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DECEMBER 2013

GREY WATER FOOTPRINT ACCOUNTING

TIER 1 SUPPORTING GUIDELINES

VALUE OF WATER

RESEARCH REPORT SERIES No. 65

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Published by:

UNESCO-IHE Institute for Water Education

P.O. Box 3015

2601 DA Delft

The Netherlands

The Value of Water Research Report Series is published by UNESCO-IHE Institute for Water Education, in collaboration with University of Twente, Enschede, and Delft University of Technology, Delft.

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Please cite this publication as follows:

Franke, N.A., Boyacioglu, H. and Hoekstra, A.Y. (2013) Grey water footprint accounting: Tier 1 supporting guidelines, Value of Water Research Report Series No. 65, UNESCO-IHE, Delft, the Netherlands.

Acknowledgement

We would like to thank the Grey Water Footprint Expert Panel, for their input and feedback in the process of developing these supporting guidelines: Colin Brown (University of York – UK), Richard Coupe (U.S. Geological Survey, Pearl, Mississippi), Julian Dawson* (The James Hutton Institute, Craigiebuckler, Scotland UK), Mark Huijbregts (Radboud University Nijmegen, The Netherlands), Himanshu Joshi (Indian Institute of Technology at Roorkee, India), Bernd Lennartz (Faculty for Agricultural and Environmental Sciences Rostock University, Germany), Roger Moussa (French National Institute of Agricultural Research, France), Alain Renard (Sustainable Business Development, C&A, Brussels), Ranvir Singh (Massey University, New Zealand), Merete Styczen (KU-Life, Copenhagen, Denmark), Aaldrik Tiktak (Netherlands Environmental Assessment Agency, Netherlands), and Matthias Zessner (Vienna University of Technology, Austria). Special thanks also to Phillip Chamberlain and the C&A Foundation for funding this project.

* In memory of Julian Dawson who tragically passed away in the period of finalizing the guidelines.

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1. Introduction

The grey water footprint (GWF) is an indicator of the water volume needed to assimilate a pollutant load that reaches a water body. As an indicator of water resources appropriation through pollution, it provides a tool to help assess the sustainable, efficient and equitable use of water resources. The application of the GWF by different stakeholders (from companies to environmental ngo's and governmental institutions) has shown its diverse usability as an indicator for water resource management.

The GWF is defined as part of the global water footprint standard in *The Water Footprint Assessment Manual* (Hoekstra et al., 2011). The GWF is an indicator of the amount of freshwater pollution that can be associated with an activity. The GWF of a product will depend on the GWFs of the different steps of its full production and supply chain. The GWF is defined as the volume of freshwater that is required to assimilate a load of pollutants to a freshwater body, based on natural background concentrations and existing ambient water quality standards. The GWF is calculated as the volume of water that is required to dilute pollutants (chemical substances) to such an extent that the quality of the water remains above agreed ambient water quality standards.

The Water Footprint Assessment Manual recommends a three-tier approach for estimating diffuse pollution loads entering a water body. The three-tier approach was the outcome of the Grey Water Footprint Working Group of the Water Footprint Network (WFN) in 2010 and is analogue to the tier approach proposed by the Intergovernmental Panel on Climate Change for estimating greenhouse gas emissions (IPCC, 2006). From tier 1 to 3, the accuracy of estimating the load reaching a water body increases, but the feasibility of carrying out the analysis decreases because of the increasing data demand.

Tier 1 simply uses a leaching-runoff fraction to translate data on the amount of a chemical substance applied to the soil to an estimate of the amount of the substance entering the groundwater or surface water system. The fraction is to be derived from existing literature and will depend on the chemical considered. This tier-1 estimate is sufficient for a first rough estimate, but obviously does not describe the different pathways of a chemical substance from the soil surface to surface or groundwater and the interaction and transformation of different chemical substances in the soil or along its flow path.

Tier 2 applies standardized and simplified model approaches and can be used based on relatively easily obtainable data (such as the chemical properties of the chemical substance considered and the topographic, climatic, hydrologic and soil characteristics of the environment in which the chemical substance is applied). These simple and standardized model approaches should be derived from more advanced and validated models.

Tier 3 uses sophisticated modelling techniques and/or intensive measurement approaches. Since this approach is very laborious, available resources should allow for it and the purpose of application should warrant it. Whereas detailed physically-based models of contaminant flows through soils are available, their complexity often renders them inappropriate even for use at tier-3 level. However, validated empirical models driven by

information on farm practices and data on soil and weather characteristics are presently available for use in diffuse-load studies at this level.

Up to date, GWF studies have been based on the tier-1 level and also in the near future this is expected to remain so, at least in practical applications by business and governments. Although it is the most feasible approach of the three tiers, practical applications have often been hampered by lack of guidance and reference values. Values chosen for leaching-runoff fractions used in the calculations were often based on limited information and assumptions. These studies have shown that the GWF methodology as described in *The Water Footprint Assessment Manual* (Hoekstra et al., 2011) could be reinforced through expert guidance on how to best estimate the values of the leaching-runoff fractions.

This has been the reason for WFN to develop the tier 1 supporting guidelines as presented in this report. In order to obtain the necessary expert inputs and feedback, a panel of experts was formed. The GWF Expert Panel contributed to this guidance document by advising on key issues that must be addressed when estimating a GWF at the tier 1 level. The report addresses three subjects: (i) how to estimate leaching-runoff fractions depending on the chemical substance, environmental conditions and application practice; (ii) what water quality standards (maximum allowable concentrations) to use in the calculations; and (iii) what to assume regarding natural background concentrations.

These guidelines support GWF accounting at its simplest level, using the least detailed approach to estimate the GWF in the case of diffuse and direct pollution. Although these guidelines are meant to support GWF accounting at the simplest level, it was quite a task to create guidelines that can be relatively easily applied globally by different stakeholders for different forms of pollution and still be scientifically acceptable. These guidelines are recommended only as a default method, as a screening level method, to be used if time and resources do not allow a more detailed study at tier 2 or tier 3 level. Results obtained from applying these tier 1 supporting guidelines must always be seen in the context of the limitations of the tier-1 approach. The guidelines are based on the current understanding and information available. They will need revision as the understanding of the transport and fate of chemicals from diffuse sources further develops.

2. Objective and scope of the guidelines

These guidelines support determining the parameter values necessary for calculating the GWF at tier 1 level. The guidelines supplement the global water footprint standard in *The Water Footprint Assessment Manual* (Hoekstra et al, 2011). The guidelines help analysts to choose default values for leaching-runoff fractions, maximum allowable concentrations and natural background concentrations, when local data are lacking. This tier-1 estimate is sufficient for a first rough estimate, but outcomes have to be interpreted with extreme care, within the context of the assumptions taken.

Tier 1 uses leaching-runoff fractions to estimate the amount of chemical substances, applied to a soil, that enter the ground- or surface water system. The fraction is to be derived from existing literature or otherwise estimated. These guidelines suggest leaching-runoff fractions to be used based on literature and experience of the GWF Expert Panel and can be considered as best estimates if no other, better information is available. The guidelines show, per type of chemical substance, a range (minimum and maximum) and also an average for the leaching-runoff fraction. The guidelines further show which factors are most relevant when assessing the leaching-runoff fraction. Without any information about the characteristics of the influencing factors at the spot where GWF accounting is done, we advise to apply the average value for the leaching-runoff fraction. Where information on the influencing factors is available, a simple table and equation can be used to derive a more specific estimate of the leaching-runoff fraction. The more specific estimate will fall somewhere in the range between the minimum and maximum value.

Regarding the maximum allowable concentrations in ambient water bodies, *The Water Footprint Assessment Manual* suggests to use local ambient water quality standards. However, for comparative studies, in which GWF estimates for different locations are to be compared, it is recommended to take the same standards throughout the study. Regarding the maximum allowable concentrations in ambient water bodies, these guidelines suggest to select the strictest standard as used in the European Union (EU, 2013), the United States (US-EPA, 2013) or Canada (CCME, 2013). These standards are up to date and scientifically most reliable.

For the natural background concentrations, local data are to be used. Should these not be available, these guidelines suggest using the natural/background concentrations referenced by Chapman (1996).

These guidelines are structured into the following chapters, based on the procedures and parameters necessary for the GWF calculation using tier-1 approach. Chapter 3 summarises how to calculate the grey water footprint for the case of point or diffuse pollution based on *The Water Footprint Assessment Manual*. Chapter 4 assists in estimating the leaching-runoff fractions for diffuse pollution. Chapter 5 suggest which maximum allowable concentrations for ambient water bodies can be used when local data are lacking and Chapter 6 which natural background concentrations.

3. How to calculate the grey water footprint

The methodology and calculation of the grey water footprint (GWF) is described in *The Water Footprint Assessment Manual* (Hoekstra et al., 2011). Here, we provide a summary.

When assessing the GWF of an activity or process, the GWF for each contaminant (chemical substance) of concern has to be calculated separately. The overall GWF is equal to the largest GWF found when comparing the contaminant-specific GWFs.

The GWF is calculated by dividing the pollutant load entering a water body (L , in mass/time) by the critical load (L_{crit} , in mass/time) times the runoff of the water body (R , in volume/time).

$$GWF = \frac{L}{L_{crit}} \times R \quad [\text{volume/time}] \quad (1)$$

The critical load is the load of pollutants that will fully consume the assimilation capacity of the receiving water body. It can be calculated by multiplying the runoff of the water body (R , in volume/time) by the difference between the ambient water quality standard of the pollutant (the maximum acceptable concentration c_{max} , in mass/volume) and its natural background concentration in the receiving water body (c_{nat} , in mass/volume).

$$L_{crit} = R \times (c_{max} - c_{nat}) \quad [\text{mass/time}] \quad (2)$$

By inserting Equation 2 in 1, we obtain:

$$GWF = \frac{L}{c_{max} - c_{nat}} \quad [\text{volume/time}] \quad (3)$$

In the case of **point sources of water pollution**, when chemicals are directly released into a water body in the form of a wastewater disposal, the added load (L) can be estimated by measuring the effluent volume and the concentration of a pollutant in the effluent. More precisely: the pollutant load can be calculated as the effluent volume ($Effl$, in volume/time) multiplied by the concentration of the pollutant in the effluent (c_{effl} , in mass/volume) minus the water volume of the abstraction ($Abstr$, in volume/time) multiplied by the actual concentration of the intake water (c_{act} , in mass/volume). The load can thus be calculated as follows:

$$L = Effl \times c_{effl} - Abstr \times c_{act} \quad [\text{mass/time}] \quad (4)$$

In the case of **diffuse sources of water pollution**, estimating the chemical load is not as straightforward as in the case of point sources. When a chemical substance is applied on or put into the soil, as in the case of solid waste disposal or use of fertilizers or pesticides, it may happen that only a fraction seeps into the groundwater or

runs off over the surface to a surface water stream. In this case, the pollutant load is the fraction of the total amount of chemical substances applied (put on or into the soil) that reaches ground- or surface water. The amount of substance applied can be measured. The fraction of applied chemical substances that reaches ground- or surface water, however, cannot be easily measured, since it enters the water in a diffuse way. Therefore it is not clear where and when to measure. As a solution, one can measure the water quality at the outlet of a catchment, but the load at this point will be the sum of contamination from different sources, so that the challenge becomes to apportion the measured concentrations to different sources. Besides, concentrations may decrease along the way due to decay processes. Therefore, it is necessary to determine the fraction of applied chemical substances that will enter the water system. The simplest method is to assume that a certain fraction of the applied chemical substances finally reaches the ground- or surface water:

$$L = \alpha \times Appl \quad [\text{mass/time}] \quad (5)$$

The dimensionless factor alpha (α) stands for the leaching-runoff fraction, defined as the fraction of applied chemical substances reaching freshwater bodies. The variable *Appl* represents the application of chemical substances on or into the soil (in mass/time), i.e. artificial fertilizers, manure or pesticides put on croplands, urine deposits on pastures by grazing animals, solid waste or sludge put in landfills, etc.

Another approach to estimate the pollutant load entering a water body, mostly relevant in the case of nutrient application in crop cultivation, is by explicitly taking into account the uptake of the chemical substance by plants. The leaching-runoff fraction can then be applied to the surplus after plant uptake and harvest. The surplus is the difference between the application rate (*Appl*) of the chemical substance and the offtake rate (*Offtake*):

$$Surplus = Appl - Offtake \quad [\text{mass/time}] \quad (6)$$

The offtake, the amount of chemical substance taken up by a crop and harvested, can be estimated by multiplying the crop yield and the chemical substance content in the crop.

$$Offtake = Yield \times \text{Chemical substance content in crop} \quad [\text{mass/time}] \quad (7)$$

The load entering a water body can now be calculated as a leaching-runoff fraction beta (β) times the surplus:

$$L = \beta \times Surplus \quad [\text{mass/time}] \quad (8)$$

How to estimate the leaching-runoff fractions α or β will be explained in the next chapter.

GWF calculations are carried out using **ambient water quality standards** for the receiving freshwater body, i.e. standards with respect to maximum allowable concentrations in the water bodies. The reason is that the GWF aims to show the required ambient water volume to assimilate contaminants. Ambient water quality

standards are a specific category of water quality standards. Other sorts of standards are, for example, drinking water quality standards, irrigation quality standards and emission (or effluent) standards. One should take care of using ambient water quality standards. For one particular chemical substance, the ambient water quality standard may differ from one to another water body. Besides, the natural concentration may differ from place to place. As a result, a certain pollutant load can result in one GWF in one place and another GWF in another place. This is reasonable, because the required water volume for assimilating a certain pollutant load will indeed be different depending on the difference between the maximum allowable and the natural concentration.

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 by regional agreement (like in the European Water Framework Directive), they do not exist for all chemical substances and for all places. Most important is, of course, to specify which water quality standards and natural concentrations have been used in preparing a GWF account.

The **natural concentration** in a receiving water body (c_{nat}) is the concentration in the water body that would occur if there were no human disturbances in the catchment. For human-made chemical substances that naturally do not occur in water, $c_{nat} = 0$. When natural concentrations are not known precisely but are estimated to be low, for simplicity one may assume $c_{nat} = 0$. However, when c_{nat} is actually not equal to zero, this results in an underestimated GWF, because the assimilation capacity for the chemical substance would be overestimated.

