Distinct pressures, like a factory emitting to water, can be another reason to divide a river into multiple WFD water bodies. In this way 724 water bodies have been identified by water managers in the Netherlands. The 724 Dutch WFD water bodies represent 36 of the water types identified by Elbersen et al. (2003). The water managers do not assign a WFD status to small waters like creeks and ponds. In addition, there are 23 groundwater bodies in the Netherlands that are assessed for quantity and quality by the WFD.

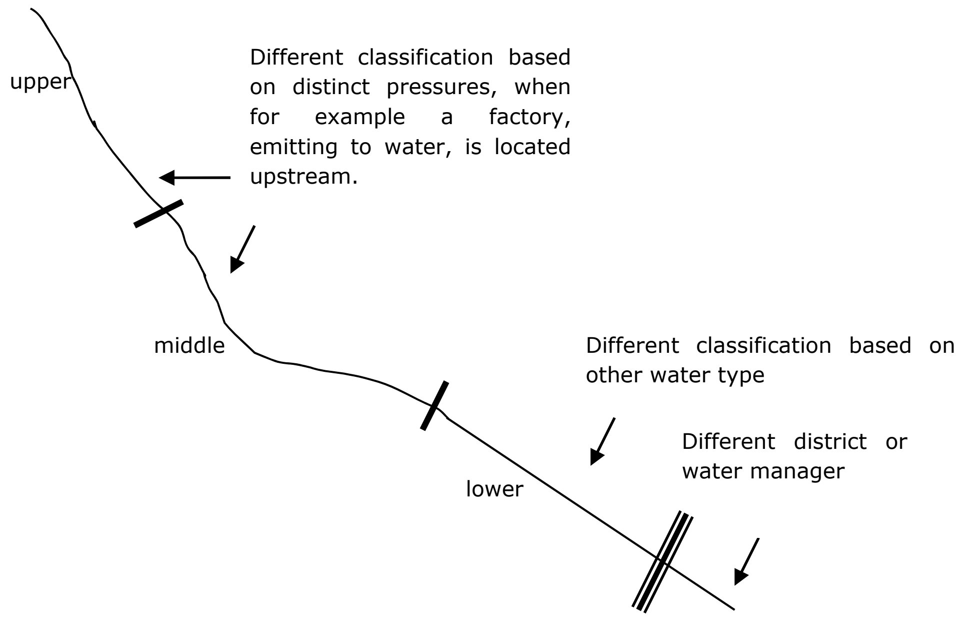

Figure 3.1.1 Allocation of WFD water bodies
Source: Based upon EC (2003).

In terms of reporting requirements, the WFD makes a distinction between obligatory and optional factors (EP, 2000). These factors differ from one water type to another. For example; obligatory factors for coastal waters are: longitude, latitude, tidal range and salinity. Optional factors are current velocity, wave exposure, mean water temperature, mixing characteristics, turbidity, retention time (of enclosed bays), mean substratum composition, water temperature range. It is mandatory to report whether a water body is part of a protected area.

In the Netherlands there are 243 water bodies with a 'protected area' status. Four sorts of protected area types are reported in the Netherlands: Bathing water, Drinking water, Shellfish water and Natura 2000 areas. The Natura 2000 areas consist of areas that either fall under the 'habitat directive', the 'bird directive' or under both (EP, 2000). The Netherlands has two shellfish areas; one in the Wadden Sea (North East of the country), and one in the Oosterschelde (South West).

In terms of quality, the WFD determines the status of a water body in terms of ecological and chemical assessments. How these assessments are constructed is illustrated in Figure 3.1.2.

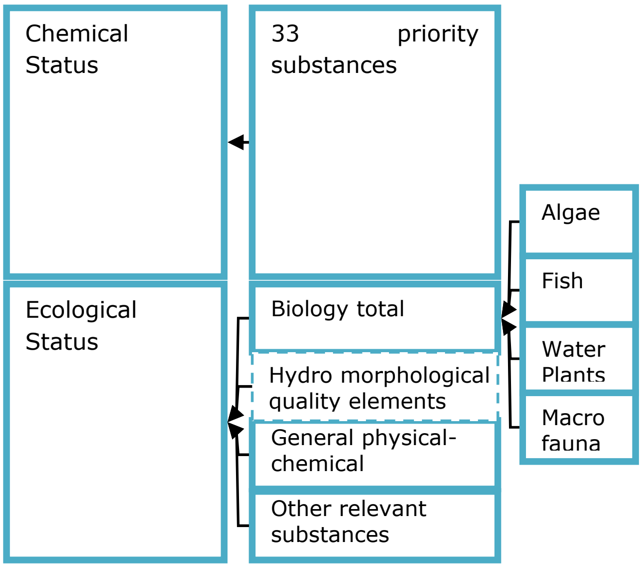

Figure 3.1.2 Derivation of status of water bodies

The chemical status is determined on the basis of 33 priority substances listed in the Directive (EP, 2000). The ecological status is determined by four sub indicators. The first indicator of biological conditions in turn is determined by four indicators on fish, algae, water plants and macro fauna. The second indicator 'hydro morphological quality elements', concerns the physical system of water bodies (like appearance of banks, soil/substrate etc.). This indicator is not fully developed yet. The third physical-chemical indicator reflects aspects like water debit and temperature. The fourth group of indicators 'other relevant substances' are those that are not part of the 33 internationally determined substances yet thought to be important.

Chemical status is classified into the two classes: good and bad. The ecological status is assessed in five classes: high, good, moderate, poor, and bad. How the measurements for the parameters are translated to the final assessment is described in more detail for the Netherlands in Rijkswaterstaat (2011).

The WFD follows the 'one out, all out' or 'worst of the worst' assessment system: the lowest assessments for a parameter of a (sub) indicator determines the final outcome. This means that when one of the 33 chemical substances fails to achieve a good status, the chemical status is 'bad'. Reporting for the WFD is required every three years. The most recent reporting year is 2009 and the next reporting year is 2012.

Quality accounts in relation to environmental accounting

To position quality accounts in the DPSIR (Driving forces-Pressures-States-Impacts-Responses) framework, quality accounts would describe the resulting States of water resources. Environmental accounting is traditionally mostly concerned with analysing the Driving forces and Pressures (e.g. in water use accounts or water emission accounts). The quality accounts could be used as a starting point to analyse Impacts (e.g. in terms of environmental quality of life) and Responses, for instance by assessing the effectiveness of policy instruments such as environmental taxes, subsidies or regulations.

Technically speaking, water quality accounting is a form of physical asset accounting, in which the state of water resources is described in both quantitative and qualitative terms. An important aspect of developing water quality accounts is aggregation across various types of water resources (e.g. in terms of volume, surface area or standard river units). Quality accounts could be used to further disaggregate water asset accounts and provide a description of water resources of a country or at the sub-national level.

Water quality accounting could also be instrumental in the emerging area of ecosystem accounting, where the environment is described in terms of ecosystems that provide various ecosystem services.
In this feasibility study, we have therefore investigated to what extent a classification of water bodies by functions or uses is available for the Netherlands.

3.2 Sources and methods

Data sources

For this feasibility study a stock-taking exercise was made of the available data sources for the year 2009, the most recent reporting year of the WFD. We chose 2009 because the data of the first reporting year, 2006, is not as reliable. This initial year was used as a base year to set up the monitoring system.

The most important data sources that we have used are:

- Aquatic Base Map (*in Dutch: Map Basiskaart Aquatisch Top10NL – Kadaster*)
  The Base Map was developed by Wageningen University and PBL (PBL, 2010). The Aquatic Base Map is a GIS (Geospatial Information System) map that indicates the location of all Dutch surface water bodies. The water bodies are classified according to the WFD water types. The map provides a classification into water types for the whole country and a clear link to the WFD. Three databases form the core of this source: (1) a polygone map, (2) a polyline map, (3) a database including a classification of water types. The map was published in 2010. The TOP10NL data is based on the year 2006 (most recent Kadaster map) and has a

scale of 1:10.000. The WFD water bodies division is based on the most recent reporting year i.e. 2009.

- WFD Portal. (*In Dutch: KRW Portaal*[3])

  This website is administered by the 'Information House Water[4]' and online since February 2012. The WFD Portal provides a large number of databases for public use. The website contains data for surface water and groundwater, the data is reported in the year 2009 and measurements are done in previous years. For our project several databases proved relevant:
  - Theme 1: databases with a description of surface water bodies;
  - Theme 2: databases with information on type of protected area;
  - Theme 3: databases with surface water quality assessments.

Integration of data sources

The Aquatic Base Map consists of two maps that do not overlap: the polygone map includes areas of the larger water bodies and the polyline map is one-dimensional and consists of lines, showing for example rivers and canals with a width of six meter or less. In the Aquatic Base Map one WFD water body can be split up into several polygone and/or polyline parts. There are 724 WFD water bodies, of which 696 appear in the polygone map and 507 in the polyline map. The two sources together cover all 724 WFD water bodies. The total surface area of Dutch water resources in the polygone map is 16.259 square kilometres. The WFD water bodies make up almost 95 percent of this area. As the actual surface area of water bodies listed in the polyline map is unknown, and in either case very small compared to the surface of the polygone map, the analyses have been done on the polygone map only.[5]

The WFD, or actually the water managers, assign unique codes to water bodies. This code is called 'OWMIDENT' and all 724 water bodies in the Netherlands have an OWMIDENT code. These unique codes have been used to link the various data sources. In order to link water data with economic and social statistics, the Aquatic Base Map was intersected with a grid of Dutch zip codes. The zip codes used are the 'PC4' areas, which are formed by the four numbers of the zip codes in the Netherlands. There are 4221 PC4

[3] http://krwportaal.nl/portaal/

[4] The Information House Water intends to gather and manage water data and to implement a uniform coding system known as the 'Aquo Standaard' which is accessible via the following link: http://www.aquo.nl/aquo-standaard/aquo-domeintabellen/.

[5] The Basiskaart and the Basiskaart enriched with PC4 codes give slightly different totals for total surface (polygone) and length (polyline). The difference is within 50 square kilometres. This is caused by numerical difference because of a technology used to assign 'empty' areas to one of the joining area types. The analyses in the first three sections are based on the 'original' maps, while the analysis including population is based on the enriched maps, which differ slightly.

codes in the Netherlands (1 January 2008). This is approximately the level of a neighbourhood.

Functions

Functions can be attributed to water bodies in several ways. One key question in water accounting is whether the functions of a water body determine its quality, or whether it is the other way around, where functions are derived from the water quality of water resources. In theory, there are several ways to attribute functions to water bodies: (1) based on characteristics such as concentrations of pollutants in combination with reference benchmarks, (2) based on functions 'assigned' by water managers, (3) based on actual or desired uses.

Arguably, with the introduction of the WFD, the policy focus has shifted from the second approach towards the first.[6] There are still function maps which can be found in the provincial environment plans. However, we found that the classification schemes often differ per province. Moreover, it is not always clear how useful they are: in the province Drenthe, 80 percent of the assigned functions falls under the category 'other'. Data is difficult to gather because since 2009 the provinces are no longer required to document the main functions of water. Even before 2009 the classification was not standardised, which led to a large variety of water functions.[7] Combining these provincial databases is a labour intensive process, but this was undertaken by RIVM in 2002, and resulted in a map that assigns one function per water body (RIVM, 2002). Given that for our base year 2009 most provincial plans were no longer valid, we decided that this would not be a viable approach for our project.

For the WFD, the desired water quality is not based on an analysis of water functions but on benchmark conditions. Indirectly functions are still part of the Dutch WFD, but this is not part of the mandatory reporting requirements. The way functions are introduced is as follows: only those functions that are negatively affected by a proposed measure are included (article 5). This means that this list of functions is not complete for two reasons. One: when functions are not affected, or not negatively affected, they are not included in this dataset. Two: when there are no proposed measures for a specific water body, this water body is not part of the dataset.

[6] Based on a conversation with F. Kragt of PBL (May 2012).

[7] For example the province Zuid-Holland distinguishes the following functions: water nature, provincial waterways, (reconsidering) bathing water location, surface water for preparing drinking water, urban area, other water. The province Groningen uses however a list of much more detailed functions.

The approach that we have eventually chosen is to relate functions to protected area type, as is documented in the WFD. The protected area type 'bathing water' could be interpreted as a proxy for the function 'recreation'; the 'Natura 2000' and 'Shellfish water' areas may serve as a proxy for the function 'nature' and the protected areas for 'drinking water' would be a proxy for the function drinking water. This method has the advantages that it is directly linked to the WFD coding and that these indications of protected areas are mandatory for WFD registration. A drawback is that it need not align with reality: people can swim in water even though it does not have an official 'protected' area status. However, these functions are the ones with the most stringent water quality requirements. Another drawback is that it only results in a limited set of functions.

3.3 Results

In this paragraph we present our main experimental results. The first section gives a general description of Dutch water bodies, the second section looks at the chemical and ecological status of water bodies according to water type. The third section describes the quality of the different types of protected areas. The fourth section discusses the results of an analysis, in which the Dutch population (by province) is classified according to the quality of water resources in their neighbourhood.

Dutch surface water types

The 724 water bodies in the WFD can be divided into water types. First, we look at the division of water bodies over all WFD water types. As shown by Table 3.3.1 counting the number of water bodies belonging to a water type gives a somewhat different picture, compared to looking at the surface area of those water bodies belonging to one water type. While the majority of WFD water bodies consists of lakes (450 out of 724), they have a joint area of only 18 percent. Rivers make up only 2.7 percent of total water surface area. The coastal water areas account for 75.5 percent of the total WFD water area. This means that the quality assessments based on surface are largely determined by the quality of these coastal areas.

Table 3.3.1 Water bodies disaggregated by water type

Water type	Description	Number	Km^2	Area (percentage)
Lakes		**450**	**2,808**	18.2%
M1a	*Freshwater ditches (well buffered)*	46	3	
M1b	*Non-freshwater ditches (well buffered)*	1	0	
M2	*Ditches (weak buffered)*	2	0	
M3	*Buffer zone (regional) canals*	99	44	
M6a	*Large shallow canals without shipping*	22	7	
M6b	*Large shallows canals with shipping*	16	36	
M7a	*Large deep canals without shipping*	1	0	
M7b	*Large deep canals with shipping*	17	52	
M8	*Buffer zone (low) fen/marsh/bog ditches*	18	3	
M10	*Weak buffer zones (high moorland) ditches*	31	38	
M12	*(Low) fen waterways and canals*	1	0	
M14	*Small very shallow weakly ponds (buffer zone)*	51	334	
M20	*Very Shallow ponds(buffer zone)*	29	112	
M21	*Moderately large deep lakes (buffer zone)*	2	1,834	
M23	*Large deep lakes (buffer zone)*	6	4	
M27	*Large shallow lime-rich ponds*	25	107	
M30	*Moderately large very shallow low fen ponds*	61	93	
M31	*Moderately brackish waters*	20	6	
M32	*Small brackish till saline waters*	2	135	
Rivers		**254**	**413**	2.7%
R4	*Permanent slow flowing upper course on sand*	47	1	
R5	*Slow flowing middle/lower course on sand*	133	17	
R6	*Slow flowing small river on sand/clay*	30	21	
R7	*Slow flowing river/side channel on sand/clay*	11	157	
R8	*Fresh tidal water (river) on sand/clay*	10	205	
R12	*Slow flowing middle/lower course on peat*	6	1	
R13	*Rapid flowing upper course on sand*	2	0	
R14	*Rapid flowing middle/lower course on sand*	3	0	
R15	*Rapid flowing river (siliceous soil)*	1	1	
R16	*Rapid flowing river/ side channel (siliceous or sandy soil)*	1	9	
R17	*Rapid flowing upper course on lime rich soil*	6	0	
R18	*Rapid flowing middle/lower course on lime rich soil*	4	1	
Coastal water types		**15**	**11,653**	75.5%
K0	*Coastal water, open and polyhaline*	5	7,760	
K1	*Coastal water, sheltered and polyhaline*	3	499	
K2	*Coastal water, open and euhaline*	5	2,621	
K3	*Coastal water, open and polyhaline*	2	773	
Transitional water		**5**	**555**	3.6%
O2	*Estuarium with moderate tidal movement*	5	555	
	Total	**724**	**15,429**	

Source: Calculation of area is based on the polygone map only; water types are based on Elbersen et al. (2003) and Marcel van den Berg (RIVM) assisted with the translation.

Surface water quality

Water quality is presented in chemical and in ecological status separately.[8] Chemical status can be classified as 'good' or 'bad', while ecological status can be classified as 'high', 'good', 'moderate', 'poor' and 'bad'. High quality does not occur in the Netherlands, and is for that reason left out of this analysis. These results concern water bodies that have both a surface in the polygone map and a WFD assessment. For the

[8] For the results the combined indicators 'chemtc' and 'ecoltc' are used for the chemical and ecological assessments respectively. These data follow the same pattern as the EEA surface water viewer (EEA, 2012).

chemical status there are 685 water bodies in total, and for the ecological status there are 716 water bodies. As Table 3.3.2 shows, the assessment presented in surface area gives a very different picture compared to the number of water bodies: while 506 water bodies have a good chemical status, together these only form 7 percent of total surface area.

Table 3.3.2 also demonstrates the weak relation between chemical status of water bodies and their ecological status, as is also indicated in the report of RIVM (2004). While most water bodies (±74 percent) have a good chemical status, the majority also has a bad or poor ecological status (±65 percent).

Table 3.3.2 Quality assessment of water bodies

Status	Number	Surface area	
		km2	*percentage*
Chemical status	685		
Good	506	1.137	7%
Bad	179	14.275	93%
Ecological status	716		
Good	3	2	0%
Moderate	249	4.707	61%
Poor	315	2.689	35%
Bad	149	270	4%

Figure 3.3.1 depicts water quality by water type, based on surface area. It demonstrates that the inland water bodies are of a relatively better chemical status than the coastal and transitional waters. Again, the chemical status does not translate directly to good ecological status. Lakes and rivers have relatively more 'bad' and 'poor' water bodies. The coastal and transitional water bodies on the other hand have high percentages of 'moderate' ecological status. The reason for this 'mismatch' could be the use of the 'one out, all out' assessment system. Transitional and coastal waters might not meet the standards for one or two chemical parameters that have little or no influence on the ecological status.

