Development of emission testing values to assess sustainable landfill management on pilot landfills: Phase 2: Proposals for testing values

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Ontwikkeling emissietoetswaarden voor het beoordelen van duurzaam stortbeheer op pilotstortplaatsen

Fase 2: Voorstellen voor emissietoetswaarden

Sinds de jaren negentig wordt internationaal onderzoek verricht naar 'duurzaam stortbeheer'. Het idee hierachter is dat de bron, de stortplaats zelf, schoner wordt, zodat er minder verontreinigingen uit de stortplaatsen kunnen weg lekken. Op deze manier worden de bodem en het nabijgelegen grondwater beschermd. Tot nu toe zijn er nog geen technieken beschikbaar waarvan het effect op grote schaal bewezen is. In dat verband heeft het RIVM, in samenwerking met Energieonderzoek Centrum Nederland (ECN), onderzoek gedaan voor drie vuilstortlocaties in Nederland. Voor deze locaties zijn 'emissietoetswaarden' afgeleid, waarmee kan worden vastgesteld hoeveel schadelijke stoffen er maximaal in het water afkomstig van de stortplaats mag zitten.

Bij duurzaam stortbeheer wordt het afval geïnfiltreerd met water en lucht. Hierdoor treden er processen op die stimuleren dat de verontreinigingen in de stortplaats worden afgebroken of zich binden aan stoffen in het afval. Na een proefperiode van tien jaar zouden de nog aanwezige concentraties in de stortplaats lager moeten zijn. Het gaat om concentraties van organische stoffen (zoals PAK's), anorganische stoffen (zoals metalen) en 'macro-parameters' als nitraat, fosfaat en chloride.

Het 'vertrekpunt' bij de berekening van de emissietoetswaarden zijn de maximaal toegestane concentraties van verontreinigende stoffen in het grondwater en oppervlaktewater dat zich naast de stortplaatsen bevindt. Vandaaruit zijn deze concentraties omgerekend naar de hoeveelheden die het water dat afkomstig is van de stortplaats (percolaat) zou mogen bevatten. Hierbij is rekening gehouden met de mate waarin stoffen in het grond- en oppervlaktewater worden verdund, door bijvoorbeeld regenwater of nabijgelegen grondwater. Ook kunnen stoffen zich binden aan bodemdeeltjes.

Het huidige beleid voor het beheer van stortplaatsen is erop gericht om verontreinigingen in het afval volledig water- en luchtdicht in te pakken (zowel aan de boven- als aan de onderkant). Op deze manier is het risico zo klein mogelijk gemaakt dat de bodem en het grondwater verontreinigd raken. Een nadeel is dat eeuwigdurende en omvangrijke nazorg nodig is. Aangezien de verontreiniging niet wordt afgebroken, moeten de isolatiematerialen die op den duur poreus worden en gaan lekken, regelmatig worden vervangen. Hieraan zijn aanzienlijke kosten verbonden.

Trefwoorden: duurzaam stortbeheer, emissietoetswaarden, stortplaatsen, grondwater, risicobeoordeling, ORCHESTRA, geochemisch transportmodel

Abstract

Development of emission testing values to assess sustainable landfill management in pilot landfills

Phase 2: Proposals for testing values

International research into sustainable landfill management has been carried out since the 1990s. The idea of this is that the source, the landfill itself, becomes cleaner, so that fewer harmful substances are emitted by landfills, and the surrounding soil and groundwater are protected. Up to now, there have been no techniques available whose effectiveness has been proven on a large scale. In that regard, the RIVM, in cooperation with the Energy Research Centre of the Netherlands (ECN), was asked to conduct research into three pilot landfills in the Netherlands. For these locations 'emission testing values' were derived that can be used to determine which emissions from landfills into the soil and groundwater are acceptable.

With sustainable landfill management, the waste is actively infiltrated with water and air (active treatment). This causes processes that stimulate the degradation and binding of the substances in the landfill during a trial period of approximately ten years. After approximately ten years, the concentrations of substances remaining in the landfill should be lower: that is, concentrations of organic substances (such as PAHs), inorganic substances (such as metals) and macroparameters (such as nitrate, phosphate and chloride).

The "starting point" in the calculation of the emission testing values is the maximum allowable concentration of substances in groundwater and surface water next to the landfills. From there, these concentrations are converted to quantities in the landfill leachate. Account is taken of the extent to which substances are diluted, by for example rainwater or groundwater nearby. In ground- and surface water substances can also bind to soil particles.