One may ask why the natural concentration is used as a reference and not the actual concentration in the receiving water body. The reason is that the GWF is an indicator of appropriated assimilation capacity. The assimilation capacity of a receiving water body depends on the difference between the maximum allowable and the natural concentration of a substance. If one would compare the maximum allowable concentration with the actual concentration of a substance, one would look at the remaining assimilation capacity, which is obviously changing all the time, as a function of the actual level of pollution at a certain time.

4. How to estimate the leaching-runoff fraction for diffuse pollution sources

4.1. Overview

The movement of a chemical substance applied on soil is mainly controlled by the physical-chemical properties of a contaminant, environmental factors and agricultural management practices. Therefore, the potential for water contamination by loads from diffuse sources varies from site to site, from chemical substance to substance and from management practice to management practice. The amount of chemical substance that will reach a water body (either ground- or surface water) will depend on the leaching-runoff fraction of the chemical applied. The leaching-runoff fraction is the percentage of a chemical that is lost to groundwater through leaching or to surface water through runoff. Figure 1 gives an overview of the different pathways of pollutants to ground- and surface water. Overland flows, inter flows and artificial drain flows generally end up in surface water within a relatively short time. In most cases, groundwater also reaches surface water (rivers, lakes), but the transport time through deeper groundwater is much longer than the transport time through overland flow, inter flow or artificial drainage directly to surface water streams.

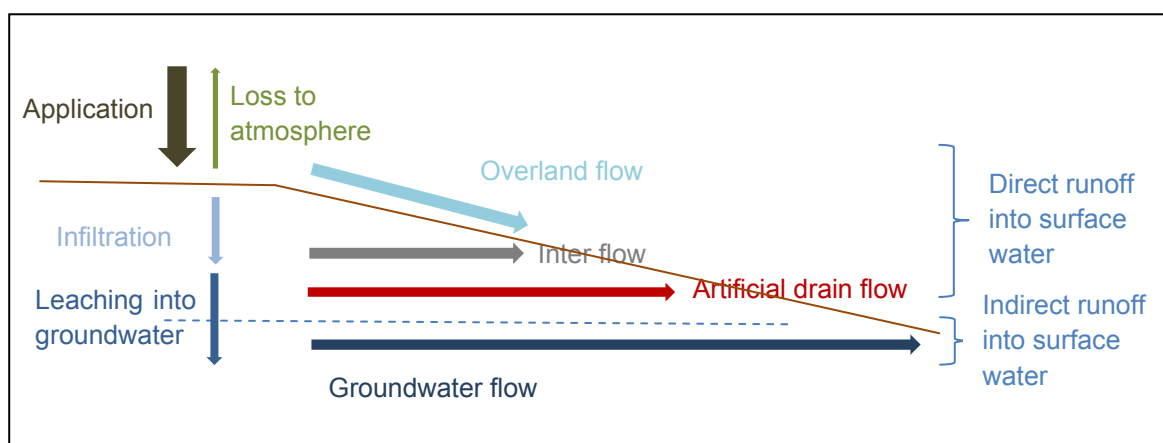


Figure 1. Different flow pathways of contaminants in the case of diffuse pollution.

To calculate the grey water footprint (GWF) of diffuse sources, the actual chemical load reaching a water body has to be estimated. Therefore the application rate of the chemical substance is multiplied by the percentage of the chemical substance reaching a water body, the leaching-runoff fraction. In Equation 5 (Chapter 3), the leaching-runoff fraction is represented by alpha (α).

Leaching and runoff are two different processes, which are influenced in different ways by the same or different factors. The value of α is the resultant of many factors and not an inherent property of the chemical substance, the soil or the way the chemical substance is applied to the field. When estimating the diffuse load of a chemical substance to surface or groundwater at tier 2 or 3, the value of α would be the output of a study of different chemical processes and pathways. At tier 1 level, the value of α is estimated based on (mostly qualitative) information about environmental factors and agricultural practice. Estimating the flows of chemical substances to groundwater and surface water separately is impossible at this level. Therefore, the approach is to estimate the

overall leaching-runoff fraction, without making explicit which part refers to the leaching to groundwater and which part to the direct runoff to surface water. More advanced methods should be used if a differentiation is to be made.

These guidelines suggest default global average leaching-runoff fractions that can be used if no local information is available, which may occur for example when companies aim to assess the GWF of their supply chain without knowing the precise origin of inputs. With some local information, one can make more site-specific estimates of leaching-runoff fractions. There are three categories of influencing factors, which should be considered to estimate the leaching-runoff fraction at tier 1 level:

- physical-chemical properties of the chemical substance applied (like the soil-water partition coefficient K_d or the soil organic carbon-water partition coefficient K_{oc} , and the persistency of the substance);
- environmental conditions (like soil properties and climatic conditions); and
- management practices (like the application rate of the chemical substance, the harvest, the presence of artificial drainage).

In each category, there are different specific factors that influence the leaching-runoff fraction. The list of influencing factors is slightly different per chemical substance group: nutrients, metals, and pesticides, whereby nutrients are further distinguished into nitrogen and phosphorus. Sections 4.2 to 4.5 describe the influencing factors per type of chemical substance.

The state of a factor determines whether the leaching-runoff potential for a chemical substance will be relatively low or high. For nitrogen, for example, soils with little water retention, such as sandy soils, generally have higher leaching (Simmelsgaard, 1998). Per factor i , a certain score s between 0 and 1 for the leaching-runoff potential will be given, based on the state of the factor. A score of 0 means a very low leaching-runoff potential, a score 0.33 a low, a score 0.67 a high, and a score of 1 a very high leaching-runoff potential. If no information about the state of a factor can be obtained, it is suggested to use a score of 0.5 for the corresponding factor.

Each separate factor will influence the leaching-runoff of a chemical substance to a greater or lesser extent. Therefore, weights are given for each factor. A weight w per factor i denotes the importance of the factor. The weights given to the separate influencing factors add up to a total of 100. Tables 3-6 in Sections 4.2 to 4.5 show, per type of chemical substance, the weight per influencing factor and what is the score per factor depending on the state of the factor. The supporting information and maps in Appendices I-II may help to estimate the state of a certain influencing factor if no local data is available.

Once the state of each factor has been determined, the leaching-runoff fraction α can be calculated using the following equation:

$$\alpha = \alpha_{min} + \left[\frac{\sum_i s_i \times w_i}{\sum_i w_i} \right] \times (\alpha_{max} - \alpha_{min}) \quad (9)$$

The value of α will lie somewhere in between the minimum leaching-runoff fraction (α_{min}) and the maximum leaching-runoff fraction (α_{max}). The minimum and maximum leaching-runoff fractions for the chemical substance of concern can be taken from Table 1. Per factor, the score for the leaching runoff potential (s_i) is multiplied by the weight of the factor (w_i). When the scores for all influencing factors are lowest (all scores equal to zero), the resultant leaching-runoff fraction will be equal to α_{min} . When the scores for all factors are highest (all scores equal to one), the resultant leaching-runoff fraction will be equal to α_{max} . An example of how to obtain an estimate of the leaching-runoff fraction based on Equation 9 is shown in Appendix IV.

Table 1. Minimum, average, and maximum leaching-runoff fractions α for nutrients, metals and pesticides.

	Nutrients		Metals	Pesticides
	Nitrogen	Phosphorus		
Minimum leaching-runoff fraction α_{min}	0.01	0.0001	0.4	0.0001
Average leaching-runoff fraction α_{avg}	0.1	0.03	0.7	0.01
Maximum leaching-runoff fraction α_{max}	0.25	0.05	0.9	0.1

If the surplus approach is used to calculate the chemical load entering a water body (Equations 6-8), one can calculate β in a similar way as α :

$$\beta = \beta_{min} + \left[\frac{\sum_i s_i \times w_i}{\sum_i w_i} \right] \times (\beta_{max} - \beta_{min}) \quad (10)$$

Table 2 shows estimates for the minimum and maximum leaching-runoff fractions β for nitrogen and phosphorus. For metals and pesticides, plant uptake is less important so that one can take the simpler approach based on multiplying the fraction α and the application rate (Equation 5).

Table 2. Minimum, average, and maximum leaching-runoff fractions β for nitrogen and phosphorus.

	Nitrogen	Phosphorus
Minimum leaching-runoff fraction β_{min}	0.08	0.0001
Average leaching-runoff fraction β_{avg}	0.44	0.05
Maximum leaching-runoff fraction β_{max}	0.8	0.1

Understanding the influencing factors that determine the leaching and runoff of a chemical substance will help to obtain a better estimate of the leaching-runoff fraction. The next sections will show how, per type of chemical substance, a rough estimate can be made of the leaching-runoff fraction based on (mostly qualitative) information about the local status of different environmental factors and agricultural practice.

4.2. Nitrogen

Nitrogen is one of the most important plant nutrients and forms one of the most mobile compounds in the soil-crop system (National Research Council, 1993). Nitrogen is added to the soil in the form of nitrate (NO_3) or ammonium (NH_4) in artificial fertilizer, as well as in the form of organic nitrogen and ammonia in different types of manure. In most soils, ammonium and organic nitrogen transform to nitrate over time. Nitrogen fixation and deposition are also important nitrogen inputs into the soil. Nitrogen fixation refers to the conversion of atmospheric nitrogen (the gas N_2) into ammonium (NH_4) by bacteria living symbiotically in the roots of leguminous crops. Deposition refers to nitrogen compounds that are emitted from industry, traffic and agriculture and return to the soil via dry and wet deposition. Especially nitrogen fixation can be a major input depending on the crop grown (leguminous crops fix nitrogen and after harvest the leaching can be substantial) and the fertilization level (high level of fertilization generally reduces fixation).

The leaching-runoff of nitrogen to the combined ground-surface water system can be estimated in four different ways, listed from least to most preferred, but also from least to most data-demanding:

1. based on the N-application rate (Equation 5) and the global average value for the leaching-runoff fraction α (Table 1).
2. based on the N-surplus in the soil (Equations 6-8) and the global average value for the leaching-runoff fraction β (Table 2).
3. based on the N-application rate (Equation 5), a rough estimate of the leaching-runoff fraction α (Equation 9) within the range of α_{min} and α_{max} (Table 1) and the estimated nitrogen leaching-runoff potential (Table 3).
4. based on the N-surplus in the soil (Equations 6-8), a rough estimate of the leaching-runoff fraction β (Equation 10) within the range of β_{min} and β_{max} (Table 2) and the estimated nitrogen leaching-runoff potential (Table 3).

The first two calculation methods are simplest, since no local data on soil and climate conditions or agricultural practice are required. However, the outcome will not depend on local factors, while in reality leaching-runoff fractions can vary over a wide range, depending on local conditions. The last two calculation methods are better because they take into account local factors, even though mostly in a qualitative way. The method based on nitrogen surplus is more precise than the method based on the nitrogen application rate. The nitrogen contained in harvested crops represents the greatest and most important output of nitrogen from croplands. The amount of nitrogen taken up varies depending on the crop and yield. Therefore, it is best to subtract the nitrogen offtake due to harvest from the nitrogen application rate before estimating the amount of nitrogen leaching or running off. The nitrogen surplus is the difference between the amount of nitrogen applied and the amount of nitrogen taken up by the crop and harvested. The nitrogen surplus should be estimated using primarily local data. Alternatively, yields can be obtained from national and global statistical databases. N-content in crops can be found in agricultural handbooks and databases, such as listed for example in Appendix I under the heading 'nutrient surplus'.

In the case of nitrogen, leaching and runoff is mainly influenced by:

- environmental factors: N-deposition, soil properties (texture, drainage) and climate (precipitation); and
- agricultural practice: N-fixation, N-application rate, N-offtake through harvest and management practice.

Table 3 can be used to estimate the leaching-runoff potential in a specific location. The table helps to identify the leaching-runoff potential (from very low to very high, with scores from 0 to 1) per influencing factor. The table further shows the importance (weight) per influencing factor. When determining the scores for the leaching-runoff potential per influencing factor, it is generally better to use local data on these factors. If no local data are available, one can choose to derive data from global databases or literature. A few relevant references and maps are provided in Appendix II. For those influencing factors for which no information can be obtained, it is suggested to use a score of 0.5.

The different factors influence the leaching-runoff fraction as follows:

- N-deposition will considerably influence the amount of nitrogen that will leach or run off. The higher the N-deposition, the higher the leaching-runoff potential.
- Regarding soil texture, sandy soils are particularly vulnerable to nitrate leaching because of their low water holding capacity, whereas loamy, silty and clayey soils retain water, and with it nitrogen, more effectively, thus lowering leaching capacity. Losses through runoff are influenced by soil texture opposite to leaching.
- The poorer natural drainage of a soil, the less nitrogen will leach to groundwater, but the higher the probability of runoff towards surface water.
- Rainfall is probably the most important climate factor affecting nitrate leaching and runoff. Heavy rain causes a peak in leaching and runoff, because water flushes nitrate from soil.
- The amount of nitrogen lost through leaching or runoff is related to the amount of nitrogen applied. The higher the application rate, the larger the fraction of loss.
- Depending on the crop grown (and the associated nitrogen uptake) and the yield, the amount of nitrogen exposed to leaching and runoff will differ. The higher the plant uptake and crop yield, the lower the potential of leaching and runoff.
- Management practices such as timing and mode of nitrogen application can affect chemical and transport processes in the soil. Excessive irrigation increases the risk of nitrate leaching (Thompson et al., 2007). Best management practice is highly specific to crop and location (National Research Council, 1993). Here we categorize management practice from 'best' to 'worst'. 'Best' includes a series of measures reducing the risk of leaching-runoff. In order to classify the management practice in a particular situation, the questionnaire provided in Appendix III can be used as a reference. If no information on management practice is available, we suggest using 'best' or 'good' for industrialized countries, 'good' or 'average' for emerging countries and 'average' or 'worst' for developing countries.