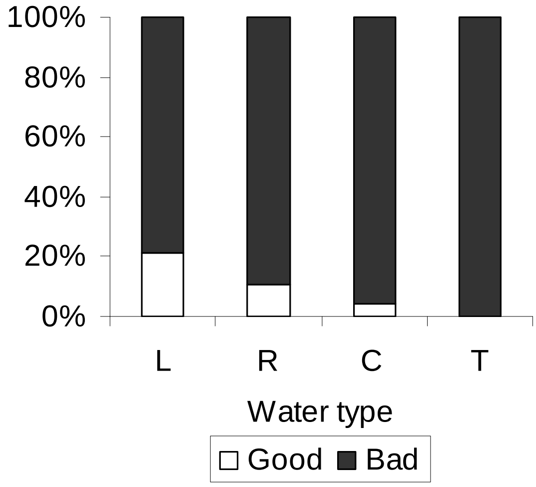

Figure 3.3.1a Chemical status of main water types (The abbreviations represent: L-Lakes, R-Rivers, C-Coastal and T-Transitional).

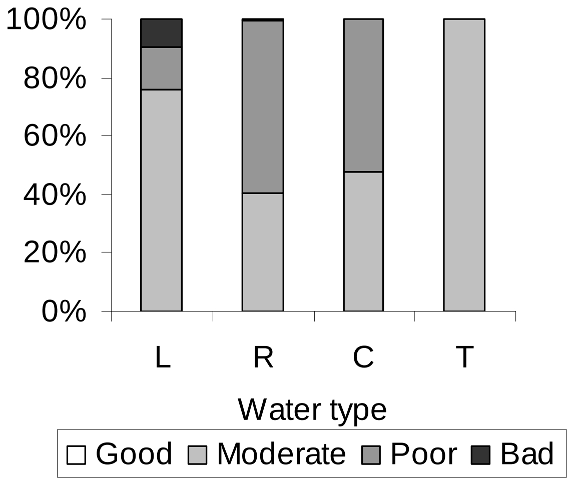

Figure 3.3.1b Ecological status of main water types

Functions

There are 243 water bodies with one or more 'protected area type'. This means that a water body (partly) intersects or overlaps with a protected area like a 'Natura 2000' area or includes a bathing water location. When multiple protected areas of one type are linked to a water body, this is counted as 'having a protected area type' status. The 243 water bodies with a protected area status have on average 1.8 types of protected area assigned to them.

Figure 3.3.2 provides an overview of the chemical status of various protected areas, where we have disaggregated the Nature 2000 areas into those that fall under the Bird Directive and/or the Habitat directive. It should be noted, however, that because most of the water bodies have multiple protected area types (and the larger the water body, the more likely it becomes that multiple protected areas are assigned), there is a large degree of double counting involved. We observe that the chemical status for the majority of types is 'bad'.

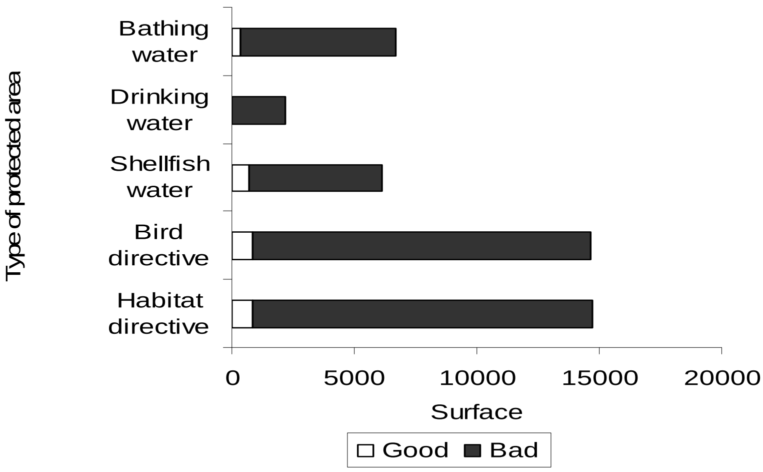

Figure 3.3.2 Chemical status of protected area types (square km)

Population and water quality

Table 3.3.3 contains the results of intersecting the Aquatic Base Map with a stratification of zip codes. We see that about one percent of the Dutch population does not live in zip codes that contain water. Nearly 21 percent of the population (3.4 million people) live in zip codes with water that is not classified as a WFD water body. The remaining 12.7 million lives in a zip code with at least one WFD water body. It is for this last category that we make an analysis of the quality of the water bodies in their neighbourhood.

Table 3.3.3 Population (provinces) by presence of water bodies in their neighbourhood

Provinces	Population	Not near water	Not near WFD water	Near WFD water
Groningen	573.245	100	64.225	508.920
Friesland	643.110	95	63.655	579.360
Drenthe	487.700	220	98.925	388.555
Overijssel	1.119.405	8.690	224.235	886.480
Flevoland	378.660	-	60.395	318.265
Gelderland	1.983.125	32.990	606.400	1.343.735
Utrecht	1.200.530	6.725	299.975	893.830
Noord-Holland	2.625.285	23.945	354.380	2.246.960
Zuid-Holland	3.460.835	18.810	699.275	2.742.750
Zeeland	380.565	-	43.865	336.700
Noord-Brabant	2.424.700	52.015	563.330	1.809.355
Limburg	1.123.385	43.930	399.660	679.795
Total	**16.400.545**	**187.520**	**3.478.320**	**12.734.705**
		1%	21%	78%

To be able to say which share of the population lives in zip codes that contain water of a certain quality, we made the following assumptions:

- The WFD water body or bodies in a PC4 area determine the water quality assessment for the entire area. This means that even when part of the water within such an area is not WFD water, the water body, or water bodies that are WFD water bodies determine the assessment.
- When two or more WFD water bodies of differing water quality intersect a single zip code, its inhabitants are assigned to different quality classes based on the respective surface areas of these water bodies.

This approach is chosen because it allows for more differentiation than following the 'worst of the worst' rule. Otherwise all people within a PC4 code that intersects with a part of a water body with 'bad' water quality would be listed as living near water of 'bad' chemical status.

Figure 3.3.3 provides a further breakdown of the 12.7 million people who live in neighborhoods with WFD water bodies, disaggregated by ecological status. We find that 18 percent of the population lives near surface water of bad ecological quality, followed by 45 percent with poor ecological quality. In Utrecht, Overijssel, Drenthe and Friesland there is a relatively large share of 'moderate' ecological status combined with a small share of 'bad'. The three water bodies with a 'good' ecological status are located in Utrecht, Gelderland and Flevoland.

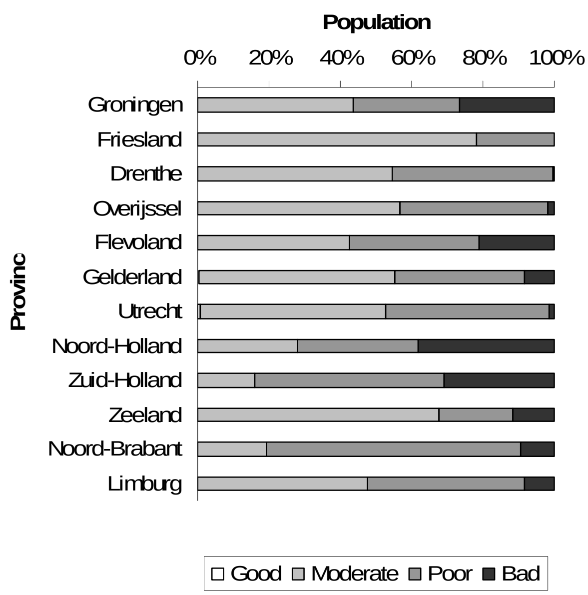

Figure 3.3.3 Population (by province) by ecological quality of water bodies in their neighbourhood

3.4 Conclusions and discussion

There are a number of outstanding issues in the area of water quality accounting as described in SEEAW (UN, 2012a). One of the most important issues, the definition of quality classes, is overcome by taking the WFD as point of departure. The Directive defines water quality classes and contains an elaborate system to measure and assess the water quality of water bodies. Another advantage of the WFD is that it serves as a standard for Europe. While using the WFD has many advantages, the system is not exhaustive because small water areas are excluded from the analysis. However, in terms of surface area, 95 percent of Dutch water bodies are covered by the system.

The one-out-all-out assessment method for water quality is considered too restrictive by some (Hering et al., 2010). Related to this is the concern that policy makers accept the assessments without investigating the underlying strategy. This is especially true for the ecological assessment because not every user is aware of the strengths and weaknesses of the underlying system (Hatton-Ellis, 2008). Therefore, it might be useful to account for quality with respect to specific substances (like heavy metals).

The search for available classifications of water resources into functions resulted in the following insights. Provinces are no longer required to report water functions. This makes it difficult to assign functions based on policy. The WFD assigns quality based on reference conditions of water bodies. This is not directly based on functions, but it does provide information on 'functions negatively affected by proposed measures'. However, this set is not complete because the functions that are positively or not affected by proposed measures are left out, just like water bodies not considered for measures. The WFD does provide a dataset that links protected area types to water bodies. In this project, we have used the different protected area types as proxies for 'drinking water', 'recreation' and 'nature'. However, due to the limited number of functions that result, this is not considered a fruitful direction for future research.

The method used for the analysis of the population living near water of a specific quality should be further improved, especially taking issues such as the treatment of coastal areas into account. There is also a need to develop a more refined method to link population to presence of water of a certain quality (e.g. using weighted distances of population to water bodies).

A key accounting issue, which is also discussed at length in the SEEAW, is how to aggregate across water resources of certain quality. One of the conclusions of our feasibility study is that information about the volume of water bodies is not generally available for the Netherlands. The integration of the Aquatic Base Map with WFD databases allowed us to aggregate results using surface areas. Aggregation using surface area gives a more nuanced image than mere counting the number of water bodies. It could be a first step towards volume based aggregation. It should be possible to provide a rough estimate of average depth for different water types based on the disaggregated water types developed for the Netherlands (Table 1). As a result, volume estimates of opening and closing stocks of surface water bodies may be obtained. This would complement work on the 'Water Balance', recently developed by Statistics Netherlands (Graveland and Baas, 2012). However, the variation of water levels throughout the year (large seasonal variation) will remain a key issue (Graveland and Baas, 2012). This issue will need to be further investigated in future work.

Another area for future research is the relationship between economic data and water quality. For example, information about the location of industries which underlies the Dutch emission inventory could be linked to spatial water quality data. This could potentially be a first step towards relating pressures as described in the water emission accounts to water quality, although we should be cautious as there could be numerous non-economic factors involved. Furthermore, when additional data with reports to the WFD will become available in 2012, we can also assess changes in water quality over time. This would allow us to develop a comprehensive water quality account in which we could assess changes in water quality over time, disaggregated into different types of water.

4. Emission accounts

4.1 Thermal pollution

4.1.1 The raw data

4.1.1.1 Power plant database overview and analysis

The raw data for all calculations of thermal emissions to freshwater from the electricity industry come from the March 2012 version of the commercially available UDI World Electric Power Plants database (WEPP), a comprehensive inventory of electric power generating units with global coverage (Platts, 2012). This database includes key elements of engineering design for over 170,000 power plants worldwide, with a total installed electric capacity of over 10,000,000 MW. The coverage for thermal power plants is > 95% for large units (> 50 MW), except for China, where coverage is estimated to be > 75%. The power plants that are relevant for this analysis in terms of thermal emissions into freshwater bodies are all the thermal power stations, that is, all the steam driven units, since these require a cooling system. From the (operational) thermal power plants available in the database, the cooling system technology is reported for 74% of the total gross generating capacity of steam-electric power plants.

Power plants included in the calculations:

- Year: all data in the inventory are valid for the year 2012, however the units taken into consideration were the ones that were operational in 2007 (that is, including those which since been taken out of operation), so as to be consistent with the accounting year of the entire CREEA database.
- Only those thermal power plants were considered, for which a cooling system was explicitly identified.
- From these power plants, only those units were retained for which it is explicitly stated that a once-through cooling system is employed (other cooling systems such as cooling towers, or cooling ponds were excluded from the calculations, because the waterborne thermal pollution resulting from these technologies is minimal compared to that from once-through technologies).
- From all units with once-through cooling technologies, those using saline water were excluded (approximately 61% of all units employing a once-through cooling technology), since the sea is considered a heat sink and very local coastal thermal pollution is not considered in this work. The units retained all use either freshwater or brackish water in their cooling systems.

Based on the above a total of 1768 power plants worldwide were retained for further calculations, and their engineering design is described in more detail in the following sections.

4.1.1.2 Power plant technologies

Table 4.1.1 and Figure 4.1.1 show the distribution of technologies among the 1769 thermal power plants worldwide with a once-through freshwater cooling system that were operational in 2007. The technology of the majority of power plants employ is a straightforward steam turbine, and this group also contributes the most to the total gross generating capacity of all units together (> 90% of the total). The contribution to the total gross generating capacity of the single unit employing an organic Rankine cycle turbine is negligible[9], so this power plant is excluded from further calculations.

Table 4.1.1 Summary of technologies and total gross generating capacity of power plants included in the analysis (operational in 2007, with once-through freshwater cooling systems).

Technology	***Number of units***	***Total gross generating capacity (MW)***	***Approx. % of sum of total gross generating capacity***
CCSS (Combined-cycle single shaft configuration)	6	1.95E+03	0.5
ST (Steam turbine)	1476	3.79E+05	91.3
ST/C (Steam turbine in combined cycle)	92	1.22E+04	2.9
ST/CP (Steam turbine in combined cycle CHP (cogeneration))	4	1.96E+02	0.05
ST/S (Steam turbine with steam sendout (cogeneration))	190	2.17E+04	5.2
Sum		4.15E+05	

9 And as a consequence, the heat rejected into freshwater is also negligible in comparison.

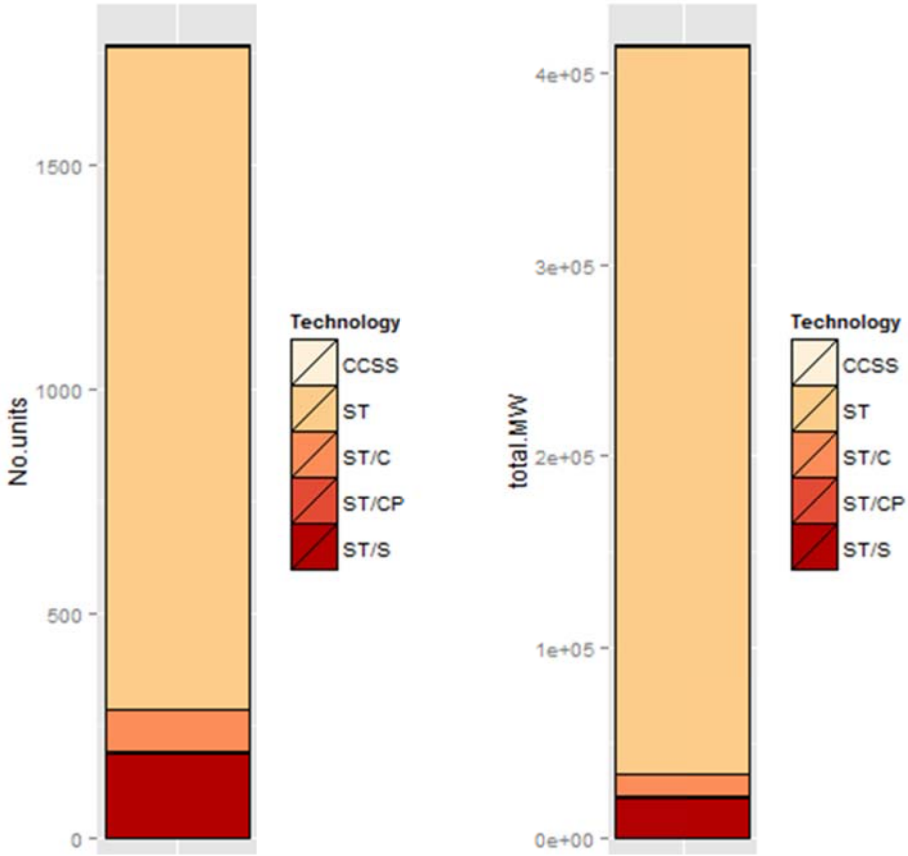

Figure 4.1.1 Left: number of units of thermal power stations per specific technology; Right: total gross electrical generating capacity per specific technology of thermal power station.

4.1.2 Methodology for the calculation of heat rejected to freshwater

The WEPP database provides no explicit information regarding the amount of heat emitted into freshwater bodies via the once-through cooling system. However, the following data are provided:

- Gross generating electrical capacity (W_{gross}, MW)
- Steam pressure at the turbine (p_3, bar)
- Type of steam (subcritical or supercritical)
- Steam temperature at the turbine(T_3, °C)
- Reheat temperature, if applicable (T_{reheat}, °C)

This information, combined with a number of assumptions, permits the estimation of the heat rejected to freshwater via the Rankine cycle, the thermodynamic cycle that describes the performance of steam engines. Given the data available from the WEPP database, the Rankine cycle can be used to predict the efficiency of each power plant, which in turn can be used to predict the amount of heat emitted into freshwater bodies.

The power plants were split into three major categories, depending on the type of Rankine cycle applicable in each case:

1. Rankine cycle, subcritical turbine pressure (Figure 4.1.2a).
2. Rankine cycle, subcritical turbine pressure, with reheat (Figure 4.1.2b).
3. Rankine cycle, supercritical turbine pressure, with reheat (Figure 4.1.2c).

4.1.2.1 Calculation of thermal efficiency - Rankine cycle, subcritical turbine pressure

Figure 4.1.2a shows the temperature-entropy plot (T-s) for a simple Rankine cycle with superheat. Work (process 1-2) and heat (process 2-3) are provided to the system through the pump and the external heat source (fuel), respectively. The cycle is completed by the production of work at the turbine (process 3-4) and the rejection of heat, in this case to the freshwater body (process 4-1). In an ideal cycle the work produced at the turbine would be isentropic, and the process would follow the line 3-4s, as shown in Figure 4.1.2a. In practice, however, the process is not reversible (there are losses) and is therefore described by the line 3-4. It is assumed in all calculations that follow that pressure drops in the system occur only at the turbine.

The thermal efficiency of the system, η_T, is given by:

$$\eta_T = \frac{(h_3 - h_4) - (h_2 - h_1)}{h_3 - h_2} \tag{4.1.1}$$

where h_1, h_2, h_3 and h_4 are the specific enthalpies at points 1, 2, 3 and 4 of the T-s plot, respectively.

The isentropic efficiency of the steam turbine, η_s , is given by:

$$\eta_s = \frac{h_3 - h_4}{h_3 - h_{4_s}} \tag{4.1.2}$$

where $h_{4_s}$is the specific enthalpy at point 4 of the Rankine cycle under isentropic conditions.