The current policy for landfill management is focused on the complete containment of substances in the waste (waterproof and airtight, with a top cover and bottom liner). The purpose of this is to minimize the risk of contaminating the soil and the groundwater. A disadvantage is that constant and comprehensive after care is needed. Since the contaminants are not reduced, the insulation materials, which eventually become porous and start leaking, must be replaced regularly, involving considerable costs.

Keywords: Sustainable landfill management, emission testing values, landfill, groundwater, risk assessment, ORCHESTRA, geochemical transport model.

Preface

This report describes the process of deriving emission testing values and as a result the emission testing values for three pilot landfills prior to applying sustainable landfill management. Because of the novelty of sustainable landfill management in The Netherlands, it took a great deal of effort and time to derive the emission testing values and to describe the methods used, in this report. This process involved extensive contact with several counterparts within a working group. We would like to thank the following members of this group for their input and efforts during the derivation process: Mr W. Kattenberg (Chair, Ministry of Infrastructure and Environment), Mr J van der Gun (Secretary, BodemBeheer B.V.), Mr H. Scharff (Afvalzorg), Mr H. Woelders (Attero), Mrs J. Wezenbeek (formerly Grontmij, currently RIVM), Mr D. Britwhistle (North Holland Province), Mr P. Bijvank (Flevoland Province) and Mr M. Romviel (North Brabant Province).

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Extended summary

Introduction
In accordance with the current regulatory frameworks for landfills in The Netherlands, it is mandatory to seal the landfill site completely with side, bottom and top liners in order to prevent any water from entering the site after the landfill has been closed to further waste deposit. This practice preserves the waste, including the enclosed pollutants. The protective liner covering the landfill (top cover) must be replaced at regular intervals because of its limited lifespan (estimates range from 50 to 75 years) – at considerable cost. As a consequence, responsibility for present day waste – namely for landfill management and good groundwater quality – is transferred to future generations.

Since the 1990s, research has been carried out on sustainable landfill management, which aims to reduce the impact of harmful substances in landfills on the soil, groundwater and surface water under and next to the landfill. The biological degradation and immobilization of substances within the landfill site are stimulated by the controlled addition of water and air into the landfill material (so-called active treatment). The idea behind sustainable landfill management is to reduce the emission potential of the waste to a level at which the use of liners is no longer needed, leading to a situation in which there is a minimal need for long-term aftercare. This approach should result in low levels of remaining harmful substances and emission potential, thus protecting the groundwater and surface water quality. In this way, maintenance costs can be significantly reduced and future generations will have to deal with fewer harmful emissions from the landfills and fewer consequences of ground- and surface water pollution.

To make sustainable landfill management possible in the future, the Dutch regulatory framework for landfills must be changed. The first step is to allow pilot projects to test whether the desired result of sustainable landfill management is achievable in practice and within a reasonable time. Such an experiment will be carried out at three landfill sites in The Netherlands in order to study the long-term processes: Braambergen in Almere, Kragge II in Bergen op Zoom (hereafter Kragge) and Wieringermeer in Middenmeer).

Objectives
To determine whether the pilot projects are successful, the Ministry of Infrastructure and Environment asked the RIVM and ECN to develop a set of landfill-specific criteria that will serve as a reference framework against which the emissions from the pilot landfills can be compared. These criteria will be called the ‘emission testing values’ or ETVs. A list of ETVs was derived for each of the pilot landfills, resulting in three sets of landfill-specific ETVs.

The aim of this reference framework is to determine whether the emissions from the landfill are sufficiently reduced after the period of active treatment (approximately ten years). If, after the period of active treatment, the concentrations in the leachate of the pilot landfills have improved and are equal to or below the ETVs, the pilot project will be deemed to be successful and a top cover is no longer mandatory. If the pilot experiments are successful and the pilot landfills meet the designated criteria (concentrations in the leachate), national policy on how to deal with landfills will be amended to permit the use of sustainable landfill management at the remaining designated landfills (approximately 20) in The Netherlands.

Relevant laws and regulations

There are various policy frameworks in the Environmental Management Act, the Soil Protection Act and the Water Act that are important for the Introduction Sustainable Landfill Management project (hereafter IDS project). On the basis of these policy frameworks requirements are set with regard to the way in which waste can be landfilled and to what extent this is allowed to have an impact on the soil and groundwater.

1. The landfilling of waste

In The Netherlands the Decree on Landfills and Landfill Bans (Ministry of VROM, 1997) applies to the landfilling of waste. This decree has adopted the policy set out in the EU Directive 1999/31/EG on the landfilling of waste. In addition, the Decree on Landfilling and Soil Protection and the associated Implementation Directive apply (Ministry of VROM, 1993).