Table 3. Factors influencing the leaching-runoff potential of nitrogen. The state of the factor determines the leaching-runoff potential, expressed as a score between 0 and 1. A weight per factor shows the importance of each factor.

Category	Factor		Nitrogen					
			Leaching-runoff potential		Very low	Low	High	Very high
			Score (s)		0	0.33	0.67	1
			Weight* (w)					
			α	β				
Environmental factors	Atmospheric input	N-deposition (g N m ⁻² yr ⁻¹) (see Appendix II Map 1)	10	10	< 0.5	> 0.5	< 1.5	> 1.5
	Soil	Texture (relevant for leaching) (see Appendix II Map 2)	15	15	Clay	Silt	Loam	Sand
		Texture (relevant for runoff) (see Appendix II Map 2)	10	10	Sand	Loam	Silt	Clay
		Natural drainage (relevant for leaching) (see Appendix II Map 3)	10	15	Poorly to very poorly drained	Moderately to imperfectly drained	Well drained	Excessively to extremely drained
		Natural drainage (relevant for runoff) (see Appendix II Map 3)	5	10	Excessively to extremely drained	Well drained	Moderately to imperfectly drained	Poorly to very poorly drained
	Climate	Precipitation (mm) (see Appendix II Map 5)	15	15	0-600	600-1200	1200-1800	> 1800
Agricultural practice	N-fixation (kg/ha)		10	10	0	> 0	< 60	> 60
	Application rate**		10	0	Very low	Low	High	Very high
	Plant uptake (crop yield)**		5	0	Very high	High	Low	Very low
	Management practice		10	15	Best	Good	Average	Worst

* When deriving the load of N to ground- and surface water as a fraction of the N application rate, one should use the weights in the α -column. When deriving the load of N to ground- and surface water as a fraction of the N surplus in the soil, one should take the weights from the β -column.

** These factors do not need to be considered when deriving the load of N to ground- and surface water as a fraction of the nitrogen surplus in the soil, because these factors have then already been accounted for in the surplus calculation.

4.3. Phosphorus

Phosphorus is added to croplands in crop residues, manures and synthetic fertilizers, and from phosphorus-bearing minerals in the soil. A large part of the phosphorus entering the soil-crop system is removed with the harvested crop. The portion of phosphorus not taken up by the crop is immobilized in the soil, incorporated into soil organic matter, or lost through surface or subsurface flows to surface water or groundwater. The majority of phosphorus is lost from agricultural lands through runoff, both in solution (soluble phosphorus) and bound to eroded sediment particles (National Research Council, 1993).

Table 4. Factors influencing the leaching-runoff potential of P. The state of the factor determines the leaching-runoff potential, expressed as a score between 0 and 1. A weight per factor shows the importance of each factor.

Category			Factor		Phosphorus					
					Leaching-runoff potential		Very low	Low	High	Very high
Weight* (w)										
α	β									
Environmental factors	Soil	Texture (relevant for runoff) (see Appendix II Map 2)	15	25	Sand	Loam	Silt	Clay		
		Erosion (see Appendix II Map 9)	20	25	Low	Moderate	High	Very high		
		P-content (g P m ⁻²) (see Appendix II Map 6)	15	20	< 200	200-400	400-700	> 700		
	Climate	Rain intensity	10	15	Light	Moderate	Strong	Heavy		
	Agricultural practice	Application rate**		15	0	Very low	Low	High	Very high	
Plant uptake (crop yield)**		10	0	Very high	High	Low	Very low			
Management practice		15	15	Best	Good	Average	Worst			

* When deriving the load of P to ground- and surface water as a fraction of the P application rate, one should use the weights in the α -column. When deriving the load of P to ground- and surface water as a fraction of the P surplus in the soil, one should take the weights from the β -column.

** These factors do not need to be considered when deriving the load of P to ground- and surface water as a fraction of the P surplus in the soil, because these factors have already been accounted for in the surplus calculation.

Similarly as in the case of nitrogen, the leaching-runoff of phosphorus (P) to the combined ground-surface water system can be estimated in four ways, again listed from least to most preferred and least to most data-demanding:

1. based on the P-application rate (Equation 5) and the global average value for the leaching-runoff fraction α (Table 1).
2. based on the P-surplus in the soil (Equations 6-8) and the global average value for the leaching-runoff fraction β (Table 2).
3. based on the P-application rate (Equation 5), a rough estimate of the leaching-runoff fraction α (Equation 9) within the range of α_{min} and α_{max} (Table 1) and the estimated P leaching-runoff potential (Table 4).
4. based on the P-surplus in the soil (Equations 6-8), a rough estimate of the leaching-runoff fraction β (Equation 10) within the range of β_{min} and β_{max} (Table 2) and the estimated P leaching-runoff potential (Table 4).

The method based on P surplus is more precise than the method based on the P application rate because the amount of P in the harvest is explicitly taken into account. In this method, the amount of P removed from the field by harvesting is subtracted from the P application rate before estimating the amount of P leaching or runoff. The P surplus is the difference between the amount of P applied and the amount of P taken up by the crop and harvested. The P surplus should be estimated using primarily local data. Otherwise yields can be obtained from national and global statistical databases. P-content in crops can be found in agricultural handbooks and databases, such as listed for example in Appendix I under the heading 'nutrient surplus'.

The leaching-runoff potential for phosphorus is mainly influenced by:

- environmental factors: soil (texture, erosion, P-content) and climate (rain intensity);
- agricultural practice: P-application rate, P-offtake through harvest and management practice.

The leaching-runoff potential in a specific location can be estimated with Table 4, which helps to identify the leaching-runoff potential (from very low to very high, with scores from 0 to 1) per influencing factor. The table further shows the importance (weight) per influencing factor. When determining the leaching-runoff potential per factor, it is generally better to use local data. If no local data are available, one can choose to derive data from global databases or literature. A few relevant references and maps are provided in Appendix II. For those influencing factors for which no information can be obtained, it is suggested to use a score of 0.5.

The different factors influence the leaching-runoff fraction as follows:

- Regarding soil texture, clayey and silty soils generally have low infiltration rates and therefore more surface runoff and erosion. These soils are therefore particularly vulnerable to surface runoff of P, whereas loamy and sandy soils have higher infiltration, allowing P to be sorbed in the soil column.
- Soil erosion contributes significantly to the inputs of P into surface water bodies. One can apply the Universal Soil Loss Equation (Wischmeier and Smith, 1978) as a simple equation that attempts to predict the annual average erosion rate through factors describing the rainfall (erosivity, which depends on rainfall energy and intensity), soil (erodibility, which depends on soil texture, structure, organic matter content and permeability), slope and slope length, the vegetation and soil conservation practices. The equation allows also inclusion of modifying factors for vegetation and agricultural practices.
- Increased residual P levels in the soil lead to increased phosphorus loadings to surface water, both in solution and attached to soil particles (National Research Council, 1993). Therefore, the P content in the soil is a critical factor in determining actual loads of P to surface water.
- The higher rain intensities, the higher the probability that P will be transported through overland flow to surface water, either dissolved or with eroded soil.
- The lower the P-application rate, the lower the risk of leaching or runoff.
- Depending on the crop grown (and the associated P uptake) and the yield, the amount of P exposed to leaching and runoff will differ. The higher the plant uptake and crop yield, the lower the leaching-runoff potential.
- Best management practice includes a series of measures reducing the risk of leaching-runoff. In order to classify the management practice in a particular situation, the questionnaire provided in Appendix III can be used as a reference. If no information on management practice is available, we suggest using ‘best’ or ‘good’ for industrialized countries, ‘good’ or ‘average’ for emerging countries and ‘average’ or ‘worst’ for developing countries.

4.4. Metals

All soils naturally contain trace levels of metals, which are primarily related to the geology of the region. Metals added to soil will normally be retained at the soil surface. An important parameter is the so-called distribution coefficient K_d , also called the soil-water partition coefficient. The K_d is expressed in L/kg and defined as the ratio of a chemical's sorbed concentration (mg/kg) to the dissolved concentration (mg/L) at equilibrium. Metals associated with the aqueous phase of soils are subject to movement with soil water, and may be transported to ground water (McLean and Bledsoe, 1992). Most of metal losses, though, are through lateral movement of soil, due to mechanical operations or erosion (Camobreco et al., 1996). Metals, unlike organic chemicals, cannot be degraded. Therefore, sooner or later, metals applied onto the soil will reach a water body either through leaching, runoff or erosion.

Because of the wide range of soil characteristics and various forms by which metals can be added to soil, evaluating the extent of metal retention by a soil is site specific (McLean and Bledsoe, 1992). Changes in the soil environment over time, such as the degradation of organic waste, changes in pH, redox potential, or soil solution composition, due to various remediation schemes or to natural weathering processes may enhance metal mobility. Therefore, field specific models for evaluating the behaviour of metals in soils should be used. Here we attempt to establish a simplified tier 1 approach to estimate the leaching-runoff potential of applied metals to soil, which should only be used if no better method is available.

The leaching-runoff of metals to the combined ground-surface water system can be estimated by multiplying the metal-application rate with the leaching-runoff fraction α (Equation 5). If no local data are available, one can assume the global average value for the leaching-runoff fraction α (Table 1). More precise, but requiring some local data, is to make a rough estimate of the leaching-runoff fraction α (Equation 9) within the range of α_{min} and α_{max} (Table 1) and the estimated metal leaching-runoff potential (Table 5).

The leaching-runoff potential of metals is mainly influenced by:

- the soil-water partition coefficient K_d (which depends on the chemical properties of the metal, but environmental conditions such as pH as well);
- environmental factors (beside the environmental factors that influence the K_d value): soil properties (texture, erosion potential) and climate (rain intensity);
- site management: artificial drainage.

The leaching-runoff potential in a specific location can be estimated with Table 5, which helps to identify the leaching-runoff potential (from very low to very high, with scores from 0 to 1) per influencing factor. The table further shows the importance (weight) per influencing factor. When determining the leaching-runoff potential per factor, it is generally better to use local data. If no local data are available, one can choose to derive data from global databases or literature. A few relevant references and maps are provided in Appendices I-II. For those influencing factors for which no information can be obtained, it is suggested to use a score of 0.5.

Table 5. Factors influencing the leaching-runoff potential of metals. The state of the factor determines the leaching-runoff potential, expressed as a score between 0 and 1. A weight per factor shows the importance of each factor.

Category Factor			Metals				
			Leaching-runoff potential	Very low	Low	High	Very high
			Score (s)	0	0.33	0.67	1
			Weight (w)				
Chemical properties	K_d (L/kg) (see Appendix I, contaminant factors)		30	>1000	1000 – 200	200 – 50	<50
Environ-mental factors	Soil	Texture (relevant for runoff) (see Appendix II Map 2)	15	Sand	Loam	Silt	Clay
		Erosion potential (see Appendix II Map 9)	20	Low	Moderate	High	Very high
	Climate	Rain intensity	15	Heavy	Strong	Moderate	Light
Manage-ment practice	Site manage-ment	Artificial drainage (relevant for runoff) (see Appendix II Map 4)	20	Poorly to very poorly drained	Moderately to imperfectly drained	Well drained	Excessively to extremely drained

The different factors influence the leaching-runoff fraction as follows:

- A high K_d value means that more metal is strongly bound to the solid phase and less available to the aqueous phase, i.e. for leaching. Factors that reduce K_d and thus enhance the mobility of metals include the properties of the metal in question, the quantity and type of soil binding sites (organic matter), the acidity (pH), the concentration of complexing anions (organic and inorganic), and competing cations in soil solution (Camobreco et al., 1996; US-EPA, 1996a). Soil organic matter plays a key role in complexing and retaining metals; the higher the organic matter content, the lower the leaching-runoff fraction, because metals are more strongly adsorbed (McLean and Bledsoe, 1992). As the organic matter in soil decomposes, however, it could release soluble metal-organic complexes (Camobreco et al., 1996). The solubility of heavy metals such as copper, lead, zinc, cadmium, and nickel, typically increase as the pH decreases (National Research Council, 1993). Metal-soil interaction is such that when metals are added at the soil surface, downward transportation does not occur to any great extent unless the metal retention capacity of the soil is overloaded (McLean and Bledsoe, 1992). This means that the higher the concentration of metal in the soil, the higher the leaching-runoff potential.
- Regarding soil texture, sandy soils are particularly vulnerable to metal leaching, whereas loamy, silty and clayey soils retain metals more effectively, but are therefore vulnerable to surface runoff and erosion.
- Soil erosion may contribute significantly to the metal inputs into surface water bodies. The Universal Soil Loss Equation (Wischmeier and Smith, 1978) can be used as a simple equation that attempts to predict the annual average erosion rate.

- The higher rainfall intensities, the higher the probability that metals will be washed out or that the soil erodes, taking along the metals contained in the soil.
- Artificial drainage increases the probability that metals end up in surface water. Soils that are poorly drained will accumulate the metals; in this case, metals can, in the long term, reach groundwater through leaching or surface water through erosion.

4.5. Pesticides

Leaching and runoff of pesticides is strongly influenced by their specific chemical properties. The term pesticides includes different chemical mixtures with different purposes (insecticides, herbicides, fungicides, etc.). They usually include one or more ‘active ingredients’ (specific chemical substances), with different properties and behaviours. Estimating the leaching and runoff potential for all of these compounds is challenging. In addition, technical difficulties and the high costs associated with measuring the fraction of pesticides present in the various compartments over time make a full understanding of the fate and transport of pesticides more difficult (National Research Council, 1993).

The leaching-runoff of pesticides to the combined ground-surface water system can be estimated by multiplying the pesticide-application rate with the leaching-runoff fraction α (Equation 5). If no local data are available, one can assume the global average value for the leaching-runoff fraction α (Table 1). More precise, but requiring some local data, is to make a rough estimate of the leaching-runoff fraction α (Equation 9) within the range of α_{min} and α_{max} (Table 1) and the estimated metal leaching-runoff potential (Table 6).

The leaching-runoff potential of pesticides is mainly influenced by:

- pesticide properties: the soil organic carbon-water partitioning coefficient (K_{oc}) and persistence (half-life);
- environmental factors: soil properties (soil texture, organic matter content) and climate (rain intensity, precipitation);
- agricultural practice.