Combining Equations 4.1.1 and 4.1.2, gives:

$$\eta_T = \frac{\eta_s(h_3 - h_{4_s}) - (h_2 - h_1)}{h_3 - h_2} \quad (4.1.3)$$

The elements of Equation 4.1.3 are estimated as follows:

- h_1 is found from the temperature table for saturated water, by assuming that the freshwater body temperature is at 15 °C [10]. This also allows the estimation of the pressure, p_1, at point 1.
- h_2 is calculated via the following relation: $h_2 - h_1 = \frac{\upsilon_1(p_2 - p_1)}{\eta_{pump}}$, where υ_1 is the specific volume of water at point 1 found from the temperature table for saturated water (by assuming that the freshwater body temperature is at 15 °C), $p_2 = p_3$ (p_3, is given in the database), and η_{pump} is taken to be 0.60 (Balmer, 2011).
- h_3 is found from steam tables, using the values for pressure, p_3, and temperature, T_3, at the turbine, which are both given in the database.
- h_{4s} is found from steam tables, using the entropy calculated at point 3 (isentropic process), and the pressure p_4, which is equal to p_1.
- η_s is taken to be 0.80 (Balmer, 2011), a conservative estimate.

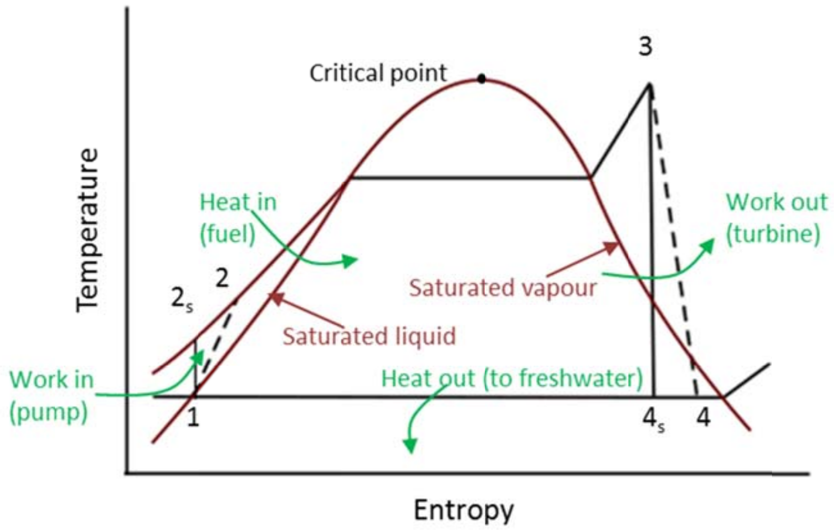

Figure 4.1.2a T-s diagram for the Rankine cycle (subcritical).

[10] While taking 15 °C to be an average yearly temperature for all freshwater bodies worldwide might appear somewhat crude, in practice, due to the much higher steam temperatures achieved (see Table 4.1.2a), differences on the order of 5-10 °C in the receiving water make little difference in the overall efficiency of the system.

4.1.2.2 Calculation of thermal efficiency - Rankine cycle, subcritical/ supercritical turbine pressure, with reheat

Figures 4.1.2b and 4.1.2c show the T-s plots for the Rankine cycles with reheat, at subcritical and supercritical turbine pressures, respectively. As in the simple Rankine cycle, work (process 1-2) and heat (process 2-3) are provided to the system through the pump and the external heat source (fuel), respectively. Work is given out at the first turbine (process 3-4) after which the temperature of the steam is raised again (process 4-5), allowing for more work to be produced at a second turbine (process 5-6). The cycle is completed by rejecting heat to the freshwater body (process 6-1). It is assumed in all calculations that follow that pressure drops in the system occur only at the turbines.

For both cases (subcritical and supercritical pressure at the turbine), the thermal efficiency of the system, η_T, is given by:

$$\eta_T = \frac{(h_3 - h_4) + (h_5 - h_6) - (h_2 - h_1)}{(h_3 - h_2) + (h_5 - h_4)} \tag{4.1.4}$$

where h_1, h_2, h_3, h_4 , h_5 and h_6 are the specific enthalpies at points 1, 2, 3, 4, 5 and 6 of the T-s plot, respectively.

The isentropic efficiency of the first steam turbine, η_{s_1}, is given by:

$$\eta_{s_1} = \frac{h_3 - h_4}{h_3 - h_{4_s}} \tag{4.1.5}$$

where $h_{4_s}$is the specific enthalpy at point 4 of the Rankine cycle under isentropic conditions.

Similarly, the isentropic efficiency of the second steam turbine η_{s_2}, is given by:

$$\eta_{s_2} = \frac{h_5 - h_6}{h_5 - h_{6_s}} \tag{4.1.6}$$

Combining Equations 4.1.4, 4.1.5 and 6 gives:

$$\eta_T = \frac{\eta_{s_1}(h_3 - h_{4_s}) + \eta_{s_2}(h_5 - h_{6_s}) - (h_2 - h_1)}{(h_3 - h_2) + (h_5 - h_4)} \quad (4.1.7)$$

The elements of Equation 4.1.7 are estimated as follows:

- h_1 is found from the temperature table for saturated water, by assuming that the freshwater body temperature is at 15 °C. This also allows the estimation of the pressure, p_1, at point 1.
- h_2 is calculated via the following relation: $h_2 - h_1 = \frac{\upsilon_1(p_2 - p_1)}{\eta_{pump}}$, where υ_1 is the specific volume of water at point 1 found from the temperature table for saturated water (by assuming that the freshwater body temperature is at 15 °C), $p_2 = p_3$ (p_3, is given in the database), and η_{pump} is taken to be 0.60 (Balmer, 2011).
- h_3 is found from steam tables, using the values for pressure, p_3, and temperature, T_3, at the first turbine, which are both given in the database.
- h_{4_s} is found from steam tables, using the entropy calculated at point 3 (isentropic process), and the pressure p_4. To estimate p_4 it is assumed that the combination of turbine pressures adopted is such that the output of the high pressure turbine is maximised, without compromising the vapour fraction (that is, without dropping below 85% vapour). This can be achieved by a setup where the pressure ratios $\frac{p_3}{p_4}$ and $\frac{p_4}{p_6}$ are equal, $\frac{p_3}{p_4} = \frac{p_4}{p_6}$, which gives $p_4 = \sqrt{p_3 p_6}$. But p_6 is equal to p_1, giving finally $p_4 = \sqrt{p_3 p_1}$.
- η_{s_1} is taken to be 0.84, for the high pressure steam turbine (Balmer, 2011), a conservative estimate.
- h_4 is found from Equation 4.1.5: $h_4 = h_3 - \eta_{s_1}(h_3 - h_{4_s})$.
- h_5 is found from steam tables, using the values for pressure, p_5, which is equal to p_4, and the reheat temperature, T_{reheat}, at the second turbine, which is given in the database.
- h_{6_s} is found from steam tables, using the entropy calculated at point 5 (isentropic process), and the pressure p_6, which is equal to p_1.
- η_{s_2} is taken to be 0.80, for the low pressure steam turbine (Balmer, 2011), a conservative estimate.

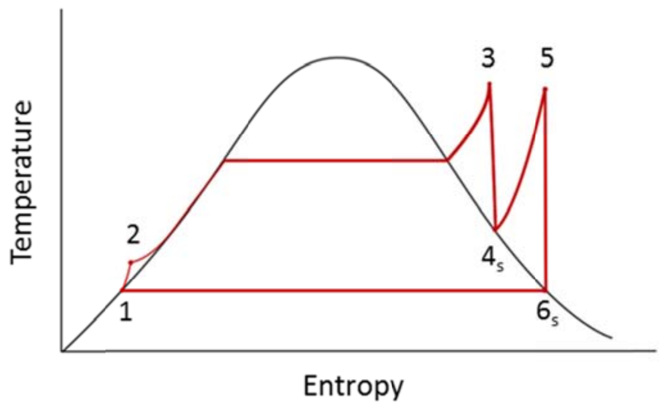

Figure 4.1.2b T-s diagram for the Rankine cycle with reheat (subcritical)

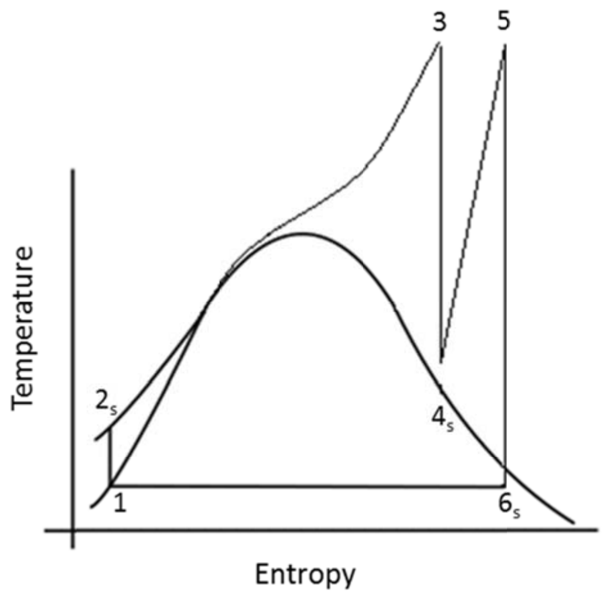

Figure 4.1.2c T-s diagram for the Rankine cycle with reheat (supercritical)

4.1.2.3 Estimated thermal efficiencies, per Rankine cycle type and specific power plant technology

Each major group, defined by its type of Rankine cycle, was further divided into subgroups according to the specific technology (defined in Table 4.1.2a). For each subgroup, the median value for the steam pressure, p_3, and the steam temperature, T_3, at the turbine were calculated, as was the reheat temperature, T_{reheat}, where applicable. For each subgroup, the thermal efficiency, η_T, was calculated according the methods described in earlier sections. The results are presented in Table 4.1.2a.

Table 4.1.2a Summary of median values estimated for key parameters of the Rankine cycle for power plants with once-through freshwater cooling systems.

A. Rankine cycle, SUBCRITICAL TURBINE PRESSURE (Figure 4.1.2a)

Technology	No. of units	Steam press. (bar)	Steam temp. (°C)	Thermal efficiency (%)
CCSS	3	94	521	**35.4***
ST	498	60	482	**33.9**
ST/C	60	75	488	**34.5**
ST/CP	4	79	513	**34.7**
ST/S	113	60	500	**34.1**

B. Rankine cycle, SUBCRITICAL TURBINE PRESSURE, WITH REHEAT (Figure 4.1.2b)

Technology	No. of units	Steam pres. (bar)	Steam temp. (°C)	Reheat temp. (°C)	Thermal efficiency (%)
CCSS	3	106	536	536	**38.1***
ST	859	135	538	538	**38.3**
ST/C	32	113	540	540	**38.1**
ST/S	68	128	537	537	**37.8**

C. Rankine cycle, SUPERCRITICAL TURBINE PRESSURE, WITH REHEAT (Figure 4.1.2c)

Technology	No. of units	Steam press. (bar)	Steam temp. (°C)	Reheat temp. (°C)	Thermal efficiency (%)
ST	119	241	538	538	**38.6**
ST/S	9	236	565	565	**39.3**

* The thermal efficiency estimated for the power plants with combined cycle single shaft (CCSS) technologies refer to the thermal efficiency of the second cycle, that is the Rankine cycle. In principle, the total thermal efficiency of the combined cycle (Brayton cycle followed by Rankine cycle) is higher, and can reach the orders of 60%.

4.1.2.4 Estimation of heat rejected to freshwater bodies

In the final step of the calculations of thermal emissions to freshwater, the gross electrical generating capacity of each unit is adjusted to reflect the thermal output of the cycle, by accounting for the mechanical, η_m, and electrical, η_e, efficiencies of the system (assumed to be 0.95 and 0.98, respectively). Furthermore, a conservative approach is adopted, in that all heat not converted to electrical power is taken as rejected to freshwater; in practice, some waste heat will also be emitted to air. Accordingly, the freshwater thermal emissions, $Q_{freshwater}$, are calculated for each power plant via Equation 4.1.8:

$$Q_{freshwater} = \frac{\frac{W_{gross}}{\eta_m \cdot \eta_e}(1 - \eta_T)}{\eta_T} \quad (MJ/s) \qquad (4.1.8)$$

where W_{gross} is given for each individual power plant in the database, and η_T has been calculated for each subgroup of power plants via the Rankine cycle, according to their technology (Table 4.1.2a)[11].

All freshwater thermal emission flows were converted to cumulative annual values (MJ/yr) for the CREEA database.

4.1.2.5 WEPP database-CREEA database country and industry alignment

The final steps of the data preparation involve assigning the 48 CREEA country codes to the countries, as presented in the WEPP database, followed by distributing the results to the relevant industrial sectors in CREEA, namely:

- Production of electricity by coal i40.11.a A_POWC
- Production of electricity by gas i40.11.b A_POWG
- Production of electricity by nuclear i40.11.c A_POWN
- Production of electricity by petroleum and other oil derivatives i40.11.f A_POWP
- Production of electricity by biomass and waste i40.11.g A_POWB
- Production of electricity by geothermal i40.11.k A_POWM

Table 4.1.2b shows the distribution of power plants in the WEPP database (with once-through freshwater cooling systems, operational in 2007) according to their fuel, as well as which CREEA industrial sector they were placed in.

[11] The freshwater thermal emissions estimated from the power plants with a combined cycle single shaft technology (CCSS) are overestimated, since the gross generating capacity quoted for these plants in the WEPP database refers to the aggregated capacity resulting from both the gas and steam turbines. With no way of separating the two, the approach adopted in this work was considered to result in a conservative estimate (or a worst-case scenario with more freshwater thermal emissions being estimated than actually taking place).

Table 4.1.2b Alignment of WEPP database fuel groups to CREEA industrial sectors

WEPP database fuel category		**Number of units**	**Corresponding CREEA industrial sector:** Production of electricity by…
BAG	Bagasse	3	biomass and waste
BFG	Blast-furnace gas also converter gas or LDG or Finex gas (approx 10% of the heat content of pipeline gas)	10	gas
BIOMASS	Biomass excluding wood chips but including agricultural waste and energy crops	1	biomass and waste
COAL	Coal	1017	coal
COKE	Petroleum coke	1	petroleum and other oil derivatives
GAS	Natural gas	333	gas
GEO	Geothermal	9	geothermal
OIL	Fuel oil	138	petroleum and other oil derivatives
PEAT	Peat	7	biomass and waste
REF	Refuse (unprocessed municipal solid waste)	31	biomass and waste
SHALE	Oil Shale	9	petroleum and other oil derivatives
UR	Uranium	97	nuclear
WOOD	Wood or wood-waste fuel	18	biomass and waste
WSTH	Waste heat[12]	94	gas

4.2 Nitrogen and phosphorus emissions

Human activities such as fertilizer production and use, fossil fuel combustion, and cultivation of leguminous crops have more than doubled the rate at which biologically available nitrogen enters the terrestrial biosphere compared to preindustrial levels (Galloway et al., 2004). The inputs of P to the environment over natural, background P from weathering have more than doubled due to human actions such as mining and use of rock phosphate as fertilizer, detergent additives, animal feed supplement and other technical uses (Bennett et al., 2001; Mackenzie et al., 1998). Large fraction of the anthropogenically mobilized N and P enter ground and surface water and are transported by rivers to coastal seas (Galloway et al., 2004; Bouwman et al., 2009; Seitzinger et al.,

[12] The category of fuel termed 'Waste heat' appears in power plants with combined cycle technologies, involving a gas turbine, followed by a steam turbine fuelled by the hot exhaust of the gas turbine. In the overwhelming majority of the cases examined here the primary fuel for the gas turbine was gas, this WEPP database fuel group is assigned to 'production of electricity by gas' in the CREEA database.

2010; Kanakidou et al., 2012). N and P lost from agricultural soils can cause groundwater pollution, eutrophication of lakes, rivers and coastal zones, loss of biodiversity, hypoxia and fish kills (Vitousek et al., 1997; Carpenter et al., 1998; Tilman, 1999; Bennett et al., 2001; Diaz and Rosenberg, 2008; Seitzinger et al., 2010).

As part of the CREEA project, we carried out an assessment of the global N and P balances on croplands with a spatial resolution of 5 arc-minute (~ 10 x 10 km near the equator) for the period 2007. The estimate of the global N and P emission to water were estimated per CREEA country and product classification. The method followed in estimating the nutrient emission is presented below.

Annual soil nutrient balances include the Nitrogen (N) and Phosphorous (P) inputs and outputs at 5 by 5 arc minute spatial resolution. For nitrogen, there are four inputs elements which include application of artificial fertilizer (IN_{fer}) and animal manure (IN_{man}), wet and dry atmospheric deposition (IN_{dep}), biological N fixation (IN_{fix}). The output in the N balance include N withdrawal from the field through crop harvesting (OUT_{harv}), nitrogen output from crop residues (OUT_{res}) and gaseous losses (OUT_{gas}). For phosphorous, the same approach was followed, with P inputs being artificial fertilizer and animal manures. The output in the P balance include P withdrawal with harvested crop and P withdrawal from crop residues. Figure 4.3.1 shows the main elements of the soil surface N and P balance.

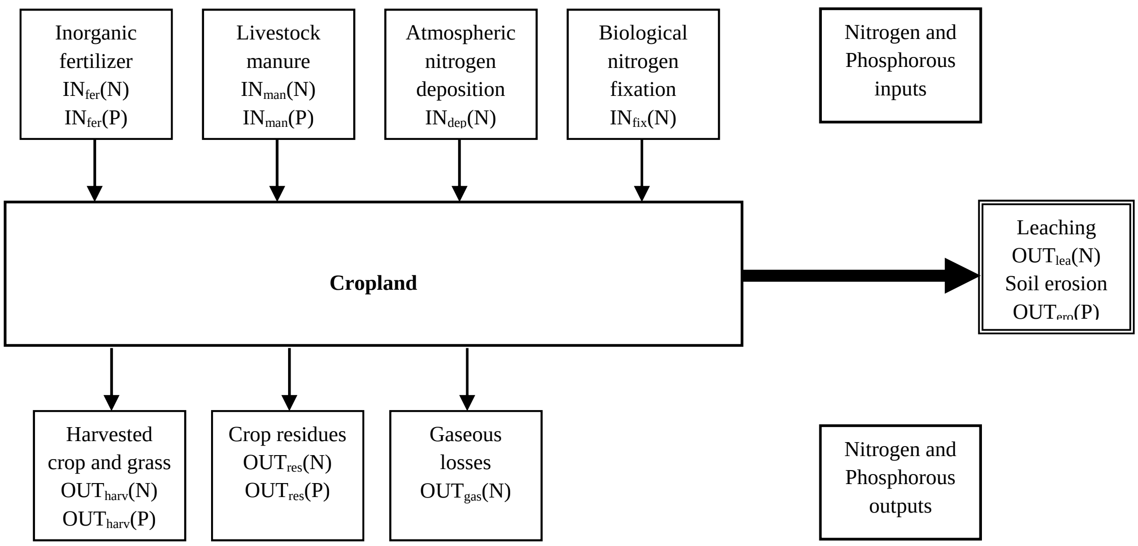

Figure 4.3.1. Main elements of the soil N and P balance

The input and output for N:

$$IN[N] = IN_{fer}[N] + IN_{man}[N] + IN_{dep}[N] + IN_{fix}[N] \quad (4.2.1)$$

$$OUT[N] = OUT_{harv}[N] + OUT_{res}[N] + OUT_{gas}[N] \quad (4.2.2)$$

The input and output for P:

$$IN[P] = IN_{fer}[P] + IN_{man}[P] \tag{4.2.3}$$

$$OUT[P] = OUT_{harv}[P] + OUT_{res}[P] \tag{4.2.4}$$

4.2.1 Calculation of the individual input and output components:

Inputs from mineral fertilizers (IN_{fer}):

The fertilizer application rate per crop per country was calculated using three sources of fertilizer data and the spatially explicit data on crop distribution from Monfreda et al. (2008). IFA/IFDC/IPI/PPI/FAO (2002) provide fertilizer application rate per crop for 88 countries. FAO (2012a) and Heffer (2009) were used to complement data for crops and countries missing from the IFA/IFDC/IPI/PPI/FAO (2002) data. Since the application rates provided in these data sources is for different years, these were adjusted to fit FAO (2012b) country average nutrient fertilizer consumption per year for the period 2002-2009.