2. Soil

For soil, the Soil Quality Decree (Bbk) is of particular importance, including the section on building materials. This decree describes the policy and requirements for the re-use of (slightly contaminated) soil and the use of building materials in large-scale soil applications.

3. Groundwater

For groundwater, the European Water Framework Directive (WFD) and the daughter directive, the Groundwater Directive (GWR), apply. In The Netherlands the stipulations of the GWR have been adopted in the Water Act and the Water Decree. For the IDS project, article 6 of the GWR is of particular importance. This article describes the measures aimed at preventing or limiting the input of hazardous substances and pollutants into groundwater.

Principles and assumptions

To derive the emission testing values several principles and assumptions were made. The most important of these are shown below.

- A source-path-receptor model is used in which points of compliance (POCs) are located in either groundwater (gw) or surface water (sw) (see Figure S1.1). Which path is relevant depends on the pilot landfill and the substances in the landfill. In this study the scenario for surface water is relevant only to the Wieringermeer pilot landfill.
- The model used to derive the ETVs consists of three POCs: POC0, POC1 and POC2. The environmental protection criterion is linked to POC2. $\mathrm{POC2}_{gw}$ is located in groundwater 20 metres downstream of the landfill (infiltration situation). The environmental protection criterion includes both human and ecological protection targets. For $\mathrm{POC2}_{gw}$ the environmental protection criterion is equal to the protection targets for groundwater, these being 1) the maximum permissible risk (MPR) for metals, 2) a negligible risk (NR) for organic substances and 3) the Dutch drinking water standards (only if those are lower than the MPR or NR). In surface water, $\mathrm{POC2}_{sw}$ is located in the channel next to the ditch surrounding the Wieringermeer pilot landfill (seepage situation). The environmental protection criterion is equal to the protection targets for surface water, these being either 1) the yearly average environmental quality standard (JG-MKN) or 2) the MPR for surface water for metals and organic substances (the MPR is used only if no JG-MKN exists) and 3) the local authority-determined protection targets for macroparameters.

- POC1$_{gw}$ is located in the first metre of the saturated zone of the aquifer under the base of the landfill (infiltration situation). POC1$_{sw}$ is located in the ditch next to the landfill (seepage situation).
- POC0 is located in the leachate drains inside the landfill which are located just above the bottom liner.
- Exceptions: in cases where substances did not reach the designated POC in groundwater or surface water within the specified time frame because of binding to soil particles, it was investigated whether the use of average concentrations equal to the environmental protection criterion for soil (MPR) under the landfill over the total soil volume between POC0 and POC2$_{gw}$/POC1$_{sw}$ (20 metres) would provide feasible ETVs.
- The landfill-specific ETVs are calculated for a time frame of 500 years. Meaning that after the period of active treatment the groundwater and surface water are protected for 500 years if the leachate complies with the ETV. The assessment of the leachate coming from the landfills will take place after a period of active treatment of approximately ten years.
- The local background concentrations in groundwater are taken into account for metals and macroparameters such as ammonium, sulphate and chloride when setting the Environmental protection criteria at POC2.
- The bottom liner of the landfill is assumed to be no longer functional after the period of active treatment and the concentration of substances in leachate coming from the landfill is assumed to be constant.