The leaching-runoff potential of pesticides in a specific location can be estimated with Table 6, which helps to identify the leaching-runoff potential (from very low to very high, with scores from 0 to 1) per influencing factor. The table further shows the importance (weight) per influencing factor. When determining the leaching-runoff potential per factor, it is generally better to use local data. If no local data are available, one can choose to derive data from global databases or literature. A few relevant references and maps are provided in Appendices I-II. For those influencing factors for which no information can be obtained, it is suggested to use a score of 0.5.

Table 6. Factors influencing the leaching-runoff potential of pesticides. The state of the factor determines the leaching-runoff potential, expressed as a score between 0 and 1. A weight per factor shows the importance of each factor.

Category	Factor	Pesticides					
		Leaching-runoff potential	Very low	Low	High	Very high	
		Score (s)	0	0.33	0.67	1	
		Weight (w)					
Chemical properties	K_{oc} (L/kg) (see Appendix I, contaminant factors)	20	>1000	1000 - 200	200 - 50	<50	
	Persistence (half-life in days) (relevant for leaching) (see Appendix I, contaminant factors)	15	<10	10 - 30	30 - 100	>100	
	Persistence (half-life in days) (relevant for runoff) (see Appendix I, contaminant factors)	10	<10	10 - 30	30 - 100	>100	
Environmental factors	Soil	Texture (relevant for leaching) (see Appendix II Map 2)	15	Clay	Silt	Loam	Sand
		Texture (relevant for runoff) (see Appendix II Map 2)	10	Sand	Loam	Silt	Clay
		Organic matter content (kg/m ²) (see Appendix II Map 8)	10	>80	41 - 80	21 - 40	<20
	Climate	Rain intensity (relevant for runoff)	5	Light	Moderate	Strong	Heavy
		Precipitation (mm) (relevant for leaching) (see Appendix II Map 5)	5	0-600	600-1200	1200-1800	> 1800
Agricultural practice	Management practice (relevant for runoff)	10	Best	Good	Average	Worst	

The different factors influence the leaching-runoff fraction as follows:

- The soil organic carbon-water partitioning coefficient (K_{oc}) is the ratio of the mass of a chemical that is adsorbed in the soil per unit mass of organic carbon in the soil to the equilibrium concentration of the chemical in solution. It is the soil-water partition coefficient (K_d) normalized to total organic carbon content. K_{oc} values are useful in predicting the mobility of organic soil contaminants: the lower the K_{oc} value, the lower the adsorption affinity of a chemical, the higher the leaching-runoff potential.
- The persistence of an active ingredient of a pesticide is commonly evaluated in terms of half-life, which is the time that it takes for 50 per cent of a chemical substance to be degraded or transformed. Pesticides with a long half-life are more persistent and therefore have a higher leaching-runoff potential (National Research Council, 1993).
- The soil texture is an important factor, because the texture determines the movement of water, which in turn determines the movement of the pesticides dissolved in water. While leaching generally increases from clayey to sandy soils, runoff decreases.

- The organic matter content in the soil will influence the biodegradability of the active ingredients of a pesticide. The organic matter content is an important variable affecting sorption of the active ingredients onto soil particles. Adsorption retains chemical substances in the soil, thus allowing more time for degradation by chemical and biological processes. Organic matter provides binding sites and is very reactive chemically. Soil organic matter also influences how much water the soil can hold before movement occurs. Increasing organic matter will increase the water-holding capacity of the soil (USDA, 1997).
- The more intense the rainfall, the higher the probability that pesticides will be washed out or that the soil erodes.
- At large rainfall rates, it is likely that more pesticides will reach the groundwater through leaching. Additionally, there is a greater potential that a rainfall event will closely follow application, which can be an important factor in pesticide runoff.
- Management practices such as the mode of pesticide application affect the amount reaching freshwater bodies. Spraying, for instance, may lead to drift away from the field, and spraying to close by streams will increase the risk of pesticides depositing directly onto the water. Best management practice includes a series of measures reducing the risk of leaching-runoff. In order to classify the management practice in a particular situation, the questionnaire provided in Appendix III can be used as a reference. If no information on management practice is available, we suggest using 'best' or 'good' for industrialized countries, 'good' or 'average' for emerging countries and 'average' or 'worst' for developing countries.

5. Which maximum allowable concentration to use

5.1. Introduction

Grey water footprint (GWF) calculations are carried out using ambient water quality standards for the receiving freshwater body (in other words, standards with respect to maximum allowable concentrations). The reason is that the GWF aims to show the required ambient water volume to assimilate chemical substances. For a particular chemical substance, the ambient water quality standard may vary from one to another water body. Besides, the natural concentration may vary from place to place. As a result, a certain pollutant load can result in one GWF in one place and another GWF in another place. This is reasonable, because the required water volume for assimilating a certain pollutant load will indeed be different depending on the difference between the maximum allowable and the natural concentration (Hoekstra et al., 2011).

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 by regional agreement (like in the European Water Framework Directive), they do not exist for all chemical substances and all places (Hoekstra et al., 2011). This is why, if no local information can be obtained, this guideline proposes to use the maximum allowable concentrations as based on the assessment of long term/chronic environmental effects from one of these sources:

- EU (2013) – European priority substances in the field of water policy.
- US-EPA (2013) – US National Recommended Water Quality Criteria - Aquatic Life Criteria.
- CCME (2013) - Canadian Water Quality Guidelines for the Protection of Aquatic Life.

These sources are recommended because the water quality standards included in these references are among the most advanced and they include relatively large sets of parameters¹. They have large application areas as well and are referenced by many countries that establish country-specific standards.

In the following sections, maximum allowable concentrations are suggested for the GWF calculation for the case in which no local standards are available. Separate tables are included for four groups of parameters: nutrients, metals & inorganics, pesticides & organics and ‘other water quality parameters’. Per chemical substance, it is recommended to select the strictest standard from the above three sources. For cross-country studies, it is recommended to use a consistent set of standards, so that differences in national legislations will not affect the GWF calculations. In any case, it is recommended to explicitly mention the standards used.

¹ EC (2008) includes about 35 parameters, US-EPA (2013) 60 parameters and CCME (2013) 125 parameters.

5.2. Nitrogen and phosphorous

The values in Table 7 can be used as maximum allowable concentrations for different forms of N and P. Make sure when calculating the GWF that the chemical substance state (e.g. unionized ammonia-N or total ammonia-N) is the same in the effluent concentration, maximum concentration and natural background concentration. The guideline value for total ammonia is temperature and pH dependent (see Table 8). For phosphorus, the maximum allowable value depends on the natural trophic state of the water body. If no local trophic state values are available, the trigger ranges as given by CCME (2004) can be used. A trigger range is a desired concentration range for phosphorus; if the upper limit of the range is exceeded, it indicates a potential environmental problem, and therefore ‘triggers’ further investigation. Natural physical and chemical water quality variables (e.g., salinity, pH, nutrients) inherently vary within and between ecosystem types, and so the preferred method for determining the trigger ranges is to use similar, high quality reference sites to determine natural levels. These ranges are then categorized according to the trophic status of the reference site (Table 9). This approach provides a trigger range that is relevant to the ecosystem type and locality. In the case that the trophic status of a water body cannot be determined, these guidelines suggest to use the value of 20 µg/L for mesotrophic water bodies to calculate the GWF. For further information, see CCME (2004).

Table 7. Maximum allowable concentration: nutrients.

Nutrients	CAS number ²	Maximum allowable concentration (µg/l)	Referenced guideline (EU ³ , CCME ⁴ , US-EPA ⁵)
Ammonia (NH ₃)	7664-41-7 (unionized)	19 unionized NH ₃ -N* see Table 8 for total NH ₃	CCME
Nitrate (NO ₃) ⁶	14797-55-8	13000 NO ₃	CCME
Nitrite (NO ₂)	14797-65-0	60 NO ₂ -N	CCME
Phosphorus (total)		Ultra-oligotrophic	4
		Oligotrophic	10
		Mesotrophic	20
		Meso-eutrophic	35
		Eutrophic	100

* The unionized ammonia guideline value is expressed as µg ammonia/L. This is equivalent to 16 µg ammonia-N/L (= $19 \times 14.0067 / 17.35052$, rounded to two significant figures)⁷.

² CAS registry is the most authoritative collection of disclosed chemical substance information. Each CAS Registry Number (often referred to as CAS number) is a unique numeric identifier, designated to only one substance. It has no chemical significance and is a link to information about a specific chemical substance (www.cas.org).

³ EU (2013): a long-term standard, expressed as an annual average concentration (AA-EQS) and normally based on chronic toxicity data.

⁴ CCME (2013): long-term exposure guidelines are meant to protect against all negative effects during indefinite exposures. They are determined generally based on chronic toxicity data.

⁵ US-EPA (2013): The Criterion Continuous Concentration (CCC) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. US-EPA derives chronic criteria from long term (often greater than 28-day) tests that measure survival, growth, or reproduction.

⁶ Conversion factors for various nitrate units to mg NO₃/L, as well as additional information can be found in CCME (2012).

⁷ See CCME (2010) for more details.

Table 8. Water quality guidelines for total ammonia for the protection of aquatic life (mg NH₃/L). Source: CCME (2010).

Temperature (°C)	pH							
	6.0	6.5	7.0	7.5	8.0	8.5	9.0	10.0
0	231	73.0	23.1	7.32	2.33	0.749	0.25	0.042
5	153	48.3	15.3	4.84	1.54	0.502	0.172	0.034
10	102	32.4	10.3	3.26	1.04	0.343	0.121	0.029
15	69.7	22.0	6.98	2.22	0.715	0.239	0.089	0.026
20	48.0	15.2	4.82	1.54	0.499	0.171	0.067	0.024
25	33.5	10.6	3.37	1.08	0.354	0.125	0.053	0.022
30	23.7	7.50	2.39	0.767	0.256	0.094	0.043	0.021

Measurements of total ammonia in the aquatic environment are often expressed as mg/L total ammonia-N. The present guideline values (in mg/L NH₃) can be converted to mg/L total ammonia-N by multiplying the guideline values by 0.8224.

Table 9. Total phosphorus trigger ranges. Source: CCME (2004).

Trophic status	Canadian trigger ranges total phosphorus (µg/L)
Ultra-oligotrophic	< 4
Oligotrophic	4-10
Mesotrophic	10-20
Meso-eutrophic	20-35
Eutrophic	35-100
Hyper-eutrophic	> 100

5.3. Metals & inorganics, pesticides & organics, and additional water quality parameters

Tables 10-11 show suggested maximum allowable concentrations for metals/inorganics and pesticides/organics, respectively, for those cases where no local standards are available or for comparative studies. There are some water quality parameters, which are neither listed in the EU standard as priority substances, nor in the CCME and US-EPA guidelines, but are often used by industry to assess their water quality limits. Therefore, if no local standards are available, these guidelines suggest using the values from EEC (1975) concerning the quality required of surface water intended for the abstraction of drinking water (Table 12).

Table 10. Maximum allowable concentrations for metals and inorganics.

Metals & inorganics	CAS number	Maximum allowable concentration (µg/l)	Referenced guideline (EU ⁸ , CCME ⁹ , EPA ¹⁰)
Aluminum	7429-90-5	5 if pH < 6.5 100 if pH ≥ 6.5	CCME
Arsenic	7440-38-2	5	CCME
Boron	7440-42-8	1500	CCME
Cadmium and its compounds	7440-43-9	≤0.08 (represents class I-high quality waters)	EU
Chloride	16887-00-6	120000	CCME
Chlorine	7782-50-5	11	EPA
Chromium (III)	7440-47-3	8.9	CCME
Chromium (VI)	7440-47-3	1	CCME
Copper	7440-50-8	Copper concentration = $e^{0.8545[\ln(\text{hardness})]-1.465} * 0.2$ (if hardness is not known the value is 2)	CCME
Cyanide	57-12-5	5 (as free CN)	CCME
Fluoride	16984-48-8	120	CCME
Iron	7439-89-6	300	CCME
Lead and its compounds	7439-92-1	2.5	EPA
Mercury and its compounds	7439-97-6	0.026	CCME
Molybdenum	7439-98-7	73	CCME
Nickel and its compounds	7440-02-0	4	EU
Reactive chlorine species (total residual chlorine, combined residual chlorine, total available chlorine, hypochlorous acid, chloramine, combined available chlorine, free residual chlorine, free available chlorine, chlorine produced oxidants)		0.5	CCME
Selenium	7782-49-2	1	CCME
Silver	7440-22-4	0.1	CCME
Thallium	7440-28-0	0.8	CCME
Uranium	7440-61-1	15	CCME
Zinc	7440-66-6	30	CCME

⁸ EU (2013): a long-term standard, expressed as an annual average concentration (AA-EQS) and normally based on chronic toxicity data.

⁹ CCME (2013): long-term exposure guidelines are meant to protect against all negative effects during indefinite exposures. They are determined generally based on chronic toxicity data.

¹⁰ US-EPA (2013): The Criterion Continuous Concentration (CCC) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. US-EPA derives chronic criteria from long term (often greater than 28-day) tests that measure survival, growth, or reproduction.

Table 11. Maximum allowable concentrations for pesticides and organics.