Inputs from animal manures (IN_{man}):

Total manure nutrients (N and P) production within the grazing, mixed and industrial animal production systems for the major livestock categories (cattle, buffaloes, sheep, goats, pigs and poultry) was calculated by multiplying the spatially-explicit global livestock density with animal-specific excretion rates then adjusted for the fraction of manure available for cropland and grassland application (Bouwman et al., 2009, 2011; Liu et al., 2010; MacDonald et al., 2011).

The manure nutrients production per production system per animal category per grid cell (M_{exc}, kg of N or P per ha of grid cell) was calculated as follows:

$$M_{exc}[a,s] = D[a,s] \times E[a,c,s] \tag{4.2.5}$$

where *D[a, s]* is the density of animal category *a* for production system *s* (head/ha of grid cell) and *E[a,c, s]* the nutrient excretion rate of animal category *a* in country *c* and production system *s* (kg of N or P per head of animal).

To calculate the manure (N and P) excretion rate per animal category per country we followed the approach of Liu et al. (2010). Sheldrick et al. (2003) provide data on animal manure excretion rates for cattle, pigs, sheep, goats, and poultry relative to the animals slaughter weight. The manure excretion rate per animal category, production systems and country was calculated by combining Sheldrick et al. (2003) global average manure excretion rates with slaughter weight of animals per production systems and per country:

$$E[a,c,s] = \frac{SW[a,c,s]}{SW_{shel}[a]} \times E_{shel}[a] \quad (4.2.6)$$

where $SW[a,c,s]$ is the slaughter weight of animal category a (kg/head) in country c and production system s, $SW_{shel}[a]$ the global average slaughter weight of animal category a (kg/head) and $E[a]$ the global average manure excretion by animal category a (kg/yr) both obtained from Sheldrick et al. (2003). The slaughter weights ($SW[a,c,s]$) of the different animal category per production systems per country was obtained from Mekonnen and Hoekstra (2010b).

We can distinguish three types of manure within each production systems and country (Bouwman et al., 2009, 2011): (a) manure produced from animals housed in stables, (b) manure produced from livestock grazing on pasture or rangeland, and (c) manure excreted for example in urban areas, forests and along roadsides, manure used as fuel or other purposes are considered to fall outside the agricultural systems:

$$M_{exc}[a,s] = M_{stor}[a,s] + M_{graz}[a,s] + M_{out}[a,s] \quad (4.2.7)$$

where $M_{stor}[a, s]$ is volume of manure of animal category a for production system s collected in storage (kg of N or P per ha of grid cell), $M_{graz}[a, s]$ volume of manure of animal category a for production system s that is produced during grazing (kg of N or P per ha of grid cell) and $M_{out}[a, s]$ volume of manure of animal category a for production system s that falls outside the agricultural systems (kg of N or P per ha of grid cell). The fraction of manure that is produced during grazing and the fraction of manure that is not available for spreading on crop and grassland for the different animal category and production systems were obtained from Bouwman et al. (2011).

Not all animal excreta is available as manure to be applied on crops and grassland. The fraction of manure that is produced and available for crops and grassland application depends on a number of factors such as the degree of animal confinement or pasture grazing, cost of transport and agricultural practices (MacDonald et al., 2011). Some manure is also lost during excretion, collection and storage through ammonia (NH_3) volatilization. Therefore, the quantity of manure actually applied on crops and managed grassland (IN_{man}, kg of N or P per ha of grid cell) is:

$$IN_{man}[a,s] = M_{stor}[a,s] - N_{vol,stor}[a,s] \quad (4.2.8)$$

where $N_{vol,stor}[a, s]$ ammonia volatilization from animal housing and storage for animal category *a* and production system *s* (kg/ha of grid cell) and is calculated as follows:

$$N_{vol,stor}[a,s] = N_{stor}[a,s] \times \beta[a,s] \quad (4.2.9)$$

where $N_{stor}[a,s]$ is quantity of N manure of animal category *a* (kg/ha of grid cell) production system *s* in animal housing and storage and $\beta[a,s]$ ammonia volatilization rate of animal category *a* production system *s* (%). According to Bouwman et al. (1997), the volatilization rate for cattle, pigs and poultry is 36% and for buffaloes, sheep and goat 28%.

The available manure which is applied to crops and managed grassland varies from country to country. We used the data on the share of manure applied on cropland and grassland for 23 European countries from Menzi (2002) and for the individual states of the US from Kellogg et al. (2000). We used the average of the 23 European countries value to other EU countries. We also used US average share of manure applied on crops and grassland for other high-income countries including Canada, Australia and Japan. For developing countries we adopted the value provided by Bouwman et al. (2009, 2011): 95% of the available manure is applied on cropland and 5% on grassland. For EU countries we used maximum application rates of 170 kg N/ha/yr based on existing EU nitrates directive.

Inputs from deposition (IN_{dep}):

Atmospheric nitrogen deposition rates (including dry and wet deposition of NH_x and NO_y) for the year 2000 were taken from Dentener et al. (2006). The 30 arc minute original data were converted to a resolution of 5 arc minute.

Inputs from biological fixation (IN_{fix}):

Symbiotic relationship between some nitrogen-fixing bacteria and a variety of leguminous plants converts dinitrogen gas (N_2) to plant-available forms of N. Some free-living bacteria are also capable of biological N fixation. Following Bouwman et al. (2009), total nitrogen fixation by leguminous crops was estimated by multiplying the N in the harvested product by a factor of two to account for all above and belowground plant parts. Nitrogen fixation by cyanobacteria in irrigated rice ranges from 20 to 30 kg per hectare during the growing seasons (Smil, 1999). In this study we used an average value of 25 kg of N per hectare. For nonleguminous crops, the nonsymbiotic biological N_2 fixation rate is assumed to be 5 kg of N per hectare (Bouwman et al., 2009).

Outputs from harvested crop and grass (OUT_{harv}):

Nutrient (N and P) withdrawal by harvested crops is the most important output of nutrients from the soil system. The N and P withdrawal in the harvested crops is calculated by multiplying the crop production by the nutrient (N and P) content of the crops. To calculate the N and P withdrawal by harvested grass and grass consumption, we adopted the method of Bouwman et al. (2009, 2011).

Nutrient loss through harvested crop (OUT_{harv}, kg per ha of grid cell) is calculated by aggregating the nutrient withdrawal from each crop harvested and adding the nutrient withdrawal due to grass consumption and harvest as follows:

$$OUT_{harv} = \sum_{p=1}^{m} \left(Y[p] \times np[p] \right) \tag{4.2.10}$$

where *Y [p]* is the yield of crop *p* (ton/ha) and *np[p]* nutrient content of crop *p* (kg/ton.

The crop yields at 5 arc minute spatial resolution were obtained from Mekonnen and Hoekstra (2010a, 2011). The N and P contents of major crop were taken from IPNI (2012). For other crops and crop groupings values from FAO (2004) and Roy et al. (2006) were used. For nuts and spices, we have adopted the values of fruits and vegetables respectively from FAO (2004).

Outputs from crop residues (OUT_{res}):

Part of the crop residues is removed from cropland and used, for example, as biofuel or for animal feeding. The nutrients withdrawal with crop residue (OUT_{res}, kg N and P per ha) was calculated by multiplying the yield of crop residue by the nutrient content of the crop residue and adjusting this by a removal factor:

$$OUT_{res} = CR[r] \times np[r] \times \gamma[r] \tag{4.2.11}$$

where *CR[r]* is volume of crop residue *r* (ton/ha), *np[r]* nutrient content of residue *r* (kg N and P per ton of crop residue), *γ[r]* removal factor of the crop residue *r*. The nutrient content of the crop residues were taken from mainly from IPNI (2012) and FAO (2004) and for few crops from Roy et al. (2006). Missing values for nuts and spices were filled by adopting the values of fruits and vegetables respectively from FAO (2004). The volume of crop residue was calculated by multiplying dry crop yield with a residue-to-product ration (RPR). The RPR values for large number of crops and crop groupings were obtained from Eisentraut (2010). For spices, we took the RPR values of vegetables. The crop residue removal factors in Ghana, Kenya and Mali for various crops were obtained from Lesseschen et al. (2004). Removal factor for 18 crops or crop groups in India were derived from Ravindranath et al. (2005). For other crops which are not covered by Ravindranath et al. (2005), we used the average removal factor of the 18 crops. For the USA, crop residue removal factors for maize and wheat for large number of states were obtained from Graham et al. (2007) and Nelson et al (2002) respectively. Removal factors of maize and wheat in other states were taken as the average removal factor in the states with data. For other crops, the residue removal factors in the USA were adopted from Perlack et al. (2005). For other countries with no data, removal factors were adapted from Krausmann et al. (2008) who have provided residue removal factors per major crop groupings and geographic regions.

Outputs from gaseous (OUT_{gas}):

Large quantity of nitrogen is lost from animal manures and fertilizers by volatilization of NH_3 (Smil, 1999). We adopted the empirical model of Bouwman et al. (2002a) to calculate ammonia volatilization from the application of animal manure and N fertilizers. The empirical model takes into account the influence of crop type, fertilizer type, manure or fertilizer application mode, soil cation exchange capacity (CEC), soil pH and climate on ammonia volatilization.

Nitrogen loss through gaseous emission (OUT_{gas}, kg/ha) is the sum of NH_3 volatilization and N_2O-N or NO-N emission:

$$OUT_{gas} = N_{vol,spr} + N_{emission} \tag{4.2.12}$$

where $N_{vol,spr}$ is NH_3 volatilization (kg/ha) during spreading of manure on the field and $N_{emission}$ the emission of N_2O-N or NO-N (kg/ha).

Following Bouwman et al. (2002a), NH_3 volatilization ($N_{vol,spr}$, kg/ha) during spreading of fertilizer and manure is calculated as follows:

$$N_{vol,spr} = IN_{fer,man} \times \varphi \tag{4.2.13}$$

where $IN_{fer,man}$ is fertilizer/manure application rate (kg/ha) and φ NH_3 volatilization rate and is calculated as:

$$\varphi = \exp(\textit{factor value for crop type} + \textit{fertilizer type} + \textit{application mode} + \textit{soil pH} + \textit{soil CEC} + \textit{climate}) \quad (4.2.14)$$

where the factor values (for crop type, fertilizer type, application mode, soil pH, soil CEC and climate) were taken from Bouwman et al. (2002a). To estimate the NH_3 volatilization, we grouped the crops into rice and other crops following Bouwman et al. (2002a).

Nitrification (oxidation of NH_4^+) and denitrification (reduction of NO_3^- or NO_2^-) are the main sources of NO_x and N_2O emitted from the soil (Smil, 1999). Generally, soil denitrification occurs in or just below the root zone under high soil water content and limited oxygen availability, forming N_2, N_2O and NO (Van Drecht et al., 2003). In this study we followed Bouwman et al. (2011) and estimated the three gases (N_2, N_2O and NO) separately.
Denitrification (emission of N_2) in soil is calculated as a fraction of the available surplus nitrogen after accounting for nitrogen withdrawal with harvested crop, crop residue and ammonia volatilization (Van Drecht et al., 2003):

$$N_{denitrification} = f_{den} \times N_{surplus} \quad (4.2.15)$$

where $N_{surplus}$ is the nitrogen surplus which is calculated as the difference between nitrogen surface balance and ammonia volatilization. The nitrogen surface balance is calculated as the difference between the total nitrogen input (*IN[N]*) and the nitrogen uptake by crops ($=OUT_{harv} + OUT_{res}$). The denitirifiation fraction (f_{den}) is calculated with a model that combines the effect of temperature, crop type, and soil and hydrological conditions (Van Drecht et al., 2003):

$$f_{den} = \min[(f_{climate} + f_{text} + f_{drain} + f_{soc}),1] \quad (4.2.16)$$

where $f_{climate}$ represents the effect of climate on denitrification rates, f_{text}, f_{drain} and f_{soc} are factors representing effects of soil texture, soil drainage and soil organic content on denitrification rates respectively. For rice the f_{den} is set at 0.75. The climate factor ($f_{climate}$) was estimated following Van Drecht et al. (2003) and the factors f_{text}, f_{drain} and f_{soc} were adopted from Van Drecht et al. (2003) for the respective soil parameters. The soil texture, drainage class and SOC were obtained from the derived soil properties on a 5×5 arc-minute global grid (version 1.2) from ISRICWISE (Batjes, 2012)

According to Bouwman et al. (2002b), the major factors influencing the emission for N_2O include nitrogen application rate, crop type, climate, soil organic carbon (SOC) content, soil texture, drainage and soil pH, and for NO they include nitrogen application rate, SOC and soil drainage. To estimate the NO and N_2O emission, we grouped the crops into four groups (i.e, rice, legumes, grass and other crops) and applied the statistical model developed by Bouwman et al. (2002b). Following Bouwman et al. (2002b), the emission of N_2O-N or NO-N ($N_{emission}$, kg/ha) is calculated as follows:

$$N_{emission} = IN_{fer,man} \times \exp\left(constant + \sum_{i=1}^{n} Factor\ class(i) \right) \tag{4.2.17}$$

where the *constant* and the *Factor classes* (for N_2O: N rate*fertilizer type, crop type, climate, SOC, soil texture, drainage and soil pH; for NO: N rate*fertilizer type, SOC and soil drainage) were adopted from Bouwman et al. (2002b).

Nutrients leached or runoff to the water system:

For nitrogen, the quantity of nitrogen leached to the water system is the difference between the input and output:

$$Leaching[N] = IN[N] - OUT[N] \tag{4.2.18}$$

For phosphorus, following Bouwman et al. (2011) the amount of nutrient emitted as runoff to the water system is assumed to be 12.5% of the phosphorus input from of fertilizer and manure application.

4.2.2 Data aggregation and allocation to CREEA classification

The grid level emission data of N and P were first aggregated to country level using the country polygon shapefile. This excercise generate estimats of N/P emission to water at a detailed country and crop level. We have studied 206 individual countries and 146 crops. On the other hand CREEA's classification provide 43 individual countries and 5 major regions. The crops are further grouped into 13 CREEA product and industry classes. The final emission data were provided after aligning our detailed level of data to CREEA country and product classification.

5. From emission to impacts

5.1 Comparison of methodologies for the environmental assessment of nutrient emissions: life cycle impact assessment and grey water footprint methodologies[13]

5.1.1 Eutrophication: concepts, causes and effects

Eutrophication can be defined as "an increase in the rate of supply of organic matter to an ecosystem" (Nixon 1995) and it is one of the most severe problems related to water quality (Carpenter et al. 1998; Howarth et al. 2002), with eutrophication incidents being thought of "getting more common and more severe" (Dyhr-Nielsen et al. 2012).

Eutrophication is thus the effect of excess nutrient inputs, mainly nitrogen and phosphorus, in inland, coastal and marine waters. Human activities have significantly intensified the problem, as they have altered the natural N cycle (Galloway 1998; Howarth et al. 2002; Vitousek et al. 1997) and caused an important increase in P fluxes to the ocean (Howarth et al. 2002). The main sources of these nutrients are synthetic fertilizers and manure applied in agricultural soil (diffuse sources) and the discharge of urban run-off in water bodies (point sources). In the case of N, a fraction is released in the air during application of fertilizers and manure to the soil and the remaining (after plant uptake) is reaching surface waters through run-off and erosion or leaching from the soil to groundwater. Atmospheric deposition of NH_3 and NO_x is also considered to contribute to eutrophication, mainly in seawater. For phosphorus, only run-off and erosion are considered to be relevant pathways for diffuse P emissions (Goedkoop et al. 2009).

Adverse effects of eutrophication in aquatic ecosystems include changes in biomass, productivity and species composition and loss of aquatic species diversity, shifts in phytoplankton composition to species that may be toxic and decreased concentrations of dissolved oxygen in bottom waters and sediments, leading to hypoxic or anoxic conditions (Camargo and Alonso 2006; Smith et al. 1998). In addition to the ecological effects, eutrophication reduces water clarity and the perceived aesthetic value of water bodies (Smith et al. 1998). Furthermore, eutrophied freshwater can face taste and odour problems, and hinder some water treatment processes, especially filtration, causing clogging of the filters and raising the need for frequent and costly cleaning (Crittenden et al. 2012; OECD 1982). In addition, adverse effects on human health have been reported, especially related to nitrates and nitrites in drinking water (Camargo and Alonso 2006).

[13] This work formed part of the Master's thesis of Anastasia Papangelou (Papangelou, 2012).

Growth of algae is influenced by various factors, e.g. temperature, light and mixing conditions (Lawrence et al. 2000), but in terms of substances, availability of nitrogen and phosphorus is the most decisive factor and all other substances (CO_2, O_2, H_2 etc.) and nutrients (Ca, Na, K etc.) are considered to be abundant or at least non-limiting (Goedkoop et al. 2009). When studying eutrophication, it is common to apply the concept of the limiting nutrient, according to which, only one nutrient is controlling the growth rate of primary producers. In general, phosphorus is considered to be the limiting nutrient in freshwater, while growth in coastal and marine water is limited by the availability of nitrogen (Carpenter et al. 1998; Crouzet et al. 1999; OECD 1982; Smith et al. 1998; Finnveden and Potting 1999). However, the concept of limiting nutrient is being debated as too simplistic and the need to assess and control both nutrients is recognised (Lewis et al. 2011; Dodds 2007).