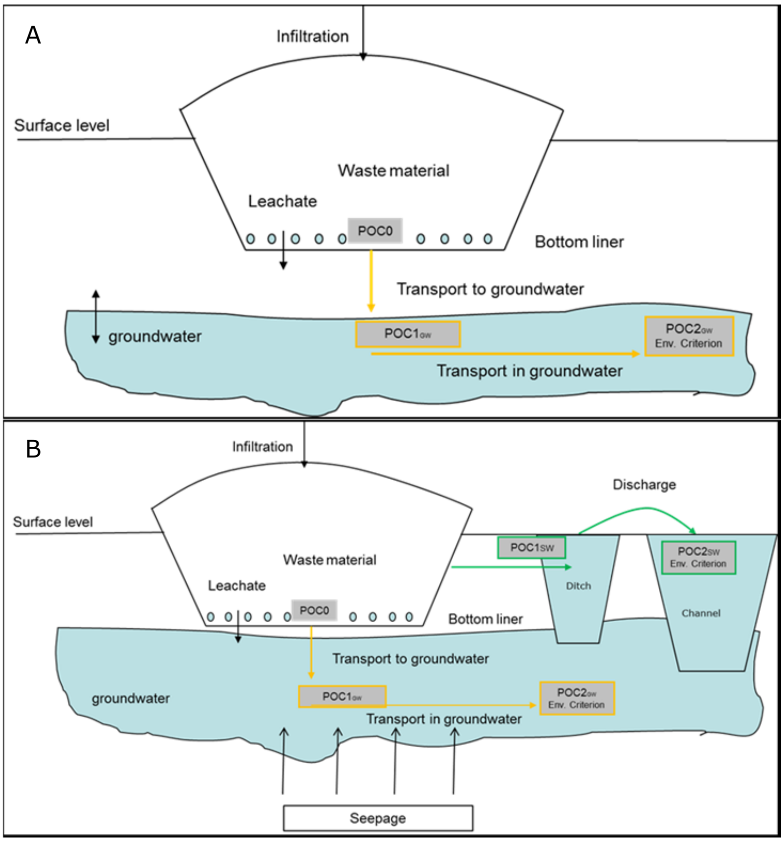

Figure S1.1: Conceptual model of the landfill and its surroundings. Figure A shows an infiltration situation. The yellow arrows indicate the conceptual pathway of the leachate towards groundwater. The green arrows in Figure B indicate the conceptual pathway of the leachate towards the surface water in a seepage situation.

- The infiltration of rainwater into the pilot landfill is assumed to be 300 mm/year, in accordance with the average Dutch net infiltration, and the same amount will flow out of the bottom of the landfill into the underlying soil. This is consistent with other policy fields (e.g. derivation of emission limits in the Soil Quality Decree for the re-use of building materials and large soil applications).
- The environmental criterion at POC2 in both groundwater and surface water is converted into a concentration in the leachate in POC0 using a backward calculation. To convert the environmental criterion at POC2 into an emission testing value at POC0, the reactive transport model ORCHESTRA is used.
- The relevant substances have been selected (A) on the basis of a generic list of substances that are deemed relevant in the regulatory framework on landfills and (B) from landfill-specific substances measured in accordance with the requirements for landfill permits (see Table S1.1).

Table S1.1: List of relevant substances, based on the generic list of substances from the regulatory framework for landfills with landfill-specific additions based on the requirements for landfill permits

Metals	**Organic substances**	
Arsenic Cadmium Chrome Copper Mercury Lead Nickel Zinc	*VOX* Vinylchloride Dichloromethane 1,1 dichloroethane 1,2 dichloroethane 1,1 dichloroethene 1,2 dichloroethene (cis,trans) Dichloropropane (1,2) Dichloropropane (1,3) Trichloromethane (chloroform) 1,1,1 trichloroethane 1,1,2 trichloroethane Trichloroethene (tri)	*Mineral oil* Sum EC10-EC40 Aliphatic EC5-EC6 Aliphatic EC6-EC8 Aliphatic EC8-EC10 Aliphatic EC10-EC12 Aliphatic EC12-EC16 Aliphatic EC16-EC21 Aromatic EC5-EC7 Aromatic EC7-EC8 Aromatic EC8-EC10 Aromatic EC10-EC12 Aromatic EC12-EC16 Aromatic EC16-EC21 Aromatic EC21-EC35
Macroparameters		**Site-specific additions**
Chloride Sulphate N-Kjeldahl/ ammonium	*PAH* PAHSum 10 Naphtalene Phenantrene Antracene Fluoranthene Chrysene Benzo(a)antracene Benzo(a)pyrene Benzo(k)fluoranthene	Phosphate Cyanide Fenols *BTEX* Benzene Xylene Toluene Ethylbenzene

Modelling of transport in soil and groundwater

For the modelling of the ETVs several assumptions were made. The most important assumptions are presented below:

- The infiltration of rainwater from the pilot landfill into the underlying soil is 300 mm/year.
- The unsaturated zone under the landfill has a generic thickness of 1 metre, and each 1 m^2 of unsaturated zone receives 300 litres of landfill leachate per year (300 mm/year).
- The ORCHESTRA model calculates concentrations of substances in the unsaturated zone and upper metre of the saturated zone (=$POC1_{gw}$) as a function of time. The model is based on published thermodynamic