Pesticides & organics	CAS number	Maximum allowable concentration (µg/l)	Referenced guideline (EU ¹¹ , CCME ¹² , US-EPA ¹³)
1,2 Dichloroethane	107-06-2	10	EU
1,2,3,4 Tetrachlorobenzene	634-66-2	1.8	CCME
1,2,3-Trichlorobenzene	87-61-6	8	CCME
1,2,4- Trichlorobenzene	120-82-1	24	CCME
1,2-Dichlorobenzene	95-50-1	0.7	CCME
1,3-Dichlorobenzene	541-73-1	150	CCME
1,4-Dichlorobenzene	106-46-7	26	CCME
3-Iodo-2-prpyl butylcarbamate	55406-53-6	1.9	CCME
Acenaphthene	83-32-9	5.8	CCME
Acridine	260-94-6	4.4	CCME
Acrolein	107-02-8	3	EPA
Alachlor	15972-60-8	0.3	EU
Aldicarb	116-06-3	1	CCME
Aniline	62-53-3	2.2	CCME
Anthracene	120-12-7	0.012	CCME
Atrazine	1912-24-9	0.6	EU
Benzene	71-43-2	10	EU
Benzo(a)anthracene	56-55-3	0.018	CCME
Benzo(a)pyrene	50-32-8	0.015	CCME
Bromacil	314-40-9	5	CCME
Bromoxynil	1689-84-5	5	CCME
C ₁₀₋₁₃ Chloroalkanes	85535-84-8	0.4	EU
Captan	133-06-2	1.3	CCME
Carbaryl	63-25-2	0.2	CCME
Carbofuran	1563-66-2	1.8	CCME
Carbon-tetrachloride	56-23-5	12	EU

¹¹ EU (2013): a long-term standard, expressed as an annual average concentration (AA-EQS) and normally based on chronic toxicity data.

¹² CCME (2013): long-term exposure guidelines are meant to protect against all negative effects during indefinite exposures. They are determined generally based on chronic toxicity data.

¹³ US-EPA (2013): The Criterion Continuous Concentration (CCC) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. US-EPA derives chronic criteria from longer term (often greater than 28-day) tests that measure survival, growth, or reproduction.

Pesticides & organics	CAS number	Maximum allowable concentration (µg/l)	Referenced guideline (EU ¹¹ , CCME ¹² , US-EPA ¹³)
Chlordane	57-74-9	0.0043	EPA
Chlorfenvinphos	470-90-6	0.1	EU
Chlorothalonil	1897-45-6	0.18	CCME
Chlorpyrifos (Chlorpyrifos-ethyl)	2921-88-2	0.002	CCME
Cyanazine	21725-46-2	2	CCME
Cyclodiene pesticides		$\Sigma=0.01$	EU
Aldrin	309-00-2		
Dieldrin	60-57-1		
Endrin	72-20-8		
Isodrin	465-73-6		
DDT total		0.025	EU
Para-para-DDT	50-29-3	0.001	EPA
Deltamethrine	52918-63-5	0.0004	CCME
Demeton	8065-48-3	0.1	EPA
Di(2-ethylhexyl)-phythalate (DEHP)	117-81-7	1.3	EU
Di(n-butyl)-phythalate	84-74-2	19	CCME
Diazinon	333-41-5	0.17	EPA
Dicamba	1918-00-9	10	CCME
Dichloromethane	75-09-2	20	EU
Dichlorophenols		0.2	CCME
Diclofop-methyl	51338-27-3	6.1	CCME
Didecyldimethylammoniumchloride	7173-51-5	1.5	CCME
Diisopropanolamine	110-97-4	1600	CCME
Dimethoate	60-51-5	6.2	CCME
Dinoseb	88-85-7	0.05	CCME
Diuron	330-54-1	0.2	EU
Endosulfan	115-29-7	0.003	CCME
Ethylbenzene	100-41-4	90	CCME
Ethylene glycol	107-21-1	192000	CCME
Fluoranthene	206-44-0	0.04	CCME
Fluorene	86-73-7	3	CCME
Glyphosate	1071-83-6	800	CCME
Guthion	86-50-0	0.01	EPA
Heptachlor	76-44-8	0.0038	EPA
Heptachlor-epoxide	1024-57-3	0.0038	EPA

Pesticides & organics	CAS number	Maximum allowable concentration (µg/l)	Referenced guideline (EU ¹¹ , CCME ¹² , US-EPA ¹³)
Hexachloro-cyclohexane	608-73-1	0.01	CCME
Lindane			
Imidacloprid	138261-41-3	0.23	CCME
Isoproturon	34123-59-6	0.3	EU
Linuron	330-55-2	7	CCME
Malathion	121-75-5	0.1	EPA
Methoprene	40596-69-8	0.09 (target organism management value: 0.53)	CCME
Methoxychlor	72-43-5	0.03	EPA
Methyl tertiary butyl ether	1634-04-4	10000	CCME
Methylchlorophenoxyacetic acid	94-74-6	2.6	CCME
Methylmercury	22967-92-6	0.004	CCME
Metolachlor	51218-45-2	7.8	CCME
Metribuzin	21087-64-9	1	CCME
Mirex	2385-85-5	0.001	EPA
Monochlorobenzene	108-90-7	1.3	CCME
Monochlorophenols		7	CCME
Naphthalene	91-20-3	1.1	CCME
Nonylphenol (4-Nonylphenol)	84852-15-3	0.3	EU
Octylphenol	140-66-9	0.1	EU
Parathion	56-38-2	0.013	EPA
Penta-chloro-benzene	608-93-5	0.007	EU
Pentachloro-phenol	87-86-5	0.4	EU
Permethrin	52645-53-1	0.004	CCME
Phenanthrene	85-01-8	0.4	CCME
Phenols (mono- & dihydric)	108-95-2	4	CCME
Phenoxy herbicides		4	CCME
Picloram	1918-02-1	29	CCME
Polychlorinated Biphenyls (PCBs)		0.014	EPA
Propylene glycol	57-55-6	500000	CCME
Pyrene		0.025	CCME
Quinoline	91-22-5	3.4	CCME
Simazine	122-34-9	1	EU
Styrene	100-42-5	72	CCME

Pesticides & organics	CAS number	Maximum allowable concentration (µg/l)	Referenced guideline (EU ¹¹ , CCME ¹² , US-EPA ¹³)
Sulfolane	126-33-0	50000	CCME
Tebuthiuron	34014-18-1	1.6	CCME
Tetrachloro-ethylene	127-18-4	10	EU
Tetrachloromethane	56-23-5	13.3	CCME
Tetrachlorophenols		1	CCME
Toluene	108-88-3	2	CCME
Toxaphene	8001-35-2	0.0002	EPA
Triallate	2303-17-5	0.24	CCME
Tributyltin compounds (Tributyltin-cation)	36643-28-4	0.0002	EU
Trichloro-benzenes	12002-48-1	0.4	EU
Trichloro-ethylene	79-01-6	10	EU
Trichloro-methane	67-66-3	1.8	CCME
Trichlorophenols		18	CCME
Trifluralin	1582-09-8	0.03	EU
Triphenyltin	892-20-6	0.022	CCME

Table 12. Maximum allowable values for additional water quality parameters. Source: EEC (1975).

Water quality parameter	Maximum allowable value
Total suspended solids (mg/l)	25
Chemical oxygen demand (COD) (mg/l O ₂)	30
Dissolved oxygen saturation rate (% O ₂)	70
Biochemical oxygen demand (BOD 5) (mg/l O ₂)	3
Temperature (°C)	22

6. What natural background concentration to use

The general definition of natural background level is the concentration that is present owing to natural and geological processes only, i.e. the background level with no anthropogenic contribution ('preindustrial' levels) (EC, 2011). Natural background concentrations within an environmental compartment may vary by several orders of magnitude between geologically disparate areas, and are determined by various factors, like site-specific bedrock composition and the effects of climate on the degree of weathering. Due to natural dynamic processes like weathering, the addition of organic material (leaves) in autumn and uptake by plants during spring and summer, natural background concentrations may show annual cycles (Quevauviller et al., 2008). This variation of water quality over time and space makes it impossible to establish a 'global' natural background level for individual chemical substances, which also would not be very meaningful because of the great variation (EC, 2011). Therefore we strongly recommend using local data on natural background concentrations.

In more or less pristine rivers, one can assume that natural concentrations are equal to the actual concentrations and thus rely on long-term daily or monthly averages as measured in a nearby measuring station. For disturbed rivers, one will have to rely on historical records or model studies (Hoekstra et al., 2011).

EC (2011), a technical guidance document for deriving environmental quality standards, mentions that the preferred procedure for determining the 'natural' background concentrations in freshwater, will usually be to determine the concentrations in springs or in water bodies in 'pristine' areas in the given region, e.g. headwaters. Another possibility is to measure concentrations in deep groundwater. In some cases, however, the concentration may be higher in the groundwater than in the surface water, for instance because of the groundwater's contact with deep lying mineral rocks or soils and subsequent dilution by rain. Yet another possibility is to use models to simulate soil processes and erosion, in combination with geochemical database (like for instance GTK, 2013).

The Canadian water quality guidelines have used the background concentration approach to establish their objectives (CCME, 2003). The natural background concentrations of a contaminant are estimated and used to define acceptable water quality conditions at a site under consideration. Three general approaches have been used to define background concentrations of water quality variables, which involve:

- Utilization of historically-collected water quality data for the site (i.e., prior to the commencement of activities that could have substantially altered water quality conditions);
- Monitoring contemporary water quality conditions at one or more stations located upstream of contaminant sources; and
- Monitoring contemporary water quality conditions at one or more reference areas, which are generally located nearby the site under consideration but have not been adversely affected by human activities.

If local data are not available and time and budget do not allow determining the natural background concentrations, we suggest using the values in the following tables, which were derived from the

natural/background concentrations referenced by Chapman (1996). Tables 13 and 14 show the values that can be used for nutrients and metals/inorganics, respectively. Natural background concentrations for anthropogenic organic substances and pesticides are zero. More information on how one can obtain country-specific background levels is given in Appendix I.

Table 13. Natural/background concentrations: nutrients

Nutrients	Natural / background concentrations (mg/L)
Ammonium N-NH ₄	0.015
Nitrate N-NO ₃	0.1
N organic	0.26
Phosphate P-PO ₄	0.01

Table 14. Natural/background concentrations: metals & inorganics

Metals & inorganics	Natural / background concentrations	Unit
Aluminum-Al	40	µg/L
Arsenic-As	1	µg/L
Boron-B	30	µg/L
Cadmium-Cd	0.001	µg/L
Chromium-Cr	0.1	µg/L
Cobalt-Co	0.1	µg/L
Copper-Cu	1.4	µg/L
Fluoride-F	100	µg/L
Iron-Fe	50	µg/L
Manganese-Mn	10	µg/L
Molybdenum-Mo	0.8	µg/L
Nickel-Ni	0.4	µg/L
Lead-Pb	0.04	µg/L
Strontium-Sr	100	µg/L
Zinc-Zn	0.2	µg/L
Calcium-Ca	8	mg/L
Magnesium-Mg	2.4	mg/L
Sodium-Na	3.7	mg/L
Potassium-K	1	mg/L
Chloride-Cl	3.9	mg/L
Sulphate-SO ₄	4.8	mg/L
Bicarbonate-HCO ₃	30.5	mg/L
Total suspended solids-TSS	150	mg/L

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Appendix I. Supporting information

General information

1. National Research Council (1993).
2. Thompson et al. (2007).
3. Garrabrants et al. (2010).

Contaminant factors

1. Pesticides:
 - a. AERU (2013).
 - b. US-EPA (1996b).
 - c. Wauchope et al. (1992).
2. Metals:
 - a. US-EPA (1996b).

Estimated K_d values depending on pH (see Appendix II Map 7 if local pH values cannot be obtained):

Metal	Estimated K_d (L/kg)		
	pH = 4.9	pH = 6.8	pH = 8.0
Antimony ^a		4.5E+01	
Arsenic (+3) ^b	2.5E+01	2.9E+01	3.1E+01
Barium	1.1E+01	4.1E+01	5.2E+01
Beryllium	2.3E+01	7.9E+02	1.0E+05
Cadmium	1.5E+01	7.5E+01	4.3E+03
Chromium (+3)	1.2E+03	1.8E+06	4.3E+06
Chromium (+6) ^b	3.1E+01	1.9E+01	1.4E+01
Cyanide ^c		9.9E+00	
Mercury (+2)	4.0E-02	5.2E+01	2.0E+02
Nickel	1.6E+01	6.5E+01	1.9E+03
Selenium ^b	1.8E+01	5.0E+00	2.2E+00
Silver	1.0E-01	8.3E+00	1.1E+02
Thallium ^b	4.4E+01	7.1E+01	9.6E+01
Vanadium ^a		1.0E+03	
Zinc	1.6E+01	6.2E+01	5.3E+02

^a Geometric mean measured value from Baes et al., 1984 (pH-dependent values not available).

^b Determined using an empirical pH-dependent relationship (Figure 10).

^c SCDM = Superfund Chemical Data Matrix (pH-dependent values not available).

- b. US-EPA (1999).
- c. McLean and Bledsoe (1992).
- d. Allison and Allison (2005).

Soil information

1. USDA (2013a).
2. FAO (2013a).
3. SAGE (2013).
4. Koirala (2013).

5. JRC (2013).
6. IIASA (2013).
7. MacDonald et al. (2011).
8. Batjes (2011).
9. Cleveland et al. (2013).
10. Yang et al. (2013).
11. Scharlemann et al. (2011).

Nutrient surplus

1. FAO (2013b).
2. Roy et al. (2006).
3. USDA (2013b)
4. EEA (2005).

Maximum allowable concentrations

1. Australia and New Zealand: ANZECC and ARMCANZ (2000).
2. Austria: Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management (2010).
3. Brazil: CONAMA (2005).
4. Canada: CCME (2013).
5. China: MEP (1994, 2002).
6. European Union: EC (2008).
7. Germany LAWA-AO (2007).
8. Japan: Ministry of the Environment (2010).
9. South Africa: DWAF (1996).
10. United Kingdom: UKTAG (2008, 2013).
11. United States: US-EPA (2012, 2013).

Natural background concentrations

1. As a reference, a global database on actual (not natural) concentrations is available through UNEP (2009).
2. Using the geochemical atlas of GTK (2013), natural background concentrations can be derived, using the guidelines of EC (2011).
3. For several parts of the world, for specific substances, good studies are available, for example: United States: Hem (1985); Austria: Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management (2010); Germany: LAWA-AO (2007); The Netherlands: Osté et al. (2011).

Appendix II. Leaching-runoff influencing factor maps

Map 1. Global N deposition. Source: Cleveland et al. (2013).

Map 2. Global average soil texture. Source: Koirala (2013).

Map 3. Soil drainage class. Source: FAO (2013a).