5.1.2 Methods to assess eutrophication

Eutrophication assessment of water bodies, by relating levels of nutrients with indicators such as Chlorophyll-ɑ concentration and classifying them according to their trophic state has received substantial attention by research (Dodds 2007; Nixon 1995; Nürnberg 1996; OECD 1982). However, the assessment of nutrient emissions generating over a product's life cycle, a specific process, a consumer or a group of consumers is also of high relevance. Life Cycle Impact Assessment (LCIA) and the Grey Water Footprint (WF_{grey}) are methods that can be used for such environmental assessments.

5.1.2.1 Water Footprint

The Water Footprint (WF) concept was first introduced in the early 00's, as an indicator to account for the direct and indirect freshwater use associated with a producer or a consumer (Hoekstra et al. 2011). It includes three components, the blue, green and grey water footprint. Blue water footprint (WF_{blue}) refers to consumption of surface and groundwater, green water footprint (WF_{green}) to consumption of rainwater minus run-off and grey water footprint (WF_{grey}) to pollution of water bodies. A full WF assessment study includes four steps (Hoekstra et al. 2011):

1. Setting goals and scope
2. Water Footprint accounting
3. Water Footprint sustainability assessment
4. Water Footprint response formulation

The outcome of WF accounting is a volume of water representing the consumptive and degradative water use of a product along its whole supply chain, of a consumer, a community, a company or a geographical area for a given period of time. With WF sustainability assessment, this water volume is compared with the available freshwater resources, in a similar manner as with ecological footprint (Hoekstra et al. 2011).

Grey water footprint is the component of WF accounting for pollution and it is defined as the volume of water needed to dilute pollutants, while water quality of the receiving body stays above agreed quality standards. This is achieved by taking into account the assimilation capacity of the water body, i.e. the difference between the maximum allowable concentration of the pollutant and its respective natural concentration in the water body. The environmental sustainability assessment for WF_{grey} is performed by dividing the estimated water volume with the amount of water available for the assimilation of the pollutant, which on a river catchment scale, corresponds to the actual run-off in the catchment, Q_{act} (Hoekstra et al. 2011).

Many studies have been recently published on grey water footprint accounting and assessment, especially of nutrient emissions related to crop production, e.g. (Chapagain and Hoekstra 2011; Liu et al. 2012; Mekonnen and Hoekstra 2010c, 2011)

5.1.2.2 LCIA

- **General framework of LCIA**

LCIA is the third phase of a Life Cycle Assessment (LCA) study and it consists of mandatory (selection of impact categories, indicators and characterization methods, classification and characterization) and optional elements (normalization, grouping, weighting and data quality analysis) (ISO 14042 2006). In the context of this study, where focus is already on one particular impact category (eutrophication of aquatic ecosystems), emphasis will be given on characterization, and by LCIA, the characterization step of LCIA will be mostly implied.

With characterization, environmental interventions (in this case nutrient emissions in the environment) are translated into impact category indicators, to the midpoint (e.g. concentration of nutrients in the water bodies), or to the endpoint level (e.g. loss in species in the aquatic ecosystems) (de Haes et al. 1999). This is achieved with the use of characterization factors, which describe a linear relationship between the impact indicators and the respective emissions (Equation 5.1.1) (Pennington et al. 2004):

$$Category\ Indicator = \sum_s Characterization\ Factor\ (s)\ \times Emission\ Inventory\ (s),\ s{:}\ chemical \quad (5.1.1)$$

- **Developments in eutrophication characterization models**

The currently proposed model framework for the characterization of eutrophication is described by Equation 5.1.2 (EC-JRC 2010):

$$Characterization\ Factor\ (CF) = Fate\ Factor\ (FF) \times Effect\ Factor\ (EF) \tag{5.1.2}$$

Within this framework, different approaches have been proposed for the calculation of a characterization factor (CF) for eutrophication. (Huijbregts and Seppälä 2001) introduced a dimensionless fate factor (FF) representing the fraction of a compound emitted that reaches the aquatic environment and an effect factor (EF) describing the potential phytoplankton biomass production per mass unit of the emitted compound (in kg PO_4^- -eq·kg^{-1}). Consequently, they came up with a set of characterization factors for N and P emissions to air, water and soil for the Netherlands, West Europe and the world. (Seppälä et al. 2004) presented a similar characterization model and calculated characterization factors for nutrient emissions from different sectors in Finland. In this model, the CF is the product of a dimensionless transport factor, η (representing the fraction of a water area potentially affected by a given emission, E), a dimensionless effect factor, μ (representing the fraction of the transported compound causing increase in biomass production) and an equivalency factor, Eqv, expressed in kg PO_4^--eq·kg^{-1} of emitted substance. (Gallego et al. 2010) adapted this model to derive regional characterization factors for aquatic eutrophication, using Galicia, Spain as a case study. In their work, emphasis is given in transport and the effect factor is set to 1.

The need for regional characterization factors was recognised earlier and several LCIA methods include regional CFs for aquatic eutrophication. EDIP2003 provides both site-generic and site-dependent factors for 32 European countries (Hauschild and Potting 2005), in LUCAS the 15 Canadian ecozones are used as the spatial resolution unit (Toffoletto et al. 2007), while in TRACI the different states of the USA (Norris 2002).

However, the above mentioned models stop relatively early in the cause-effect chain, providing CFs to the midpoint level and recognizing the need for the introduction of factors that will assess eutrophication to the damage level. In ReCiPe (Goedkoop et al. 2009) an effect factor expressed in Potentially Disappeared Fraction (PDF) of species is presented for phosphorus emissions in freshwater. The effect factor was derived studying the relationship between the number of macrofauna species occurring in Dutch freshwater systems and the respective phosphorus concentration, C_P. In a more recent study, (Struijs et al. 2011b) studied the relationship between the occurrence of macro-invertebrate genera and C_P in Dutch inland waters and developed CFs to the endpoint level for different phosphorus sources on European level (Struijs et al. 2011a). Finally, (Azevedo et al. In prep.-a; Helmes et al. 2012) present spatially explicit characterization factors to the damage level for P emissions on a global scale.

Despite the developments in the modelling of impacts of phosphorus emissions in freshwater, an effect factor for nitrogen in coastal and marine waters is currently missing. In addition, brackish waters have not drawn much attention so far and are mostly addressed together with freshwater. However, they represent a special case of waters as they can be either N- or P-limited or both (Finnveden and Potting 1999). However, there is equivalence factor for N emissions relating N to P (Guinee 2001).

5.1.3 Research Objectives

A study comparing LCA and WF as methods to assess potential impacts of products on water consumption has been recently published (Jefferies et al. 2012). Jefferies et al. studied tea and margarine along their life cycles and found that results of the two methods at the inventory level are quite similar, when similar data sources are used, but key differences occur in the impact assessment. However, no similar study exists that compares the two methods with regard to water pollution and more specifically, eutrophication. In addition, it would be interesting to compare the two methods in a case study different than the one of a product.

In this work, a country, the Netherlands, is employed as a case study for the comparative assessment of WF_{grey} and LCIA when assessing nutrient emissions in freshwater. The Netherlands was chosen as a country where aquatic eutrophication is highly relevant, since its land is intensely cultivated and it lies in the mouth of four major international rivers. In addition, data on pollutant emissions are abundant in the Netherlands. In order to compare the performance of the methods when data are relatively scarce, the assessment was also done for Greece.

Though not strictly in line with LCIA (the concept of life cycle is not relevant in the case of a country), the environmental assessment of national nutrient emissions can still be performed by using LCIA characterization models. The latest models (Azevedo et al. In prep.-a; Struijs et al. 2011a; Helmes et al. 2012) have not been applied in a case study yet, but it would be interesting to compare them in the case study of the Netherlands.

To sum up, the objective of this study is:

- To perform a comparative study of WF_{grey} and LCIA, as methods for the assessment of nutrient emissions in freshwater, and assess the relative advantages and disadvantages of each one regarding completeness, environmental relevance, ease of application and communication of the results etc.

5.2 Methods and study area

5.2.1 Study area

The Netherlands lies on the Delta of four international rivers (Rhine, Scheldt, Ems and Meuse) and is part of the North Sea catchment area. The fast demographic and economic growth of the country in the twentieth century, together with modernization and intensification of agriculture, have caused the deterioration of water quality (RIZA 2002) and eutrophication has been a severe problem in the country for several decades (Van der Molen and Portielje 1999). The first piece of environmental legislation came into act in the 70's (RIZA 2002) and after the implementation of several national and international plans nitrogen and phosphorus concentrations have decreased in the majority of Dutch lakes during the period 1980 – 1996 (Van der Molen and Portielje 1999). Currently, water pollution is controlled also as a requirement of the Water Framework Directive (WFD), however, excessive nutrient inputs remain a key problem of Dutch waters (Anon 2010).

ıds

5.2.2 Methods

5.2.2.1 Data on nutrient emissions in freshwater for the Netherlands

Data on nutrient emissions in Dutch surface waters were derived from the Netherlands Pollutant Release and Transfer Register (PRTR). In the register, the user can choose among different datasets for a variety of pollutants, environmental compartments and emissions sources (activities). Emissions can be displayed allocated to communities, water catchments, river basins or a 5 km x 5 km grid. The register provides emission data for the years 1990, 1995, 2000, 2005, 2008, 2009, 2010 and estimations for 2011.

For the purposes of this study, emissions of eutrophying substances (total nitrogen and total phosphorus) released directly in surface waters were used. As shown in Figure 5.1.2, the compartment "Discharge load to surface waters" covers all the possible pathways of substances from their source until they reach water. This simplifies to a large extent the calculation of WF_{grey} especially, as no distinction between point and diffuse sources of nutrients has to be made.

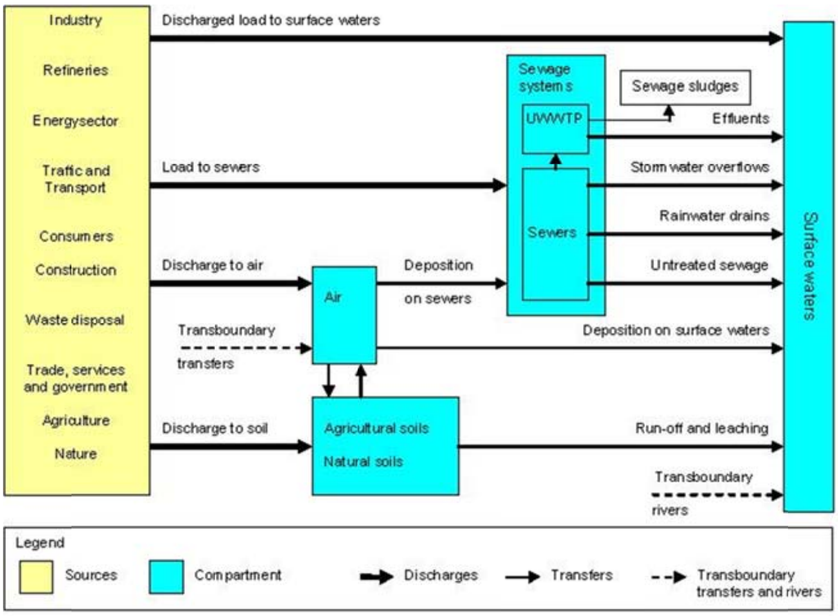

Figure 5.1.2 Relationship between the different water compartments (Source: Dutch PRTR http://www.emissieregistratie.nl/ERPUBLIEK/content/explanation.en.aspx accessed 31.08.2012)

The complete datasets of phosphorus emissions as downloaded from the PRTR and used in this study are given in Appendix 5.1.1.

5.2.2.2 Data on nutrient emissions in freshwater for Greece

Data on nutrient emissions in Greece are not as readily available as they are for the Netherlands. Therefore, emissions from point and diffuse sources had to estimated, based on available data from literature.

- **Diffuse sources**

Data on phosphorus application rates from manure and fertilizer were retrieved from the OECD iLibrary. (OECD 2010). The OECD datasets include data on gross phosphorus application rates for the period 1985 – 2004. In order for the assessment for the two countries to be comparable, data for the years 1998 – 2004 were extrapolated, to predict the application rates of both manure and fertilizers for the years 2005 – 2009 (Figure 5.1.3).

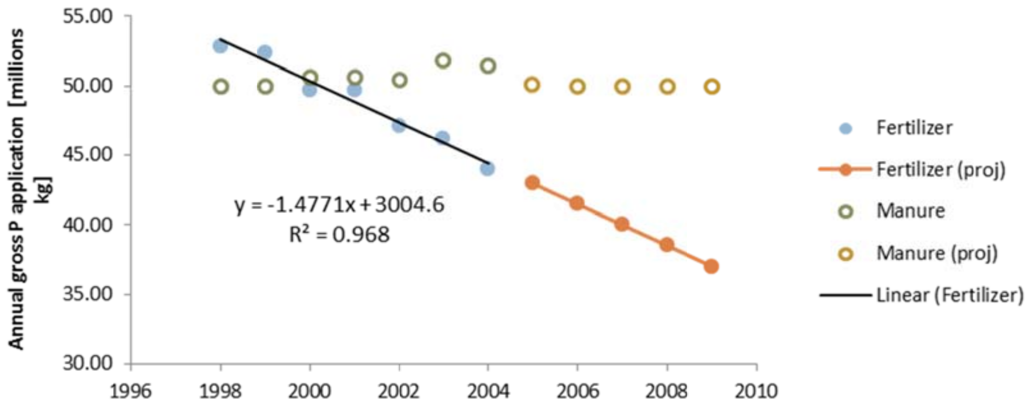

Figure 5.1.3 Actual data for P application rates from manure and fertilizers in Greece and extrapolation for the years 2005 – 2009

- **Point Sources**

The estimation of phosphorus reaching Greek surface waters as effluent from sewers and WWTPs was based on Equation 5.1.3. For this calculation it is assumed that all WWTPs are normal activated sludge systems. In addition, it is assumed that 21% of the non-treated sewage is discharged in freshwater (same percentage as for the treated effluent). Explanation of the symbols used in Equation 5.1.3 with the respective values and sources are listed in Table 5.1.1.

$$L_{P,point}\,[kg/yr] = \left[PE \cdot f_{WWTP} \cdot \eta_{P\,rem} \cdot f_{fw} \cdot \alpha_P + PE \cdot (1 - f_{WWTP}) \cdot f_{fw} \cdot \alpha_P\right] \cdot 10^{-3} \cdot 365 \qquad (5.1.3)$$

Table 5.1.1 Symbols and values[14] used for the estimation of P point emissions in freshwater in Greece (Equation 5.1.3)

Symbol	Description	Unit	Value	Source
$L_{P,point}$	Phosphorus load to freshwater from point sources	kg/yr		-
PE	Population equivalents	cap.	11’260’402	(EUROSTAT)
f_{WWTP}	Percentage of population connected to a WWTP	%	67	(OECD)
$\eta_{P,rem}$	P removal efficiency in normal activated sludge systems	%	20	(Henze et al. 2008)
f_{fw}	Percentage of WWTPs discharging in freshwater and estuaries	%	21	(Y.P.E.K.A 2009)
α_P	Specific P production in Greece	$g \cdot cap^{-1} \cdot d^{-1}$	1.5	(Metcalf&Eddy 2003)

14 Values presented in Table 5.1.1 are for 2009. For a complete table with all values along time see Appendix 5.1.2

5.2.2.3 Grey water footprint

The grey water footprint within the Netherlands was calculated following the methodology as set out in the Water Footprint Assessment Manual (Hoekstra et al. 2011). The WF_{grey} within a nation (or WF_{grey} of national production) is given by Equation 5.1.4 (Hoekstra et al. 2011):

$$WF_{grey,area,nat} = \sum_q WF_{proc}[q] \qquad (5.1.4)$$

where the $WF_{proc}[q]$ is the grey water footprint of a single process, q, and is given by Equation 5.1.5:

$$WF_{proc,grey} = \frac{L}{c_{max} - c_{nat}} \quad \left[m^3/yr\right] \qquad (5.1.5)$$

L : pollutant load [kg/yr]

c_{max} : maximum acceptable concentration for the pollutant [kg/m^3]

c_{nat} : natural concentration of the pollutant in the water body[kg/m^3]

The estimation of pollutant loads from point sources is straightforward and without any problems when relevant data are available. However, for diffuse sources of pollutants, modelling techniques are required for the estimation of the loads reaching the water bodies. In the WF manual a three-tier approach is recommended for estimating diffuse pollution loads; tier I (default method, Equation 5.1.6) assumes a fixed fraction of the applied chemical reaching freshwater, while tier II and III use simple and more sophisticated model approaches respectively.

$$L_{diffuse} = \alpha \cdot Q_{applied} \qquad (5.1.6)$$

$L_{diffuse}$: pollutant load from diffuse sources[kg/yr]

α : fraction of chemical applied to soil reaching freshwater [-]

$Q_{applied}$: application rate of chemicals to the soil [kg/yr]

In the case of the Netherlands, net emissions of nutrients in surface waters are given for all sources (point and diffuse). However, no such data are available for Greece, so values for α had to be assumed based on literature. Two different α values were used for the assessment of WF_{grey} in Greece: 10% (Chapagain et al. 2006) and 5% (Powers 2007).

The difference c_{max} – c_{nat} (Equation 5.1.5) is also called the dilution factor. The maximum acceptable concentration of a substance, c_{max}, is based on water quality standards usually set by policy makers. The natural background concentration, c_{nat}, refers to the concentration of the specific pollutant in the water body, if human disturbance had never occurred. c_{max} and c_{nat} were derived from literature and different combinations of c_{max} and c_{nat} were used for the assessment of the WF_{grey} for the Netherlands and Greece (Table 5.1.2).