(geochemical) sorption reactions in combination with one-dimensional transport.
- The geochemical approach allows site-specific calculations based on the local chemical soil properties and has been used previously for deriving emission limit values for the re-use of building materials and large soil applications in the Soil Quality Decree.
- The geochemical model is validated by laboratory data and field data (references in main report).
- In the case of infiltration into groundwater, adsorption of the substances in soil will take place over the first metre of the unsaturated zone and the first metre of the saturated zone. No binding will take place in the saturated zone between $POC1_{gw}$ and $POC2_{gw}$, which is a distance of 20 metres.
- In the case of seepage, binding will take place between POC0 and $POC1_{sw}$, which is the soil passage between the landfill and the ditch surrounding the Wieringermeer landfill (seepage situation).
- Dilution of the leachate in groundwater will take place over the total depth of the aquifer (landfill specific). Dilution of the leachate will also take place in the ditch surrounding the Wieringermeer pilot landfill.
- The reactive transport model requires specific soil data of the first 2 metres under the landfill. This information is not present in the monitoring reports of the landfill sites. The required soil properties are therefore selected from nearby soil profiles, as listed in a large Dutch database (STONE database). For each landfill a nearby plot (within 2 km) was selected to obtain the required data.

Sensitivity analysis

A sensitivity analysis was carried out in order to study the influence of several important parameters/assumptions on the magnitude of the ETVs. The selected parameters/assumptions were derived from discussions within the project team and from ‘points of special attention’ highlighted in the recommendations of the Technical Committee on Soil Protection (TCB).

Factors that were studied in the sensitivity analysis are: the effect of the pH of the receiving soil, the effect of assuming a decreasing concentration of substances in the leachate from the landfill (instead of the current assumption of a constant concentration) and the effect of varying the time frame (shorter and longer than 500 years). Another aspect studied was the effect that an increased emission of phosphate from the landfill might have on the mobility of the other substances and the magnitude of the corresponding ETVs. Finally, the sensitivity analysis studied the effects of the use of the local thickness of the unsaturated zone of the receiving soil (instead of the generic 1 metre used in accordance with the policy on the re-use of building materials) and of the effects of variations in the background concentrations in groundwater’.

The following conclusions were drawn:
- Highly soluble salts (chloride, sulphate and ammonium) are not sensitive to variations in chemical and physical factors such as time frame, thickness of the unsaturated layer and a reducing vs. constant concentration of substances in the leachate.
- Highly soluble salts are sensitive to variations in background concentrations in groundwater.
- Variation in the time frame (from 500 years to 100 years and from 500 years to 1,000 years) has the greatest influence on the concentrations of all metals at $POC2_{gw}$. This is in accordance with the findings in the derivation of

emission limit values for the re-use of building materials (with time frame = 100 years).

- The most influential chemical factor by far is an increased dissolved organic carbon (DOC) content, followed by pH. Redox (reduced Fe oxide content) and increased concentration of phosphate in the leachate are important for several anions (in particular cyanide in the Braambergen landfill and arsenic in the Wieringermeer and Kragge landfills).
- The most influential physical factors are the thickness of the unsaturated zone layer thickness and the choice of reducing versus constant concentrations in the leachate.

Results

The results of the modelling take the form of curves presenting the time of arrival of substances at $POC2_{gw}$ (infiltration situation) or $POC1_{sw}$ (seepage situation). It is possible that a substance does not arrive at $POC2_{gw}$ or $POC1_{sw}$ because of binding to soil particles. The ETVs at POC0 can be determined from the calculated curves.

If, after period of active treatment, the concentrations in the leachate of the pilot landfills are equal or below to the ETVs, the demonstration project will be deemed to have been successful and a top cover is no longer necessary. There are landfill-specific lists of ETVs for the pilot landfills in this project.

It should be mentioned that the derivation of these ETVs is based on current knowledge and understanding of the pilot landfills. If, after the ten-year period of active treatment, the circumstances at the landfill deviate from the current input (especially the content of DOC), we recommend that new ETVs be derived for these particular circumstances.

Table S1.2 presents the lists that were calculated for each pilot landfill.