Map 4. Global map of artificially drained agricultural areas. Source: Feick et al. (2005).

Map 5. Global average annual precipitation. Source: FAO (2013a).

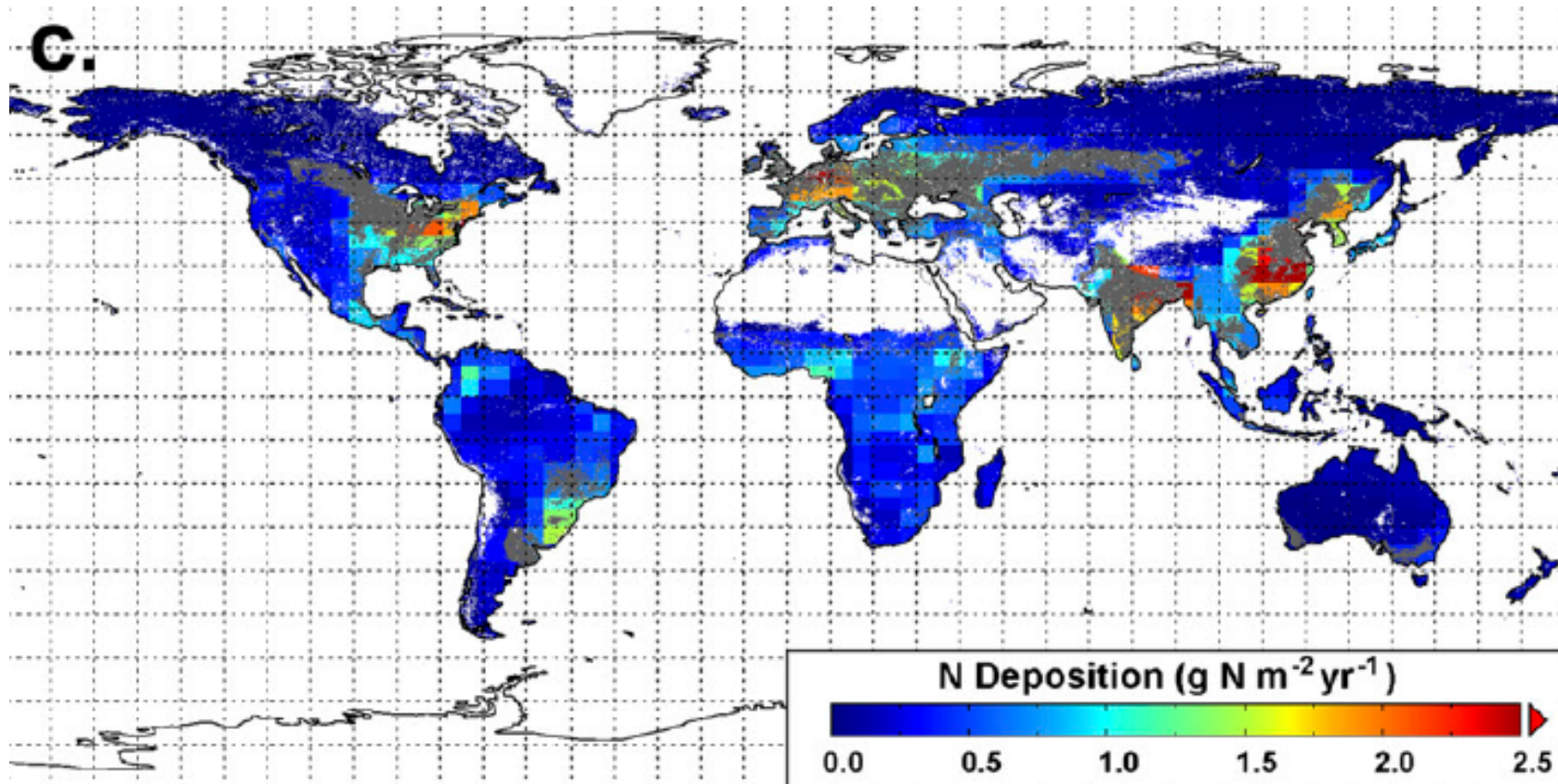
Map 6. Global distribution of total phosphorous content (g P/m^2) in soil. Source: Yang et al. (2013).

Map 7. pH topsoil. Source: FAO (2013a).

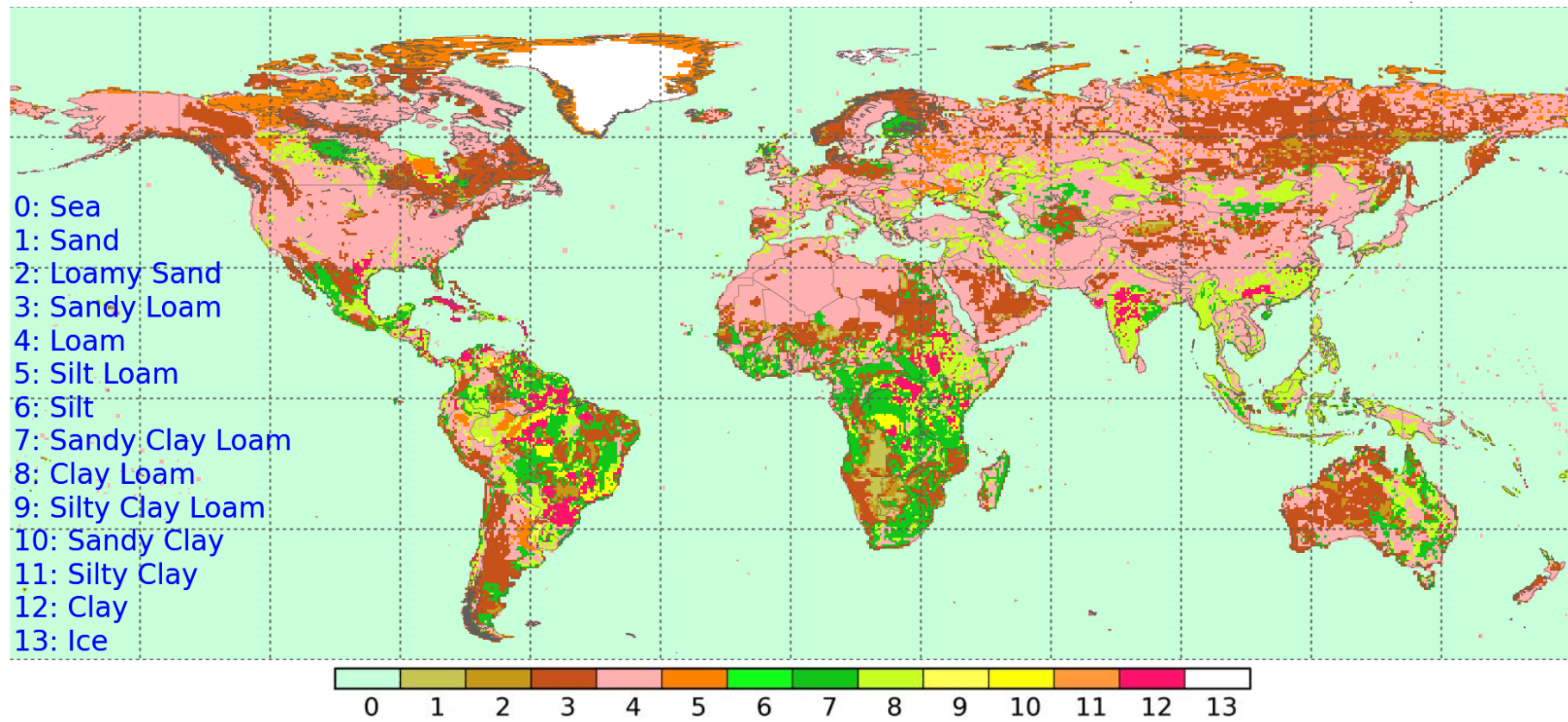
Map 8. Organic carbon. Source: Scharlemann et al. (2011).

Map 9. Water erosion vulnerability. Source: USDA (2013a).

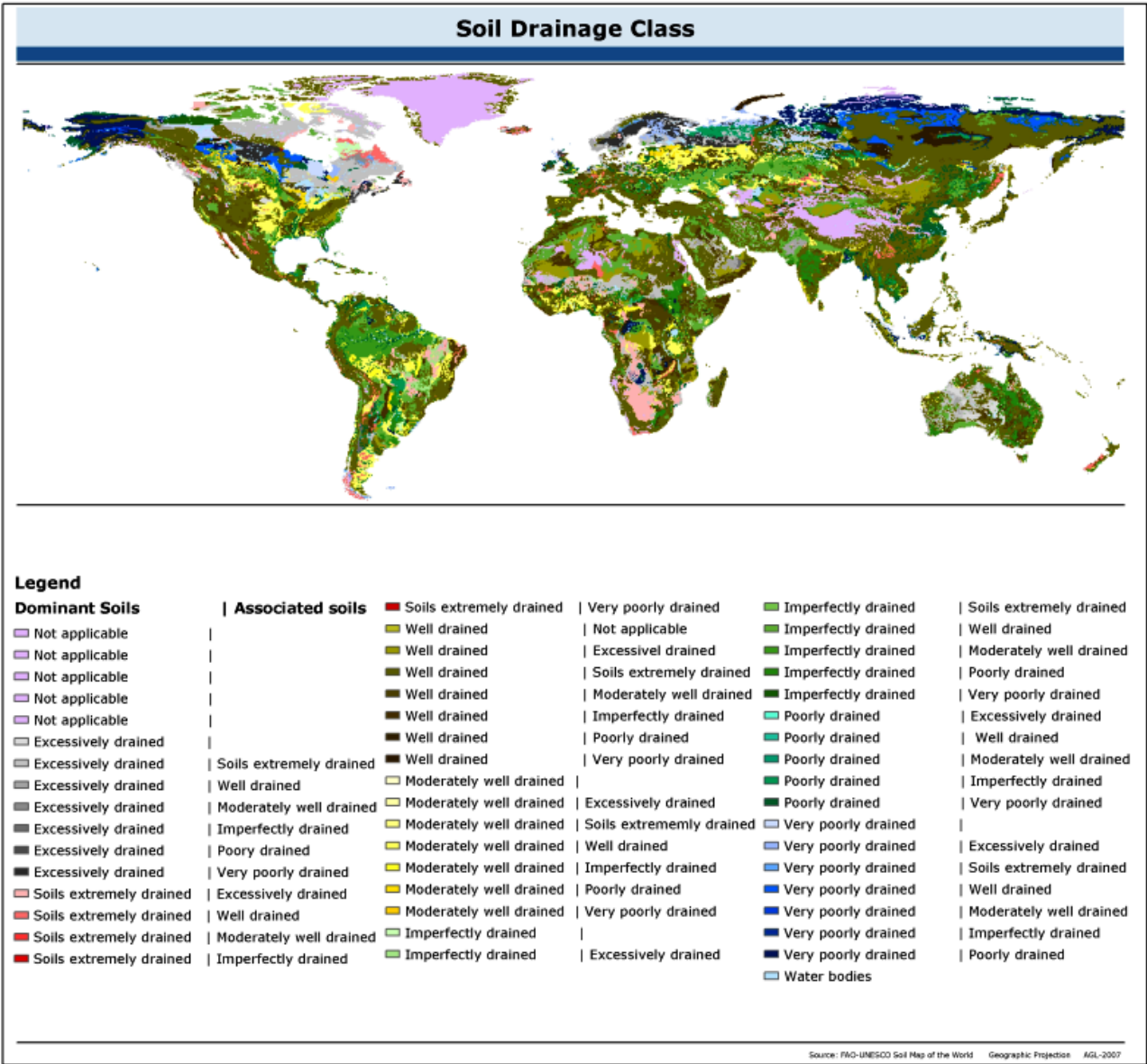
Map 1. Global N deposition. Source: Cleveland et al. (2013).



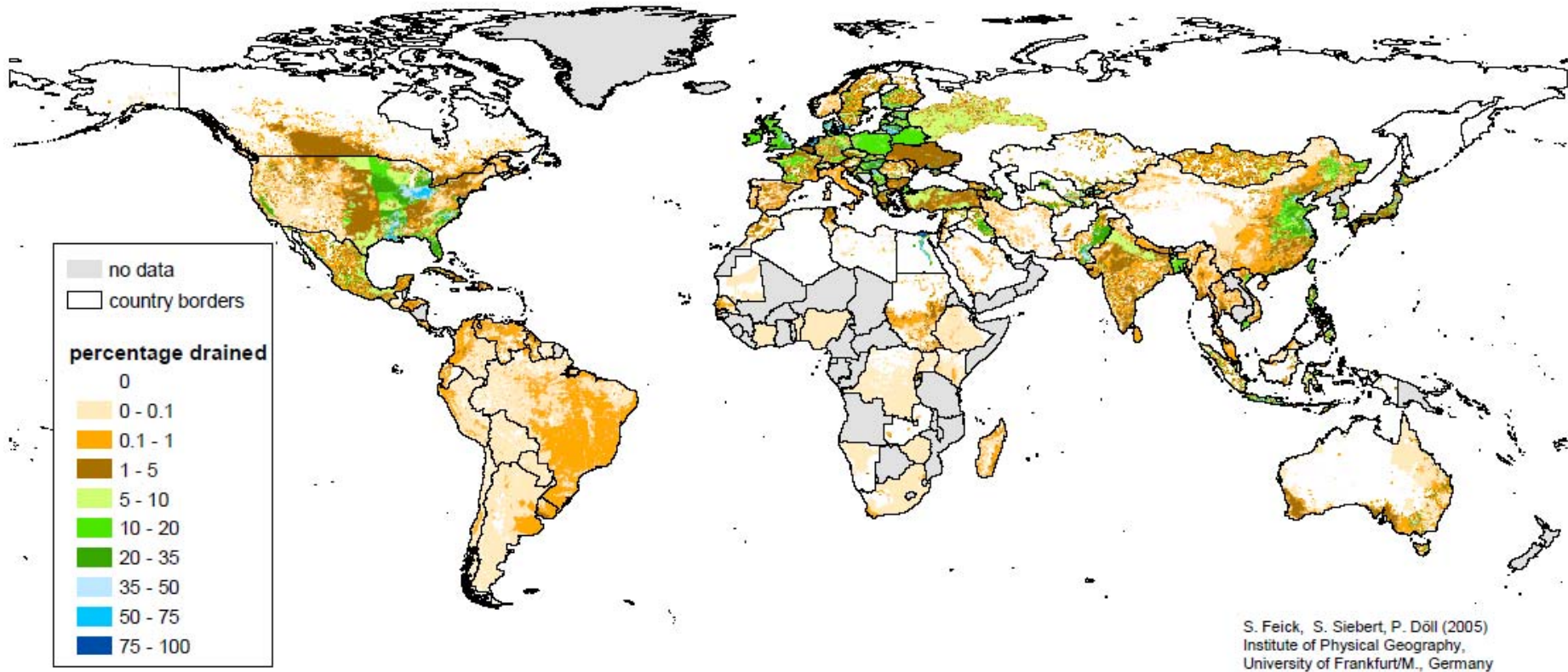
Map 2. Global average soil texture. Source: Koirala (2013).