Table 5.1.2 c_{max} and c_{nat} according to different literature sources

Source	TP [mg/L]		TN [mg/L]		Comment
	c_{max}	c_{nat}	c_{max}	c_{nat}	
(Liu et al. 2012)	0.95	0.52	3.1	1.5	Values globally applicable
(RIZA 2002)	0.15	0.05	2.2	1	For c_{nat} the respective “target values” are presented
(Laane et al. 2005)	0.15	0.05		1	Values for the Netherlands
(Smith et al. 2003)		0.023		0.14	Values for the USA
(Mekonnen and Hoekstra 2011)			10	0	c_{max} as NO_3-N
(FEK 2010)	0.31	0.015			Values for Greece

In order to get an idea of the size of the water footprint, one has to compare it with the water available to assimilate the given pollutant load. In the WF manual this procedure is called the (environmental) sustainability assessment of the WF_{grey}, and it is best performed for a whole catchment area or river basin. In this case, the WF_{grey} is divided with the actual run-off of the catchment (Q_{act}), to give the Water Pollution Level (WPL), which is a measure of the waste assimilation capacity consumed (Equation 5.1.7). The actual run offs Q_{act} for the different river basins in the Netherlands (Table 5.1.3) were derived from the Global *NEWS* model (Mayorga et al. 2010).

$$WPL\,[x,t] = \frac{\sum WF_{grey}[x,t]}{Q_{act}\,[x,t]}\,[-] \qquad (5.1.7)$$

Table 5.1.3 Actual run-off Q_{act} for the three basins which The Netherlands is part from after (Mayorga et al. 2010)

River Basin	Q_{act} [km^3 / yr]
Rhine	58.47
Maas (Meuse)	12.73
Scheldt	3.86

5.2.2.4 LCIA

The grey water concept allows for a simplified assessment of pollution based on a generic distance-to-target approach. It gives volumes of water as an impact score, which can be interpreted as the maximum water pollution volume possible from a specific emission. However, it does not relate pollution to an environmental impact, since the fate of the emission and ecosystem vulnerabilities are not addressed. These aspects are typically covered in life cycle impact assessment (LCIA) methods as discussed above. A special focus in LCA is considering the fate of emissions, which accounts for residence in each system of concern and exchanges between different environmental compartments.

The emissions to water covered in CREEA are BOD, N and P. Additionally emission from air have impacts on water sources due to deposition and exchanges between air and water, as described for the case of nitrogen.

For freshwater eutrophication, emissions of phosphate and phosphorous are addressed in recommended LCIA methods. The emissions can be addressed as total P with a factor of 56.2 PDF m3 yr per kg P, based on (Goedkoop, Heijungs et al. 2009. Available at http://lcia.wik.is). PDF m3 yr can be interpreted as the volume of freshwater ecosystem which is deprived of all species during a year, i.e. a kg P potentially destroys 56.2m3 of freshwater during a year (or eliminates 10% of species in 56m3 during ten years or in 562m3 during 1 year). Based on the CML method (Guinée, 2001), a kg of total nitrogen emissions is equivalent to 0.14 kg P (resulting in 7.7 PDF m3 yr / kg N). Thermal emissions are so far only addressed for the case of a Nuclear power plant in the Rhine river (in Switzerland, Verones et al. 2011). The impacts can be translated into 3.55E-05 PDF m3 yr / MJ of heat release. While this factor is site-specific, a range of 2.5 E-06 to 2.5 E-04 PDF m^3 yr per MJ is a 95% confidence interval for power plants in the US. This range matches closely with the Swiss case study and hence it seems appropriate to account for heat emissions by applying a factor of ~3E-05 PDF m3 yr / MJ of heat release.

Improved methodologies for the assessment of eutrophication in the framework of LCIA are currently being developed and refined. For the assessment of nutrient emissions in the Netherlands, two newly developed methods were used, as described in (Struijs et al. 2011a), referred to as "Struijs" from now on and (Helmes et al. 2012; Azevedo et al. In prep.-a), ("LC Impact"). Both methods developed factors for the characterization of P emissions in freshwater. Table 5.1.4 lists the CFs for P emissions from point sources; the complete set of CFs is given in Appendix 5.13.

Table 5.1.4 Fate, Effect and Characterization Factors for aquatic eutrophication from P emissions in freshwater (point sources) according to Struijs and LC Impact.

	Struijs et al. 2011a	LC Impact
FF [d]	111	33.7
EF [$DF \cdot m^3 \cdot kg^{-1}$]	203	
CF [$PDF/PNOF \cdot m^3 \cdot d \cdot kg^{-1}$][15]	21’685	95’947
Reference Area	EU – CFs are meant to be site-generic	NL – aggregated CFs from a 0.5° resolution global model

Struijs et al. (2011) are using the CARMEN model to derive a FF for P in freshwater per river basin, in a similar manner as in ReCiPe (Goedkoop et al. 2009). These FFs can then be aggregated to give the FF for the whole of Europe. Struijs et al. are deriving three different characterization factors, for P emissions to soil (from manure and fertilizer) and directly to freshwater (from point sources, namely effluent of WWTPs). As the Dutch PRTR is providing the P emissions from all sources directly to freshwater, only the CF for point sources was used for the assessment of all emissions. An alternative approach would be to multiply the characterization factors for gross P emissions to soil (from manure and fertilizer) with a factor of 19.33 (Struijs et al. 2011a). In the case of Greece, where only the gross application rates of phosphorus were available, the 3 different CFs were used for the assessment.

For the EF (or ecological damage factor, EDF), Struijs et al. correlated the occurrence of invertebrate genera in Dutch surface waters to the concentration of total phosphorus (Struijs et al. 2011b) and calculated an EF for each one of the river basins included in CARMEN (C_P for 1995), adopting the marginal approach. The resulting EF, as given in Table 5.1.4, is the arithmetic mean of the EFs for the different basins.

Helmes et al. developed a new model for the calculation of phosphorus fate on a 0.5° resolution. The fate factor presented in Table 5.1.4 is the aggregated fate factor for the Netherlands, as is the respective characterization factor. In this fate model, not only transport, but also retention and water use are included as processes for P removal. EFs are based on the relationship between the potentially not occurring fractions (PNOFs) of freshwaters species and total phosphorus concentrations (Azevedo et al. In prep.-a; Azevedo et al. In prep.-b). Azevedo et al. calculated different effect factors for different types of waters (streams and lakes) and different aquatic species (auto- and heterotrophs). In addition, they developed four different types of effect factors (linear,

[15] PDF for Struijs et al., PNOF for LC Impact

marginal and average EFs). In Table 5.1.4 the marginal EF for heterotrophs in streams, used in the assessment of the Netherlands, is presented.

5.3 Results

5.3.1 Grey water footprint within the Netherlands

The results of the total WF_{grey} accounting within the Netherlands for the period 1990 – 2009 are presented in Figures 10 – 13. In Figure 5.1.4 the footprints for different combinations of c_{max} and c_{nat} (Table 5.1.2) are shown, illustrating the significant influence the choice of these values has on the final score for the WF_{grey}. The trend over the years is the same for the three footprints, while the absolute values differ by a factor of more than 4 for the two extreme cases. In Figure 5.1.5 the grey water footprints for nitrogen and phosphorus emissions are compared. The WF_{grey} of nitrogen has also decreased during the years, but not as sharply as the one of phosphorus and after 1990 it is steadily bigger than the WF_{grey} for P.

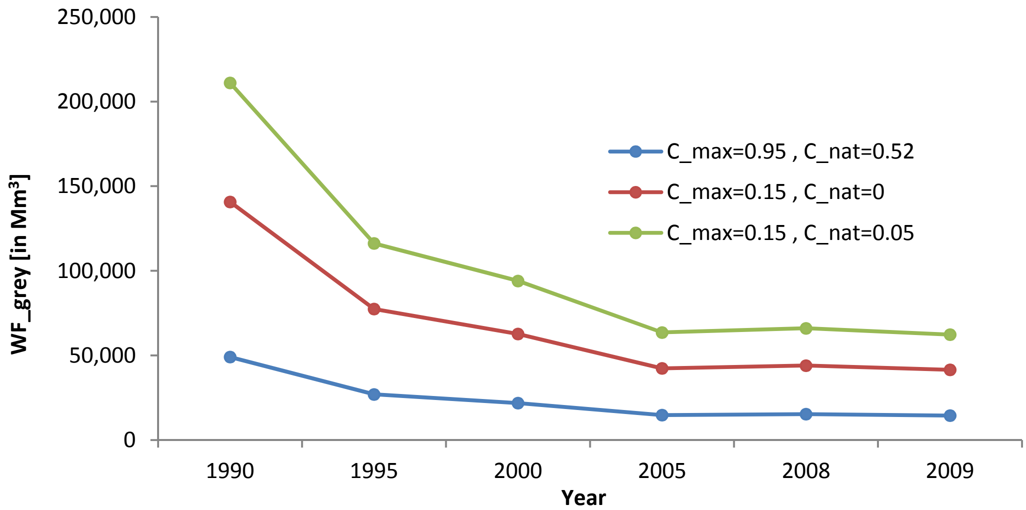

Figure 5.1.4 Grey Water Footprint of total phosphorus emissions within the Netherlands for the period 1990-2009. The WF_{grey} is calculated for 3 different combinations of c_{nat} and c_{max}. (c_{nat} and c_{max} in mg/L)

In Figure 5.1.6 and Figure 5.1.7 the breakdown of the total WF_{grey} per river basin (as given in the Dutch PRTR) and sector respectively is shown. The results are for the WF as calculated for c_{max}=0.15 mg/L and c_{nat}=0.05 mg/L, as they represent the water quality standards the Netherlands (RIZA 2002).

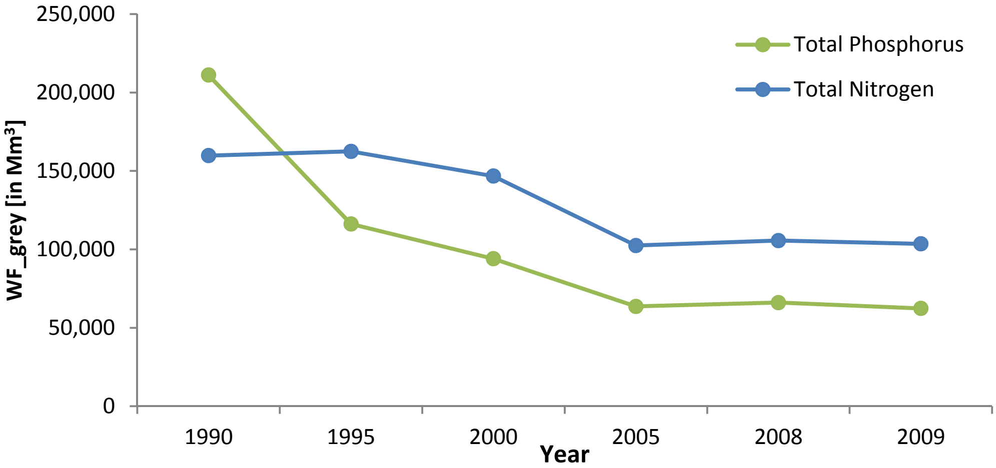

Figure 5.1.5 Comparison of WF_{grey} within the Netherlands for nitrogen and phosphorus emissions (P : c_{max}=0.15 mg/L, c_{nat}=0.05 mg/L, N: c_{max}=2.2 mg/L, c_{nat}=1 mg/L)

The very high WF_{grey} in West Rhine (Figure 5.1.6), compared to the other basins, can be explained combined with Figure 5.1.7. As shown there, the emissions from the chemical industry had the biggest share in WF_{grey} for the year 1990. Studying the emissions occurring per basin, we can see that 96% of the P emissions in water attributed to the chemical industry were actually emitted in the West Rhine. What is more, most of these emissions were reported by just two facilities, both belonging to the manufacturing of fertilizers and nitrogen compounds industry. Both these facilities reduced drastically their P emissions to surface water by 1995 and there are no records for them after 2000, indicating closing down or dislocating outside the Netherlands.

Apart from the chemical industry, the major contributors to P emissions in freshwaters are agriculture and sewage and wastewater treatment (Figure 5.1.7). The WF_{grey} of agriculture is rather stable over the years, oscillating between 30 and 40 billion m^3, while for sewage and wastewater treatment, there is a clear decrease in the WF_{grey}, especially from 1990 to 1995.

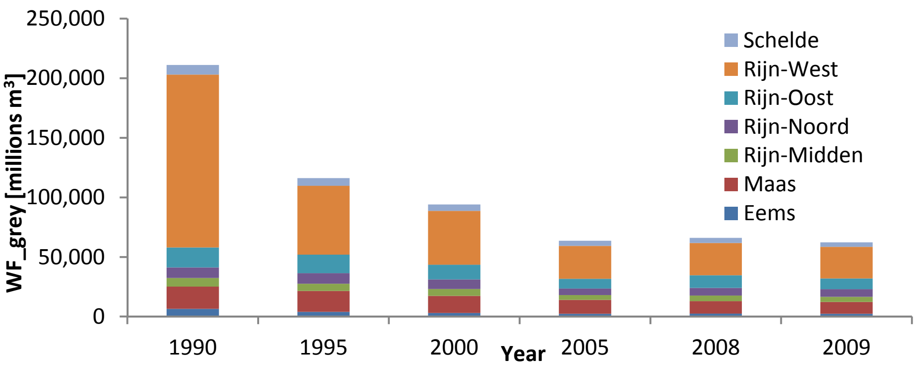

Figure 5.1.6 Grey Water Footprint of Total Phosphorus emissions within the Netherlands per year and river basin (for c_{nat} = 0.05 mg/L and c_{max} = 0.15 mg/L).

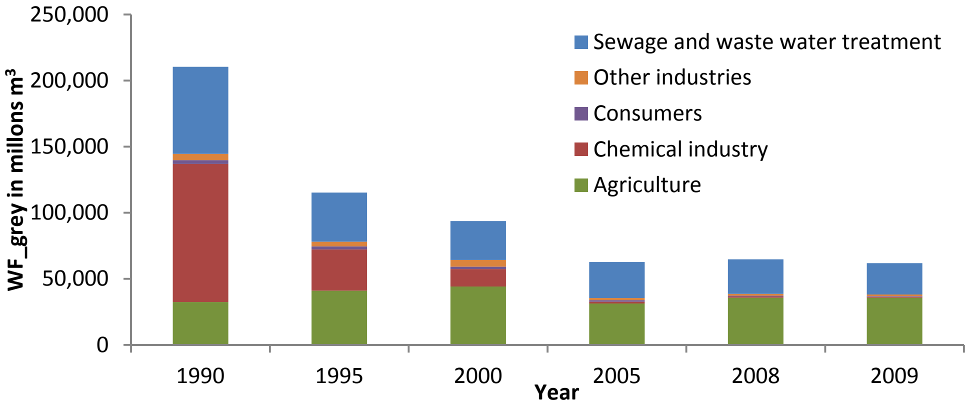

Figure 5.1.7 Contribution to WF_{grey} within the Netherlands per sector, for the years 1990 - 2009. The results are for c_{nat} = 0.05 mg/L and c_{max} = 0.15 mg/L.

5.3.2 Life Cycle Impact Assessment in the Netherlands

The results of the assessment of phosphorus emissions in the Netherlands with the two LCIA methods (Struijs and LC Impact) are presented in Figures 14 and 15. The trends in both the annual impact for the whole of the country and the contributions of the different sectors are similar to each other and to the ones for the WF_{grey} (5.3.1) However, the absolute values for the final impact score differ substantially for the two different methods (Figure 5.1.8).

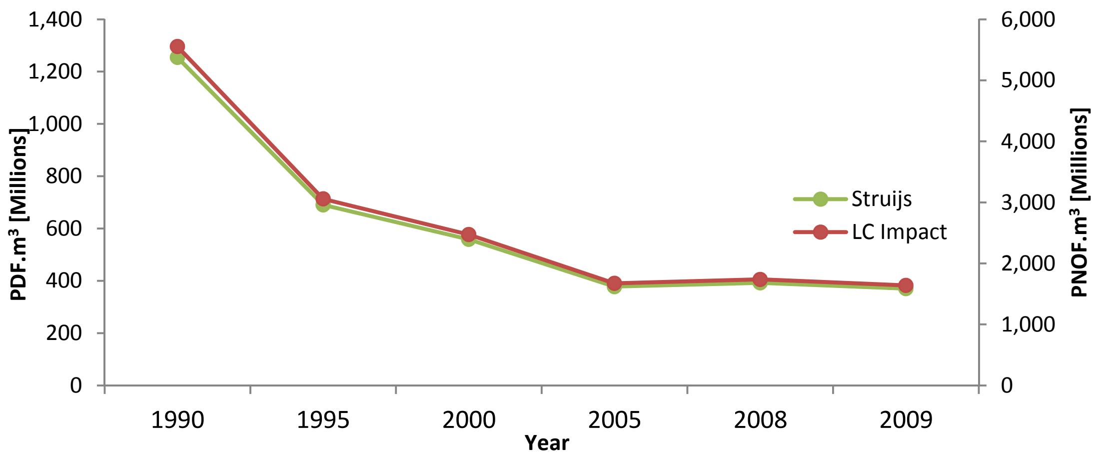

Figure 5.1.8 Assessment of the Eutrophication Impact for all freshwater in the Netherlands (Endpoint Assessment) after Struijs (in PDF·m^3, left y-axis) and LC Impact (in PDF·m^3, right y-axis)

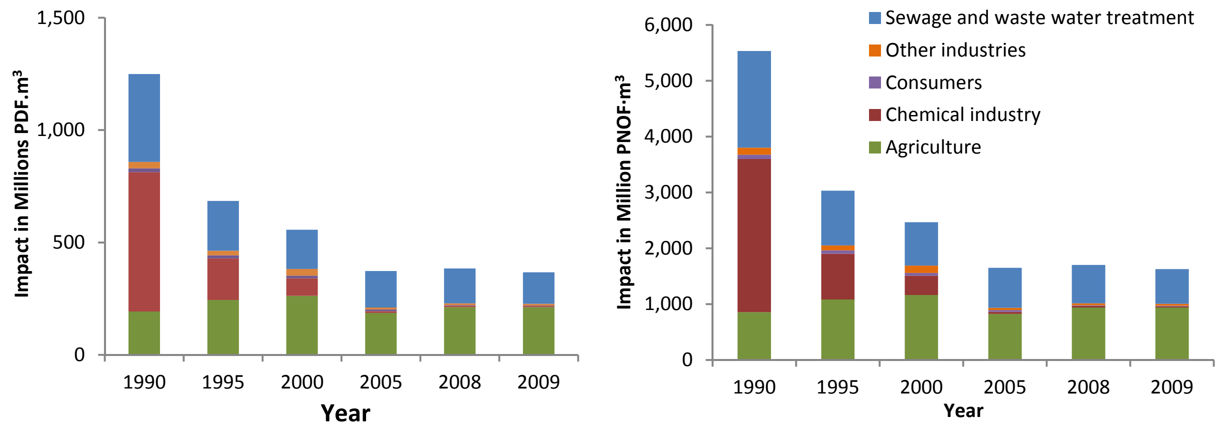

Figure 5.1.9 Assessment of the Eutrophication Impact per sector for all freshwater in the Netherlands (Endpoint Assessment) after Struijs (left) and LC Impact (right)

5.3.3 Nutrient emission assessment in Greece

The assessment of nutrient emissions in Greece was not possible to be as detailed as for the Netherlands. The contributing activities are only three (manure and fertilizer application and discharge of treated and untreated wastewater) and no information of the contribution per basin is available (Figure 5.1.10).The results for the WF_{grey} in Figure 5.1.10 are the ones corresponding to c_{max}=0.15 mg/L, c_{nat}=0.05 mg/L and α=5% (refer to section 5.2.2.1).

Both WF_{grey} and LCIA scores for Greece are smaller than the respective ones in the Netherlands, but in the same order magnitude. What is interesting in the case of Greece, is the small contribution of the wastewater discharge both to the WF_{grey} and the LCIA score, even though the percentage of population connected to wastewater treatment units is smaller than for the Netherlands. This is due to the fact that the majority of the WWTPs discharge directly in coastal waters. The fraction of treated wastewater ending up in freshwater is roughly 21% (Table 5.1.1) and the same was assumed for the untreated sewage. The results of the assessment are sensitive to this assumption, though: if all the untreated sewage ended up in freshwater, the total WF_{grey} for 2009 would rise from around 50 billion m^3 to 65 billion m^3 and the share of wastewater from 12% to 34%.

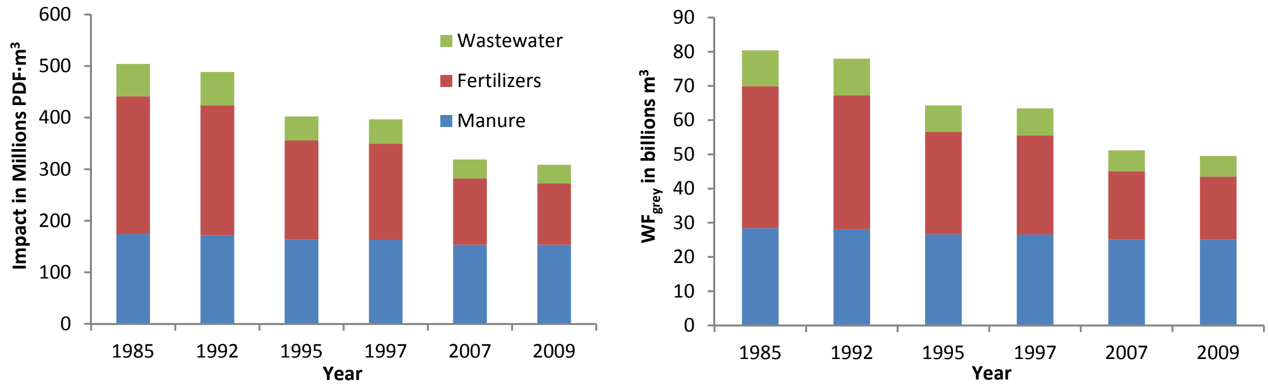

Figure 5.1.10 Eutrophication Impact (after Struijs, left) and WFgrey (right) for total phosphorus emissions in freshwater in Greece per source of P.

5.4 Discussion

5.4.1 Grey water footprint

For the assessment of the total WF_{grey} in the Netherlands and in Greece, the methodology as described in the WF manual (Hoekstra et al. 2011) and relevant publications, e.g. (Liu et al. 2012; Mekonnen and Hoekstra 2011), was followed. Several issues arise during the application of the methodology and the study of the results, which are discussed hereafter.

- **Natural and maximum allowable concentrations**

A significant difficulty in WF_{grey} accounting is the determination of the natural background and maximum allowable concentrations, c_{nat} and c_{max}. In section 5.3.1 (Figure 5.1.4), the significant influence these parameters have in the estimation of the WF_{grey} is illustrated.

The natural background concentration is the concentration of the nutrient in a water body, if no human disturbances existed in the catchment. The natural concentration can be estimated using historical data, reference sites or through modelling (Andersen et al. 2011). There are concerns regarding all these three methods:

- First, historical data (when available) may not be comparable to recent data due to different analytical methods applied. This point holds especially for data before the 1930's (Laane et al. 2005)
- Reference sites are practically non-existent in the Netherlands, as in most of the industrialized world (Smith et al. 2003)
- Given the possible unreliability of historical data and the lack of reference sites, the question arises what input to provide to a model for the estimation of c_{nat}.

The maximum allowable concentration cannot be defined in a straightforward way either. C_{max} is based on quality objectives usually on the country level, while ideally it should be catchment specific (Hoekstra et al. 2011). In addition, such quality objectives are set by policy makers, reflecting views and priorities of particular policy groups and policy organisations in addition to scientific observations. Laane et al. also indicate the need for differentiation of the target and background concentrations between different water bodies and geological areas.

Therefore, the systematic collection of such data and storage in a database would constitute a substantial improvement of WF_{grey} accounting, facilitating the application of the method and reducing the uncertainty of the results.

- **Emissions from diffuse sources**

Pollutant emissions from diffuse sources can be considered as a weak point of the WF_{grey} assessment. In the WF manual it is proposed that a factor alpha (α) is applied to estimate the amount of pollutants reaching freshwater from the total amount applied in soil. This method is only proposed as the "default" method when not enough time or resources are available for modelling. In (Chapagain et al. 2006), as well as in most of later similar studies, this factor is assumed to be 10% for leaching of nitrogen from agricultural soils. In Liu et al. 2012 the emissions are modelled using the Global *NEWS* model and calculated in detail.

The determination of nutrient emissions from diffuse sources does not have implications in the estimation of the WF_{grey} in the Netherlands, since all the pathways are taken into account and the emissions given are all directly to surface water. For Greece, on the other hand, a simplified assumption for the value of α had to be made for the calculation of the WF_{grey}.

In order to get an idea of the ratio of applied nutrients on agricultural soil reaching surface water, data from Netherlands Statistics (CBS 2012) and the Dutch PRTR were compared (Table 5.1.5). The data from CBS are gross phosphorus emissions to agricultural soil through manure and fertilizer, while data from PRTR are the net emissions to water from agriculture. The ratio of the net phosphorus emissions to water to total gross emissions to soil is given in the last column of Table 5.1.5, ranging from almost 7% to approximately 11%. Despite the fact that simply assuming the fraction of nitrogen leaching from agricultural fields to freshwater to be 10% seems rather an oversimplified approach, it could actually be an assumption good enough for the Netherlands.

Table 5.1.5 Comparison of gross phosphorus emissions to soil and net emissions to surface waters for the Netherlands and estimation of the respective α factor

Year	Manure supply to soil	Fertilizer supply to soil	TN emissions in surface water from Agriculture	Ratio of applied manure and fertilizer reaching surface water
	10^6 kg	10^6 kg	10^6 kg	%
1990	406	400	54.62	6.8%
1995	495	395	83.15	9.3%
2000	409	329	83.03	11.3%
2005	360	268	43.14	6.9%
2008	352	230	54.2	9.3%
2009	338	218	54.2	9.7%

- **Comparison of results with other studies**

Several studies have been recently published on WF accounting, where usually emphasis is given on blue and green water footprints. Grey water footprints are mostly estimated for nitrogen leaching from crops using a set of assumptions for α (10% or 5%), c_{max} (10 $mgNO_3$-N/L) and c_{nat}. (0 mg/L) (Chapagain et al. 2006; Mekonnen and Hoekstra 2010; Chapagain and Hoekstra 2011; Gerbens-Leenes and Hoekstra 2012; Mekonnen and Hoekstra 2011).

In a recent study, (Liu et al. 2012) are estimating the water pollution levels (WPL) of both nitrogen and phosphorus on a global scale, using the Global *NEWS* model to estimate the different parameters needed for the assessment (nutrient loads, c_{max} and c_{nat}, Q_{act}).

The results from (Liu et al. 2012) for the Netherlands have been compared with the respective ones from this study. The WPL in this study is smaller, though in the same order of magnitude, than the one in (Liu et al. 2012). This difference could be explained by the fact that Liu et al. did their assessment on a catchment area level. In this study the assessment was performed for all emissions within the Netherlands and the resulting WF_{grey} per basin (Rhine, Schelt or Meuse) was divided by the actual discharge, Q_{act} of each basin. It is thus obvious that in our case the WPL is underestimated, since emissions occurring in a part of a river basin are compared with the discharge of the whole catchment.

Table 5.1.6 Comparison of the values for WF_{grey} and WPL with ones from similar studies. The two different sets of values for the year 2009 correspond to (in mg/L):c_{max} = 0.95 and c_{nat} = 0.52 (1) and c_{max} = 0.15 and c_{nat} = 0.05 (2)

Year	THIS STUDY		LIU ET AL. 2012
	$WF_{grey}[Mm^3]$	WPL	WPL
1995 - 2005	21’227	0.87	
2000	21’866	0.88	1.57
2009 (1)	14’490	0.60	
2009 (2)	62’309	2.58	

The results of a WF_{grey} assessment are expressed in volume of water, a concept easily conceivable by everyone. However, a volume alone does not provide much useful information, unless a yardstick is provided for comparison. WPL concept serves this purpose, comparing the WF with the yearly available water within a basin. In the case the WF of countries is being studied, though, it was shown that this can be misleading, especially when these countries are parts of international river basins. Alternatively, the WF_{grey} could be compared with the annual water demand or consumption of the specific country or the annual freshwater availability. This comparison would not be an environmental sustainability assessment in any case, as the WPL is claimed to be. Rather, it would serve as an approach for better communication of the results. A risk of grey water volumes is that it gets compared to blue water or green water. While green and blue water are real water volumes without impact assessment, grey water is a theoretical volume based on impact assessment. Therefore it cannot be compared to blue water for instance. A water volume polluted to the legal threshold can still serve many purposes and eventually be cleaned by natural processes (see below for LCIA methods). A further limitation is the focus on nitrogen and phosphorous: for agriculture, this might be useful but for industrial production other emissions are much more relevant, such as heavy metals or other toxic effects.

5.4.2 LCIA

For the assessment of nutrient emissions with LCIA, the two newly developed methods by Struijs et al. and LC Impact were used. These methods include both fate and effect factors for phosphorus emissions in freshwater, based on the same framework for the development of the models; however, the deviations in the results of the application of each method are significant (Figure 5.1.8). Table 5.1.7 summarizes some key differences in the fate and effect factors between the two methods, which could explain the deviations in the respective results.

First of all, the aggregated fate factor of LC Impact for Europe is almost 3 times smaller than the one by Struijs, explained mainly by the fact that in LC Impact two additional mechanisms of P removal are included in addition to transport, namely retention and water use (Helmes et al. 2012). Also, for the derivation of a PDF vs C_P relationship, Struijs et al. sum all the genera occurring in one C_P interval. Azevedo et al., on the other hand, generated C_P ranges of occurrence for each species and species were considered to be present in C_P classes within their range of occurrence and absent outside it.

Table 5.1.7 Key differences between the fate and effect factors of Struijs et al. 2011a (Stuijs) and Helmes et al. 2012 and Azevedo et al. In prep. (LC Impact).

		Struijs	LC Impact
Fate	Model	CARMEN (Europe west of Uralia)	Newly developed global model
	Spatial Resolution	1/6°x1/6°	0.5°x0.5°
	Aggregation level	River basin and Europe	Continent or country
	Processes included in the fate	Transport	Transport, retention and water use
	Sources of nutrients	Point sources, manure and fertilizer	Point sources only
Effect	Unit	PDF·L^3·M^{-1}	PNOF·L^3·M^{-1}
	Reference area	Dutch inland waters	Global study
	Spatial reference	Generic EF for Europe	Aggregated EF for NL
	Approach used to derive the EF	Marginal	Marginal, Average, Linear
	Type of freshwater	All freshwater	Lakes and streams separately
	Type of organism	Macro-invertebrates at genus level	Autotrophs and heterotrophs at species level
	Stressor – effect relationship	$PDF = \frac{1}{1 + 4.07 \cdot C_P{}^{-1.11}}$	$PNOF = \frac{1}{1 + \exp(-\frac{logC_i - \alpha}{\beta})}$

The assessment of nutrient emissions in the Netherlands with the LCIA method, results in approximately 400·10^6 PDF·m^3 for the year 2009 (after Struijs) and 1'600 PNOF·m^3 for the same year (after LC Impact), and provides an estimation of the ecological damage caused by phosphorus emissions in the country's freshwaters. However LCIA results are meant to be part of a greater LCA and are only put into context when compared with scores of other impact categories, which was out of the scope of this study, leaving the results, in a sense, incomplete.

5.4.3 Comparison of WF_{grey} and LCIA

Making a synthesis of points discussed above and some further remarks concerning the methods, comparative advantages and disadvantages of WF_{grey} and LCIA as methods to assess nutrient emissions, are discussed in the following:

- **Comparability of results**

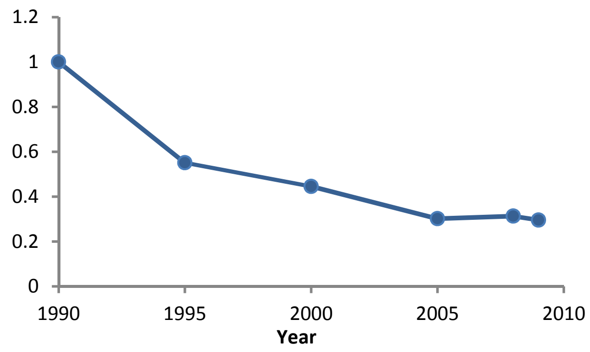

Figure 5.1.11 Relative emissions of total phosphorus to surface waters per year (emissions in 1990 = 1)

The absolute results of the methods are not directly comparable, since they are given in different units and they represent very different concepts. WF_{grey} results are in m^3, corresponding to a volume of water needed to dilute the respective pollutant load, without modelling the entire cause effect chain and specific vulnerabilities. LCIA results, on the other hand, are in PDF (or PNOF), describing the potential ecological damage the pollutant emissions can cause to freshwater ecosystems.

What is comparable, though, is the trend of the results in time and by sector. In fact, these trends are identical, following the respective trends of the emissions (Figure 5.1.11). This should be expected, since for both methods the procedure for assessment is basically the multiplication of the relevant emissions with a factor, either this is called dilution (for WF_{grey}) or characterization factor (for LCIA).

- **Ease of application**

Despite the initial expectations, it was proved that once the inventory data are available, LCIA method is somewhat simpler to apply than the WF_{grey}, at least when the total annual emissions within a country are assessed. This is due to the fact that WF_{grey} requires the estimation of c_{nat} and c_{max} that is not always an easy task (section 5.4.1). Apart from these two concentrations, the demand on data is the same for the methods, namely the emissions of nutrients to freshwater. For this reason there was no substantial difference when applying the methods in the Netherlands and in Greece.

This holds for the case that the "default" method for handling diffuse sources in the WF_{grey} assessment is used. If a more detailed assessment is required, then model approaches should be employed, increasing the time, data demand and complexity of applying WF_{grey}, while in LCIA this modelling is already included in the characterization models. However, this is likely to change soon, once more studies on WF_{grey} are being published and the methodological steps are refined.

- **Ease of communicating the results**

Obviously, a volume of water is something directly and intuitively perceivable from a non-expert audience, while PDFs, PAFs or PNOFs are complicated and abstract concepts, mainly serving communication within experts. However, the volume unit is also misleading, since it does not relate to time and suggests direct comparison with water consumption volume. While the WF_{grey} has much higher potential of reaching the public audience or the decision-makers and raising environmental awareness or influencing policy making respectively, it is problematic due to its impact unit. In order for a volume of water as the WF_{grey} to acquire an actual meaning, it has to be compared with a yardstick, such as the Q_{act} (section 5.4.1). In addition, it should be kept in mind that in a usual LCA, the results for the eutrophication assessment would not be presented independently, but together with other impact categories.

Therefore, following the suggestion of Ridoutt and Pfister (2012) allows calculating LCIA impacts in equivalence terms of a volume of water consumed. Since environmental relevance of water consumption depends on the location, the equivalence is also location specific. The ecosystem impact of water consumption based on Pfister et al. 2009 can be translated into units of PDF m3 yr (assuming an average water depth of 6.25m). Consequently, the impacts for Greece amounts to 1.12 PDF m3 yr and consequently emission of one kg P is equivalent to ~50 m3 water consumption in Greece. This procedure is also in line with the draft ISO standard on water footprint (ISO, 2013), where impacts on water resources due to pollution and consumption should be accounted for a full profile. Using the endpoint units available in LCIA methods allows aggregating the different impact in to one number, and allows communicating it as a water volume equivalent.

- **Final remarks**

Keeping in mind all the above discussed points, it has to be stressed out that the two methods are not meant to substitute each other. As it is stated in the Water Footprint manual, WF belongs to the family of footprints, which "show the pressure of humans to the environment, not the impacts" (Hoekstra et al. 2011), while LCIA exactly aims at describing impacts as far in the cause-effect chain as possible. From this point of view, the use of LCIA or WF_{grey} should be decided based on the scope on the study and on whether a water resources management or a complete environmental assessment perspective is adopted.

In the case of different pollutants emitted in water bodies, WF_{grey} is determined by the most critical one. This is a reasonable approach, however, it overlooks the possible interactions between the pollutants and the possible cumulative or buffering effects that may occur. (Hoekstra et al. 2011) argues that these effects are taken into account indirectly, by using the maximum allowable concentration, which include such interactions. In LCIA different pollutants are considered by assigning them to the respective impact categories, although overlaps between them may be observed.

The WF_{grey} was developed as a measure of pollution expressed in volume of water polluted, so that it can be compared with the blue and green water footprints of the same product, process, nation etc. As such, alone it is by definition an insufficient way of assessing pollution and it gets into context when compared with the blue and green water footprints, similarly to LCIA results getting into context when different impact categories are compared with each other. Additionally, although the grey water unit is a water volume, it needs to be highlighted that it is not directly comparable to blue or green water.

For CREEA, the limitation is mainly the emission coverage which is very limited (N,P and BOD).

5.5 Conclusions

This study focused on the comparison of WF_{grey} and LCIA as methods for the environmental assessment of nutrient emissions in freshwater in the Netherlands. The WF_{grey} within the Netherlands, as well as the impact in ecosystems due to phosphorus emissions were estimated and comparative strengths and weaknesses of the methods were discussed. The comparative assessment indicated that the two methods are rather complementary than competitive and the most appropriate one should be chosen based on the scope of the study, the intended area of application, the intended audience etc. This work aims to provide a good reference when it has to be decided which of the methods should be used for the environmental assessment of nutrient emissions.

6. Discussion

6.1 To what extent can results of thermal pollution and grey water footprint be made comparable?

As described above, thermal pollution and emissions impacts can be compared using the endpoint metrics in LCIA, which is typically PDF m3 yr or a variation of it. By applying endpoint assessment to thermal emissions, N and P emissions as well as blue water consumption, we can aggregate and compare these impacts affecting water resources. The green water consumption as such is affecting land use and therefore is assessed by land occupation characterization factors that can also be aggregated with water consumption impacts.