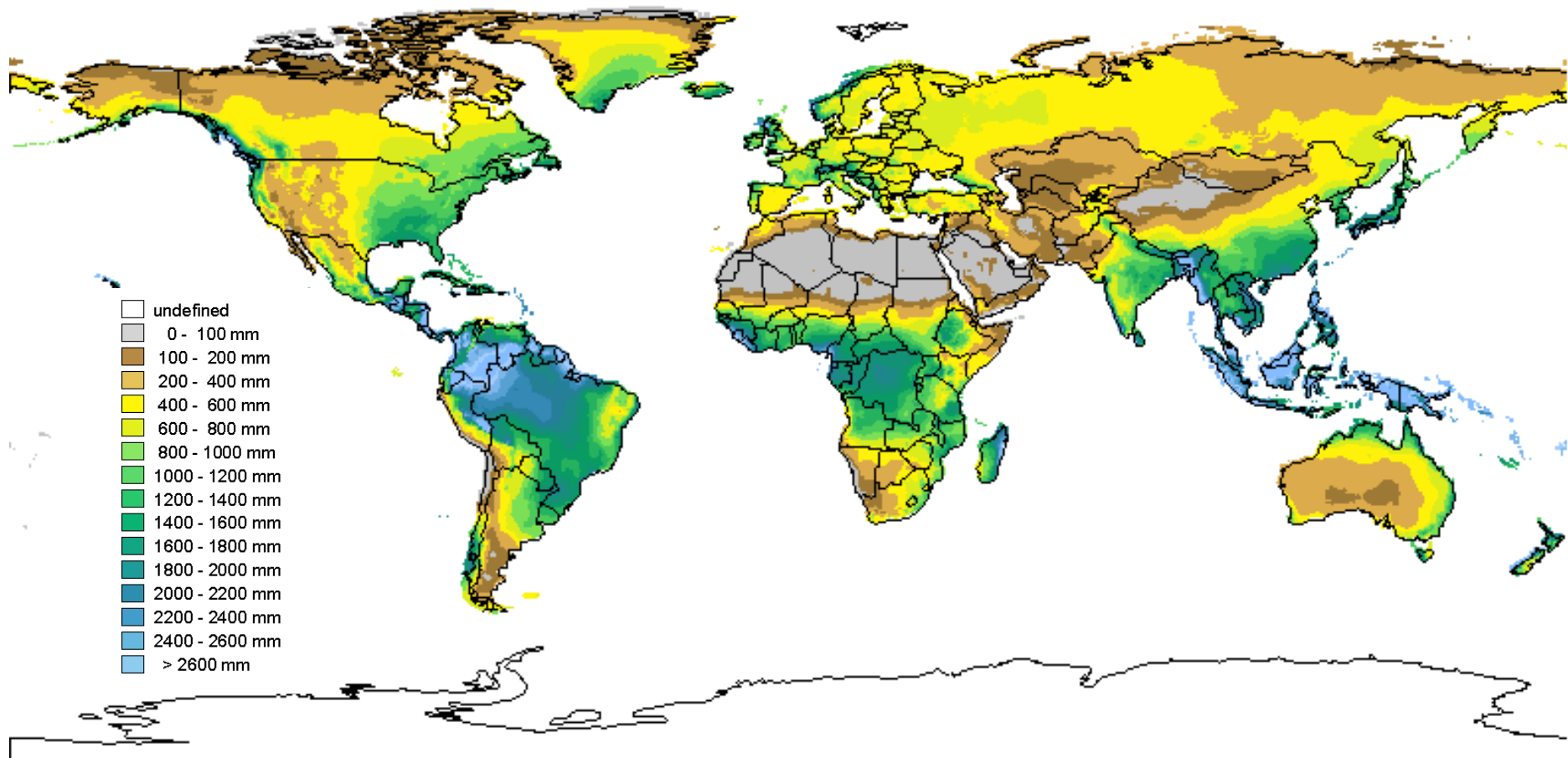
Map 3. Soil drainage class. Source: FAO (2013a).



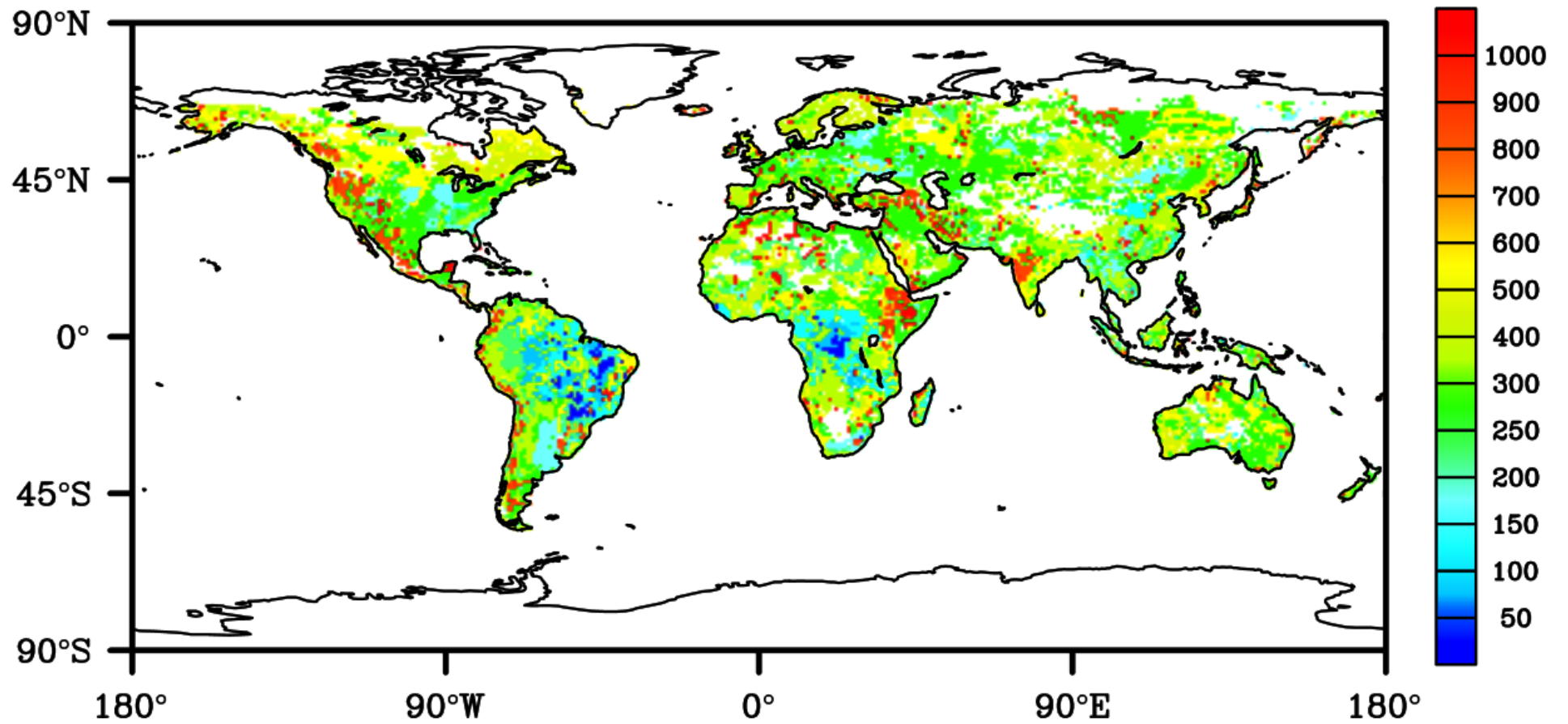
Map 4. Global map of artificially drained agricultural areas. Source: Feick et al. (2005).



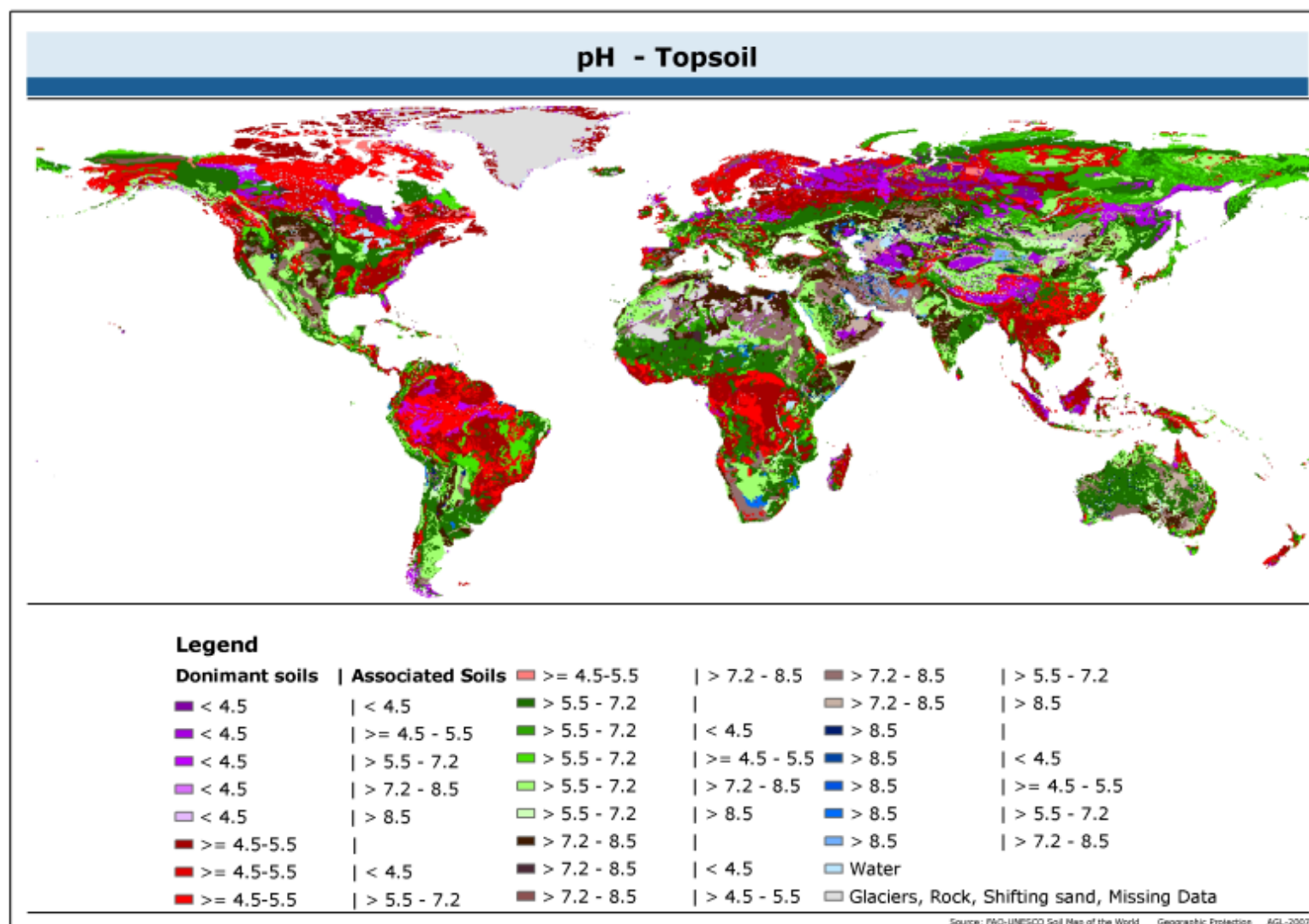
Map 5. Global average annual precipitation. Source: FAO (2013a).



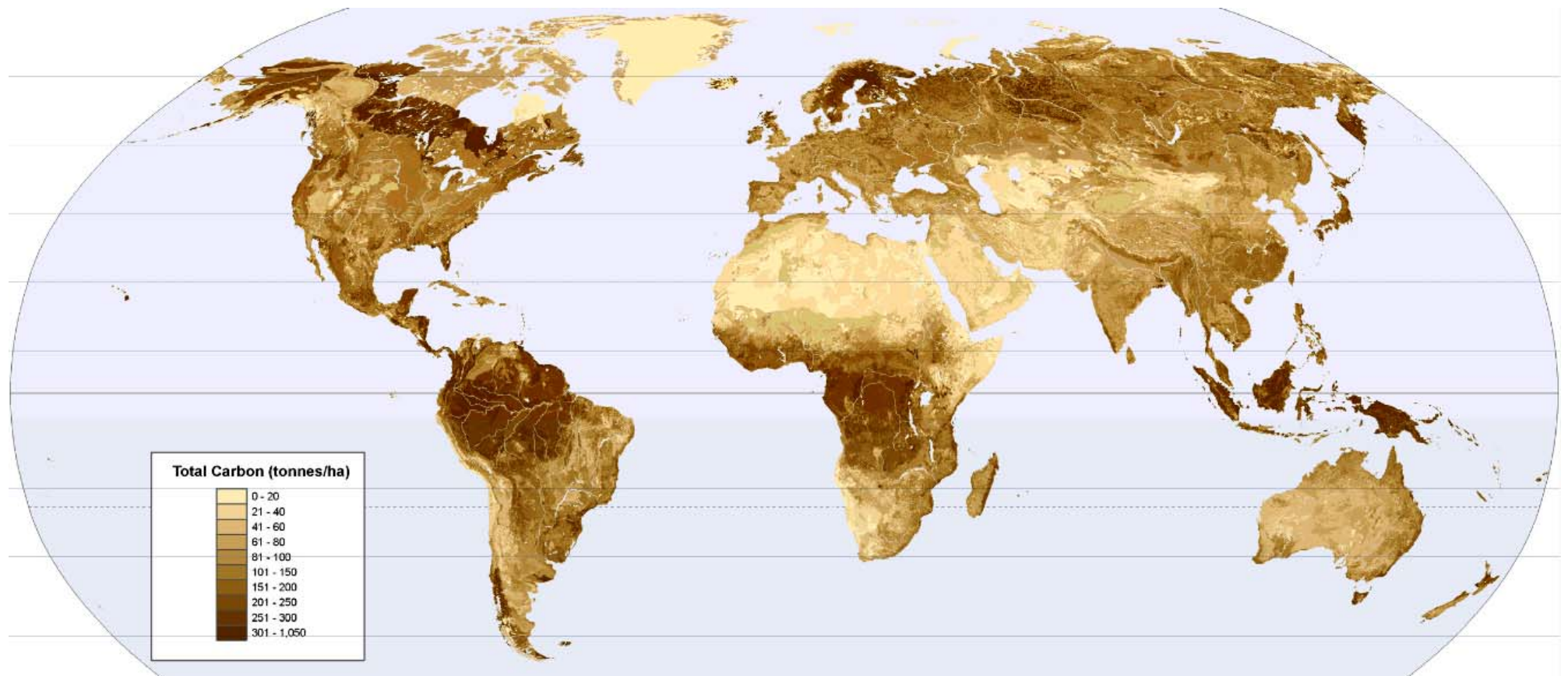
Map 6. Global distribution of total phosphorous content (g P/m^2) in soil. Source: Yang et al. (2013).



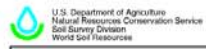
Map 7. pH topsoil. Source: FAO (2013a).



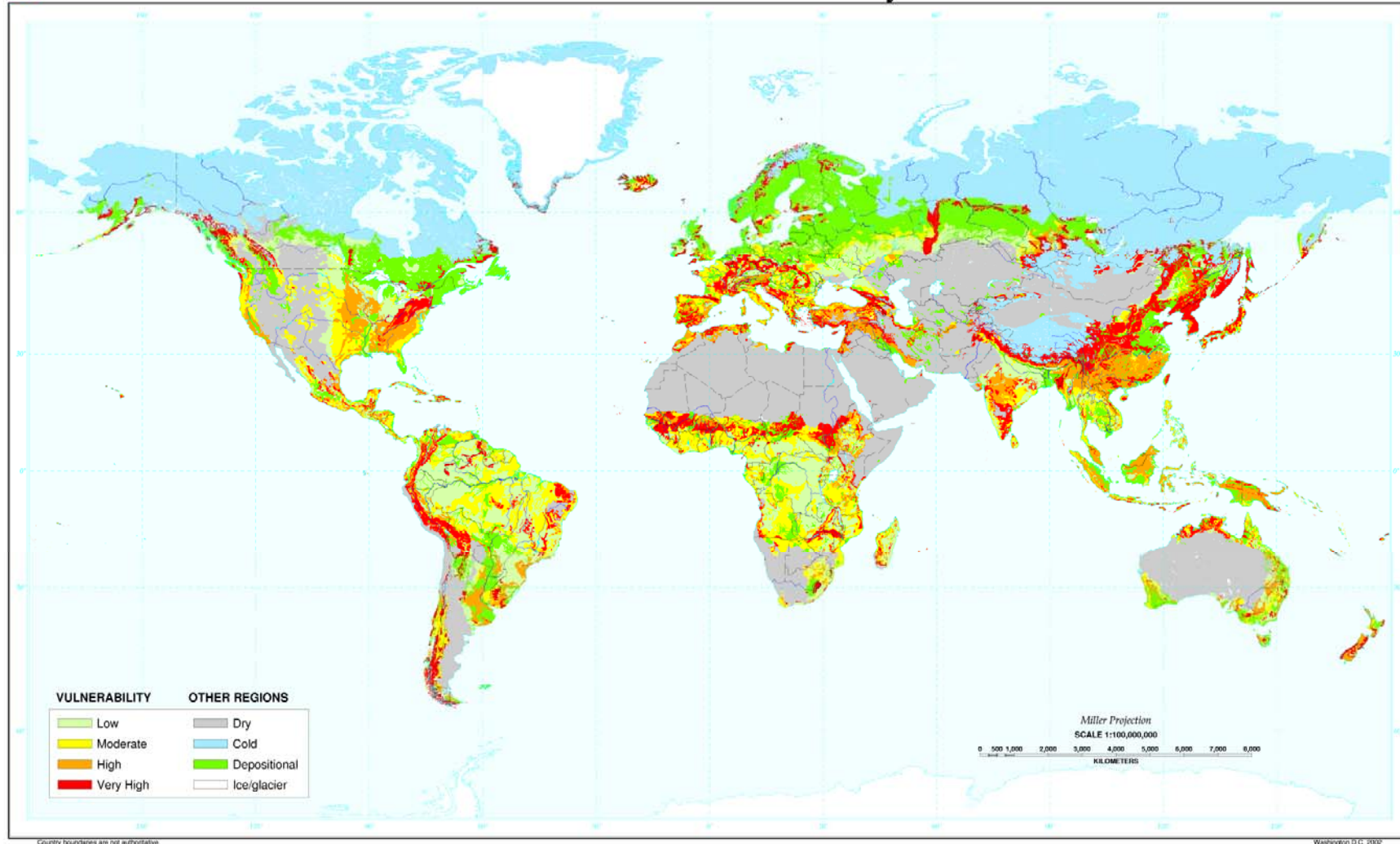
Map 8. Organic carbon. Source: Scharlemann et al. (2011).



Map 9. Water erosion vulnerability. Source: USDA (2013a).



Water Erosion Vulnerability



Appendix III. Agricultural management practice questionnaire

Best management practice includes a series of measures reducing the risk of leaching-runoff of nutrients or pesticides. Table 15 lists some of these measures. By checking whether the various measures listed in this table are used or not, one will be able to obtain a very rough classification of the management practice. If all of the nine listed measures are applied, one can classify the management practice as ‘best’. If 7 or 8 of the listed measures are applied, the management practice is classified as ‘good’; with 5 or 6 measures as ‘average’; and with less than 5 measures as ‘worst’. In the case in which it is not known whether a certain measure is applied, we recommend taking the precautionary approach and answer with ‘no’. If no information on management practice is available at all, we suggest assuming ‘best’ or ‘good’ management practice for industrialized countries, ‘good’ or ‘average’ for emerging countries and ‘average’ or ‘worst’ for developing countries.

Table 15. Agricultural management practice questionnaire

Measure	Applied?	
Controlled application of chemicals. Explanation: through aerial application, considerable losses may occur through spray drift and volatilization; with soil-incorporated application methods, losses are much lower (National Research Council, 1993).	Yes	No
Diffuse pollution mitigation measures. Explanation: buffer zones, stream fencing, and cattle management can reduce the fraction of the contaminant entering a water body (USDA, 1997).	Yes	No
Careful handling of chemicals, e.g. during storage, transport or disposal.	Yes	No
Application immediately before heavy rainfall or irrigation is avoided. Explanation: during heavy rainfall and in case of excessive irrigation, runoff can be very substantial (National Research Council, 1993).	Yes	No
Controlled irrigation. Explanation: sprinkle or drip irrigation do not easily flush out chemicals.	Yes	No
Field is only naturally drained. Explanation: artificial drains can lead to a faster loss of the contaminant (USDA, 1997).	Yes	No
Spreading on frozen ground or foliage is avoided. Explanation: losses through runoff may be severe if this is not avoided (National Research Council, 1993).	Yes	No
Usage of winter cover crops. Explanation: this may reduce runoff (USDA, 1997).	Yes	No
Soil organic matter management. Explanation: returning crop residues and animal wastes to soils helps to maintain soil organic matter content; practices that harvest or destroy residues tend to reduce soil organic matter, leading to greater losses from the field (USDA, 1997).	Yes	No

Appendix IV. Example on how to calculate the grey WF based on these guidelines

A cotton farm in India in the state of Gujarat (see approximate location marked as a red arrow in the map) has a production area of 6 hectares (ha) and a total production of 1.2 tonnes (t) per year. The farm has therefore a yield of 0.2 t/ha per year. The farmer applies 0.0005 t of the pesticide Endosulfan per hectare in the growing season.

In this example we want to calculate the grey water footprint (GWF) of the production process of growing cotton of the farm and the corresponding GWF of the cotton, associated to the application of the pesticide Endosulfan, based on these tier 1 supporting guidelines.



The grey water footprint (GWF) is calculated as described in Chapter 3, by dividing the load of Endosulfan entering the freshwater system (L) by the difference between the maximum and natural concentration of Endosulfan in the freshwater system, following Equation 3 (Chapter 3):

$$GWF = \frac{L}{c_{max} - c_{nat}}$$

In the case of diffuse pollution, the load (L) to the freshwater system depends on the fraction of applied chemical substance on the field that will reach the water system. Therefore it is necessary to know the amount of Endosulfan applied on the field ($Appl$) and the leaching-runoff fraction (α). The load (L) is calculated using Equation 5 (Chapter 3):

$$L = \alpha \times Appl$$

$Appl$ is the amount of chemical applied per area. In this example, Endosulfan has an application rate of 0.0005 t/ha and the farm has a total area of 6 ha. Therefore $Appl = AR \times Area = 0.003$ t.

To estimate the leaching-runoff fraction (α) of the pesticide Endosulfan at tier 1 level, the guidelines in Section 4.5 can be followed. There it is suggested, if no local data on environmental factors or agricultural practices are available, to use the estimated global average leaching-runoff fraction for pesticides in general, which is 0.01 (Table 1, Chapter 4). The load entering the water body would therefore be:

$$L = 0.01 \times 0.003 \text{ t} = 0.00003 \text{ t} (=0.03 \text{ kg})$$

If local data are available (either from the farms directly, through literature or the supporting information and maps in Appendices I and II), one can determine the leaching-runoff potential (Table 6 Section 4.5) and use Equation 9 (Chapter 4):

$$\alpha = \alpha_{min} + \left[\frac{\sum_i s_i \times w_i}{\sum_i w_i} \right] \times (\alpha_{max} - \alpha_{min})$$

The variables necessary are the minimum ($\alpha_{min} = 0.0001$) and maximum ($\alpha_{max} = 0.1$) leaching-runoff fractions for pesticides (Table 1, Chapter 4), as well as the score (s) and weight (w) per leaching-runoff influencing factor, With the help of Table 6, we find the following scores for the leaching-runoff potential per factor:

- The average K_{oc} value of Endosulfan = 11500 L/kg (see AERU, 2013). The score for the leaching-runoff potential is therefore 0.
- The persistency = 50 days (see AERU, 2013), which implies a score for the leaching-runoff potential of 0.67.
- The texture where the farm is located in Gujarat is loamy (see Appendix II, Map 2); the score for the leaching-runoff potential is therefore 0.67 for leaching and 0.33 for runoff. The probability therefore that Endosulfan will rather end up in groundwater than in surface water is higher.
- The organic matter content is between 41 and 80 (see Appendix II, Map 8); the score for the leaching-runoff potential is therefore 0.33.
- For the rain intensity there is no information available, therefore (as suggested in Chapter 4) a score for the leaching-runoff potential of 0.5 is used.
- Net-precipitation is below 600 mm per year (see Appendix II, Map 5), which equals to a score for the leaching-runoff potential of 0.
- For the agricultural management practice, if there is no information available, these guidelines suggest classifying depending on the development stage of the region. In our case the farmers in Gujarat are not trained regarding better management practices, so we assume the management practice to be “worst”. The score for the leaching-runoff potential would therefore be 1.

Once the scores for all influencing factors are obtained, the values can be inserted into the above equation, as well as the corresponding weights as follows:

$$\alpha = 0.0001 + \left[\frac{(0 \times 20) + (0.67 \times 15) + (0.67 \times 10) + (0.67 \times 15) + (0.33 \times 10) + (0.33 \times 10) + (0.5 \times 5) + (0 \times 5) + (1 \times 10)}{20 + 15 + 10 + 15 + 10 + 10 + 5 + 5 + 10} \right] \times (0.1 - 0.0001)$$

$$\alpha = 0.0001 + \left[\frac{45.9}{100} \right] \times 0.0999$$

$$\alpha = 0.046$$

We can recalculate the load of Endosulfan entering the water system with this estimated leaching-runoff fraction:

$$L = 0.046 \times 0.003 \text{ t} = 0.00014 \text{ t} (=0.14 \text{ kg})$$

As the result shows, due to the chemical characteristics of Endosulfan, the local environmental circumstances in Gujarat and the agricultural practices of the farmer, the load of Endosulfan entering the water system is about five times higher than if the load was calculated based on the global average leaching-runoff fraction of pesticides. The example shows once more how important regional specific influencing factors are, when more regional specific studies are to be assessed.

The GWF can then be calculated by using the maximum allowable concentration of Endosulfan (= 0.003 µg/l, see Table 11 Section 5.3) and the natural background concentration (= 0 µg/l, since pesticides do not naturally occur). Note that 0.003 µg/l can be converted to $3 \times 10^{-12} \text{ t/m}^3$.

The GWF of the farm due to Endosulfan is:

$$\text{GWF} = \frac{0.00014 \text{ t}}{3 \times 10^{-12} \text{ t/m}^3 - 0 \text{ t/m}^3} = 46 \times 10^6 \text{ m}^3$$

The GWF of one tonne of cotton produced at the farm is:

$$\text{GWF} = \frac{46 \times 10^6 \text{ m}^3}{1.2 \text{ t}} = 38 \times 10^6 \text{ m}^3/\text{t}$$

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