However, comparability is always limited since different emissions have different effects on the ecosystem and even if the units of the results are the same, the ecological meaning is not necessarily comparable. However, all impacts are characterized by a fate and an effect function, which guarantees at least some consistency. Further research along this problem is definitely required.

Without looking into site-specific aspect, the following factors are retrieved:

	1kg N emission to water	1kg P emission to water	1 MJ heat emission to water	1 m3 water consumption (Global average)
Impacts in PDF m^3 yr	7.7	56	3 E-05	4.4
Impacts m^3-eq. of global water consumption	1.75	12.7	6.8E-06	1

Spatial variation needs to be accounted for properly addressing the impacts as discussed above.

6.2 How water pollution accounts can be made comparable to water consumption accounts using the water footprint concept?

The water footprint is an indicator of human appropriation of freshwater resources. It measures both the direct and indirect ‘water use’ of consumers and producers. The term ‘water use’ represent both the consumptive water footprint (green and blue water footprint) and the water required to assimilate the pollution (grey water footprint). The grey water footprint refers to the volume of water that is required to assimilate waste, quantified as the volume of water needed to dilute pollutants to such an extent that the quality of the ambient water remains above agreed water quality standards. As stressed in the 2006-UN Human Development report, water quantity is not the only measure of water scarcity, but quality also plays an important role in the availability of water for human use (UNDP, 2006). Pollution of freshwater resources not only poses a threat to environmental sustainability and public health but also increases the competition for freshwater (Pimentel et al., 1997; Pimentel et al., 2004; UNDP, 2006; UNEP GEMS/Water Programme, 2008). Vörösmarty et al. (2010) have shown that water pollution together with other factors pose a threat to global water security and river biodiversity. Expressing water pollution in terms of a water volume needed to dilute pollutants has been recognized earlier by for example Postel et al. (1996).

By expressing the water pollution level in terms of a water volume needed to assimilate the pollution (grey water footprint), it can be made comparable with the green and blue water footprint. While the green and blue water footprints are consumptive, the grey water footprint represents the volume of water required to assimilate pollution. Expressing water pollution in the same term as water consumption, one is able to compare the use of runoff as a source (blue water footprint) to the use of runoff as a sink (grey water footprint). However, it should be noted that the pressure exerted by the grey WF on the freshwater resources is quite different to blue WF. As a result, a one-to-one comparison of the grey WF with the consumptive water footprint is difficult. By estimating all components separately, one is able to measure the total pressure (consumptive and pollution) on the freshwater resources.

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Appendix 2.2: Summary of water quality standards

Parameters	Units	China			Japan			Malaysia			Australia				EU (and Germany)			Netherlands		United Kingdom		Canada		USA		
		Ecosystem	Recreational	Aquaculture	Ecosystem	Recreational	Aquaculture	Ecosystem	Recreational	Aquaculture	Ecosystem	Recreation	Aquaculture	Consumption	Ecosystem AA-EQS	Ecosystem MAC-EQS	Recreational	Ecosystem Guide	Ecosystem Mandatory	Ecosystem	Aquaculture	Ecosystem	Recreational	Ecosystem CMC	Ecosystem CCC	Consumption
Inorganic N	µg/L	200 – 500	150 – 1,000 [a]			10,000 [j]			400 [a], 7000	400 [a]					ba			25,000 [ao]	50,000 [ao]	140 – 420 [at]		16,000 [bc]				10,000 [bc]
Total Nitrogen	µg/L				200 [k]	300 [k]	600 [k]				100 – 230 [t]				ba			1,000		980 – 3,794 [au]						
Total Phosphorus	µg/L	15 – 45 [b]	20 – 50	1 [c]	20 [k]	30 [k]	50 [k]		200	100	10 – 40 [t]				ba			400 [ap]							0.1	
Suspended solids	mg/L	10 – 150 [d]		10 [d]				25 [o]	50 [o]	150 [o]		x	10		ba			25				bb		bb	bb	bb
Dissolved oxygen	mg/L	3 – 6	3 – 5	3 – 5	≥ 7.52	≥ 7.5	≥ 5		5 – 7 o	3 – 5 [o]	>90%	>80%	>5 [ac]		ba			> 70%		2.4 – 5.7 [av]		>8				
Turbidity	NTU								50 [o]		0.5 – 20	x			ba							bb	50	bb	bb	bb
Colour	mg/L Pt-Co	nil	< 25	nil		Transparent [l]			150 [o, p]			x	30 – 40					10	20				bd	bb	bb	bb
Light penetration	Secchi (m)		0.5 – 1.2			0.5 – 1						x			ba								1.2			
pH		6.8 – 8.8	6.5 – 8.5	7 – 8.5	7.8 0 – 8.3	7.8 – 8.3	7.8 – 8.3		6 – 9 [o]	5 – 9 [o]	8.0 – 8.5	6.5 - 8.5	6 – 9					6.5 – 8.5				7.0 – 8.7	6.5 – 8.5		6.5 – 8.5	5.0 – 9.0
Salinity													33,000 –37,000 [ad]		ba							<10% [bd]				250,000
Temperature	oC	< 1 – 4 [e]	< 2 – 4 [e]							± 2 [o]		16 - 34	± 2 [z]		ba			22	25	± 2 [aw]	± 2 [aw]	± 1	≤ 30			
Arsenic (total)	µg/L	20 – 50		50		10		100	50	400			30					10	50	25 [ax]		12.5 [be]		69	36	0.018
Cadmium	µg/L	1 – 10		5		10		100	10	10	0.7		0.5 – 5		0.2	0.45 – 1.5 [ag]		1	5			0.12		40	8.8	
Chromium (total)	µg/L	50 – 500		100				500					20						50							
Cr III	µg/L									2500	27											56 [be]				
Cr VI	µg/L	5 – 50				50			50	1400	4.4									0.6 [ax]		1.5		1,100	50	
Copper	µg/L	5 – 50	10 – 100	10				100	1000		1.3		5	1,000				20	50	5 [ax]				4.8	3.1	1,300
Lead	µg/L	1 – 50		50		10		100	50	20	4.4		1 – 7		7.2				50					210	8.1	
Mercury (total)	µg/L	0.05 – 0.5		0.5		0.5		1	1	4	0.1 [ae]		1		0.05 [ah]	0.07 [ah]		0.5	1			0.016 [be]		1.8 [bg]	0.94 [bg]	
Mercury organic	µg/L					0 [m]																0.004 bf				0.3 bj
Nickel	µg/L	5 – 50	50 – 100	50		10			50	900	7		100		20									74	8.2	610
Silver	µg/L								50	0.2	1.4		3											1.9		
Zinc	µg/L	20 – 500	100 – 1000	100					5,000	400	15		5	5000				500	3,000	40 [ax]				90	81	7,400
Phenol	µg/L	5 – 50	5 – 100	5					10 [o]		400			1,000 - 10,000						7.7 [ax], 46 [ay]						21,000
Phenolics	µg/L			10 [f]							various			various	0.01 – 0.4 [ai]	1-2 [ai]			1 [aq]	20 [ax, az]		0.7 [br]		7.0 - 13 [bl]	1.7 – 7.9 [bl]	0.27 – 1,800 [bn]
PAHs (total)	µg/L										ID				aj	aj			0.2			ID				670 – 8,300 [bo]
PAHs as specified	µg/L	0.0025 [g]									aa			af	0.002 – 0.05 [ak]	0.1 [al]						1.4 [be,bh]				0.0038 [bp]
Tributyltin (TBT)	µg/L												0.01		0.0002	0.0015						0.001 [be]		0.42	0.0074	
TBT as Sn	µg/L										0.006															
PCBs	µg/L					0 [m]			0.1 [o]	6 [o]	ID		2												0.03 [bi]	0.000064 [bi]
DDT	µg/L	0.05 – 0.1		1					0.1 [o]	1 [o]	ID				0.01, 0.025 [al]									0.13	0.001	0.00022
Hexachlorobenzene (HCB)	µg/L								2 [o]	9 [o]	ID				0.01	0.05						ID				0.00028
Ammonia – total as N	µg/L		500 [h]						300 [o]	900 [o]	910 [v]		1,000											bl,bm	bl,bm	
Ammonia – unionised as NH3	µg/L			50									100					50								
Ammonia – unionised as N	µg/L	20	20 – 200	20																21 [ax]						
Cyanide	µg/L	5 – 200		5		10			20	60	4 [w]		5 [w]						50	1 [ax], 5 [ay]				1	1	140
Sulphide (total)	µg/L	20 – 250		200					50 [q]		ID		2 [w]												2	
Surfactants	µg/L	10 – 30	200								ID	x						200								
Oil & grease	µg/L				nil [m]	nil [m]	nil [m]	nil [m]	40 – 7,000 [r]	nil [m]		x											nil	bb	bb	bb
Total Petroleum Hydrocarbons	µg/L	50 – 500		50			10 –100					x							50							
Chlorine — total residual	µg/L								200,000		ID		3					200,000 [ar]		10 [ay]		0.5 [bk]		13	7.5	
Escherichia coli	cfu/100 mL	1,000	1,000	70, (50 – 500 [i])	1,000 [i]	400 – 1,000 [i]	70 [i, n]	100 [i]	5,000 [i, o]	50,000 [i, o]							an	50 [as]					200			
Enterococci	cfu/100 mL											y					an						35			35 [bq]
Faecal streptococci	cfu/100 mL																	20								
Faecal coliforms	cfu/100 mL	200	200	14					400 [o, s]	5,000 [o]				14 [ab]				20					200			

Appendix 2.2 (contd.)

Note

a: NO_2 only	aj: not applicable; individual compounds have an specific EQS
b: reactive phosphorous	ak: range for individual compounds; naphthalene 1.2 µg/L.
c: yellow phosphorous	al: benzo(a)pyrene
d: above level in natural ambient	am: p-p-DDT and total respectively
e: range of variation above monthly average of last 10 years	an: assessment and classification based on levels of Enterococci and *E coli*, supplemented by development and regular review of the bathing water profile
f: volatile phenols	ao: nitrates
g: benzo[a]pyrene	ap: phosphates
h: for temperatures above 20oC and pH>8	aq: phenol index
i: total coliforms	ar: chlorides
j: NO2 and NO3	as: total coliforms, assuming >90% are E. coli.
k: annual averages, applicable only where marine phytoplankton blooms may occur	at: 15 to 30 µM from offshore to transitional waters respectively
l: bathing beaches	au: 70 to 271 µM from clear to very turbid estuaries
m: not detectable	av: mg/L in moderate to high water conditions
n: for the fishery class I to cultivate oyster to be eaten raw	aw: interim, applicable to marine Special Protection Areas
o: proposed standard	ax: long-term (AA-EQS)
p: TCU units	ay: short-term (MAC-EQS)
q: sulphur	az: 2,4-dichlorophenol
r: Class II waters, from nil to 40 µg/L mineral or nil to7,000 µg/L emulsified edible	ba: Member states to assess ecological status with respect to nutrient conditions, transparency, oxygenation conditions, salinity & thermal conditions based on findings at type-specific reference sites
s: maximum 20,000 cfu	ID = insufficient data
t: Actual value depends on broad region of Australia; excludes "South central Australia" where TN = 1000, TP = 100	bb: narrative
v: At pH 8.0	bc: NO3- only, equivalent to 3,600 µg N L-1
w: Un-ionised	bd: not significantly increased above natural background
x: Aesthetically acceptable	be: interim guideline
y: Risk based assessment framework developed around results of sanitary inspection and 95th percentile levels of enterococci	bf: freshwater guideline
z: Over 1 hour	bg: total mercury
aa: Naphthalene, 50 ug/L; others, insufficient data - low reliability figures only.	bh: naphthalene
ab: Median should be below this value in units of MPN/100mL with no more than 10% of samples exceeding 43 MPN/100mL; or in Western Australia median can be below this value in units of CFU/100mL with no more than 10% of samples exceeding 21 CFU/100mL.	bi: total PCBs congeners
ac: mg/L	bj: µg/g wet weight in fish/shellfish
ad:: Total dissolved solids (TDS)	bk: hypochlorous acid and monochloramine
ae: Inorganic mercury	bl: pH dependent
af: Naphthalene, 1000 ug/L; acenaphthene, 20 ug/L	bm: range depending on temperature for 20 g/kg salinity
ag: range for five different water classes	bn: pentachlorophenol, chlorophenols, dinitrophenols and nonylphenols
ah: total Hg and its compounds	bo: anthracene, pyrene, acenaphthene
ai: octylphenol, nonylphenol and pentachlorophenol	bp: benzo(a)pyrene, benzo(a)anthracene, benzo(a)fluoranthene, indeno(1,2,3-cd)pyrene
	bq: Recreational guideline; maximum values should not exceed the range 104-500 cfu/100 mL depending on frequency of use
	br: nonylphenol TEQ

Appendix 5.1.1: Phosphorus Emissions (kg / yr) in the surface waters in the Netherlands per activity and river basin

Per river basin

AREA	1990	1995	2000	2005	2008	2009
Eems	644'827	391'748	298'118	238'240	252'129	237'211
Maas	1'869'720	1'754'420	1'438'840	1'155'270	1'039'110	986'235
Rijn-Midden	721'531	609'226	571'582	406'914	470'158	436'386
Rijn-Noord	879'483	881'288	811'749	555'484	638'684	634'687
Rijn-Oost	1'687'820	1'557'680	1'234'990	815'539	1'068'150	900'443
Rijn-West	14'513'200	5'786'710	4'509'040	2'764'380	2'704'530	2'658'730
Schelde	795'423	642'786	538'119	421'336	430'289	376'473
NOORDZEE	0.1211	0.1292	0.1446	16.7232	15.8771	15.3156
TOTAL	**21'112'004**	**11'623'858**	**9'402'438**	**6'357'926**	**6'603'777**	**6'230'866**

Per sector

Sector/Activity	1990	1995	2000	2005	2008	2009
Agriculture	3'249'000	4'110'000	4'416'000	3'136'000	3'571'000	3'571'000
Chemical industry	10'440'000	3'119'000	1'325'000	136'800	103'000	75'760
Construction		84'080	124	1'208		
Consumers	289'300	228'600	183'900	124'500	53'030	44'050
Energy production	570	43	397	872	94'010	6'762
Other industries	479'000	346'000	505'500	152'200	131'700	125'500
Refineries	60'610	2'200	8'636	50'700	27'070	25'700
Sewage and waste water treatment	6'581'000	3'720'000	2'945'000	2'728'000	2'615'000	2'370'000
Trade and services	4'996	3'170	3'248	21'310	312	632
Transport	9'722	8'673	7'438	7'056	7'914	8'686
Waste disposal	972	2'224	7'552	234	1'052	3'567
TOTAL	**21'115'170**	**11'623'990**	**9'402'795**	**6'358'880**	**6'604'088**	**6'231'657**

Appendix 5.1.2: Manure and fertilizer application in Greece and calculation of P from point sources

Year	Manure [mln kg]	Fertilizer [mln kg]	(1)	(2)	(3)	(4)	(5)	(6)	(7)
1985	56.81	83.07	9.92	10%	108'619	22'775	1'024'867	1'047'642	1.05
1986	61.39	79.88	9.95						
1987	59.49	72.12	9.99						
1988	58.89	76.47	10.02						
1989	58.04	77.66	10.06						
1990	57.48	84.39	10.12						
1991	55.82	77.53	10.19						
1992	56.11	78.32	10.32	11%	128'820	27'011	1'049'627	1'076'637	1.08
1993	56.03	55.44	10.42						
1994	51.92	55.44	10.51						
1995	53.21	59.84	10.60	45%	522'072	109'467	668'963	778'429	0.78
1996	51.72	63.80	10.67						
1997	53.07	58.08	10.74	45%	529'443	111'012	678'407	789'419	0.79
1998	49.91	52.80	10.81						
1999	49.99	52.36	10.86						
2000	50.58	49.72	10.90						
2001	50.56	49.72	10.93						
2002	50.40	47.08	10.97						
2003	51.83	46.20	11.01						
2004	51.47	44.00	11.04						
2005	*51*	*42.93*	11.08						
2006	*50*	*41.46*	11.13						
2007	*50*	*39.98*	11.17	65%	795'149	166'724	448'874	615'598	0.62
2008	*50*	*38.50*	11.21						
2009	*50*	*37.03*	11.26	67%	823'653	172'701	429'168	601'869	0.60

Data in Italic for manure and fertilizer (2005 – 2009) are extrapolated
(1) Population
(2) % connected to ww treatment (OECD data)
(3) P in effluent of all WWTPs [kg/yr]
(4) P in fw from WWTPs [kg/yr]
(5) P from not connected pop. [kg/yr]
(6) Total P in fw from sewage and WWTPs [kg/yr]
(7) Total P in fw from sewage and WWTPs [mln kg/yr]
Assumptions:
20% removal efficiency for P in normal activated sludge system
Specific P production for Greece : 1.5 g/cap.d

Appendix 5.1.3: Full list of Characterization Factors for assessment of phosphorus emissions in freshwater with LCIA

Study	Factor	Reference Area	Nutrient Source	Unit	Value
Struijs et al.2011a	Fate Factor	Europe	Point	d	111
	Effect Factor (M)[1]	Europe	-	$DF{\cdot}m^3{\cdot}kg^{-1}$	203
	Characterization Factor	Europe	Point	$DF{\cdot}m^3{\cdot}d{\cdot}kg^{-1}$	21’685
	Characterization Factor	Europe	Manure	$DF{\cdot}m^3{\cdot}d{\cdot}kg^{-1}$	1’119
	Characterization Factor	Europe	Fertilizer	$DF{\cdot}m^3{\cdot}d{\cdot}kg^{-1}$	1’174
LC Impact	Fate Factor	Europe	Point	d	40.4
	Fate Factor	Netherlands	Point	d	33.7
	Characterization Factor (M, hetestreams)[2]	Netherlands	Point	$PNOF{\cdot}m^3{\cdot}d{\cdot}kg^{-1}$	95’947
	Characterization Factor (M, hetelake)[3]	Netherlands	Point	$PNOF{\cdot}m^3{\cdot}d{\cdot}kg^{-1}$	12’313
	Characterization Factor (A, hetestreams)[4]	Netherlands	Point	$PNOF{\cdot}m^3{\cdot}d{\cdot}kg^{-1}$	134’006
	Characterization Factor (A, hetelake)[5]	Netherlands	Point	$PNOF{\cdot}m^3{\cdot}d{\cdot}kg^{-1}$	124’272

[1] *(M) stands for Marginal*
[2] *Marginal, for heterotrophs in streams*
[3] *Marginal, for heterotrophs in lakes*
[2] *Average, for heterotrophs in streams*
[3] *Average, for heterotrophs in lakes*