*Table S1.2: Calculated ETVs for the Braambergen, Kragge and Wieringermeer pilot landfills. Values with a *, ** or *** require additional explanation (see footnotes).*

Substance	Braambergen pilot landfill	Kragge pilot landfill	Wieringermeer pilot landfill
Inorganic substances (µg/L)			
Arsenic	190	100	190
Cadmium	6.4	3.6	1.3
Chromium	210	140	37
Copper	50	64	19
Mercury	5.8	4.1	1
Lead	60,000*	130	11,000*
Nickel	21	47	21
Zinc	160	120	39
Free cyanides	61	6.8	35
Macroparameters (mg/L)			
Chloride	450	160	2400
N-Kjeldahl/ ammonium	1.8**	1.1**	50
Sulphate	700	200	1400
Phosphate	n.a.	n.a.	***
Organic substances (µg/L)			

Substance	Braambergen pilot landfill	Kragge pilot landfill	Wieringermeer pilot landfill
Mineral oil aliphatic (µg/L)			
EC5-EC6	0.8	0.17	0.17
EC6-EC8	0.37	0.039	0.039
EC8-EC10	0.047	0.005	0.01
EC10-EC12	0.00127	0.00127	0.0025
EC12-EC16	0.00071	0.00071	0.0014
EC16-EC21	-	-	-
Mineral oil aromatic (µg/L)			
EC5-EC7	4.7	1.4	1.2
EC7-EC8	3.9	2.3	0.83
EC8-EC10	2.6	1.5	0.55
EC10-EC12	1.5	0.87	0.32
EC12-EC16	1.3	0.38	0.28
EC16-EC21	0.36	0.21	0.076
EC21-EC35	0.06	0.035	0.0064
Mineral oil sum EC10-EC40	470	270	100
VOX (µg/L)			
Vinylchloride	0.047	0.014	0.01
Dichloromethane	0.047	0.014	0.01
1,1 dichloroethane	4.7	1.4	1
1,2 dichloroethane	14	4.1	3
1,1 dichloroethene	0.047	0.014	0.01
1,2 dichloroethene (cis,trans)	0.047	0.014	0.01
Dichloropropane (1,2)	3.8	1.1	0.8
Dichloropropane (1,3)	3.8	1.1	0.8
Trichloromethane (chloroform)	4.7	1.4	1
1,1,1 trichloroethane	0.047	0.014	0.01
1,1,2 trichloroethane	0.047	0.014	0.01
Trichloroethene (tri)	47	14	10
Tetrachloromethane (tetra)	0.047	0.014	0.01
Tetrachloroethene (per)	0.047	0.014	0.01
PAH (µg/L)			
Naftalene	0.047	0.014	0.01
Phenantrene	0.028	0.016	0.006
Antracene	0.0066	0.0038	0.0014
Fluoranthene	0.056	0.033	0.006
Chrysene	0.056	0.033	0.006
Benzo(a)antracene	0.0019	0.0011	0.0002
Benzo(a)pyrene	0.0094	0.0054	0.001
Benzo(k)-fluoranthene	0.0075	0.0044	0.0008

Substance	Braambergen pilot landfill	Kragge pilot landfill	Wieringermeer pilot landfill
Indeno(1,2,3cd)-pyrene	0.0075	0.0044	0.0008
benzo(ghi)perylene	0.0056	0.0033	0.0006
PAH (sum10)	1.9	1.1	0.2
BTEX (µg/L)			
Benzene	0.94	0.27	0.2
Xylene	0.94	0.27	0.2
Toluene	4.7	1.4	1
Ethylbenzene	4.7	1.4	1
Other (µg/L)			
Phenols	0.94	0.27	0.2

n.a. = not applicable

* *= By policy decision, this value is lowered to 130 µg/l. In Section 7.2.2 of the main text a further explanation on this topic is given.*

** *= If there is reason to expect that the specific pilot landfill will not meet the calculated ETV, it can be argued that a higher emission with a maximum 50 mg/L for ammonium can be allowed, subject to the terms and conditions described in Appendix 1. This is a policy decision that is not taken in this report.*

*** *= For phosphate no reliable ETVs can be calculated (see also Section 4.3.5). From a sensitivity analysis it can be concluded that as long as the concentrations of phosphate in the leachate remain below 150 µg/L, phosphate will probably not reach the surface water. This value should, however, not be interpreted as an ETV of any kind and monitoring of phosphate after the period of active treatment is advised.*

Recommendations

For various substances information on concentrations in groundwater and leachate for one or more of the pilot landfills was scarce. It is therefore recommended that during the period of active treatment of the pilot landfills a representative monitoring of the concentrations in groundwater (upstream of the pilot landfill) and in the leachate of the relevant landfill compartments is undertaken. Representative monitoring means, sufficiently low limits of quantification (LOQs) for the total range of substances described in this report.

Furthermore, it is recommended that a benchmark study is carried out at the pilot landfills, to determine which other substances (other than the ones described in this report) are present. If it turns out that other substances of concern are present in relevant quantities, the derivation of additional ETVs for these substances should be considered.

Ammonium proved to be a critical substance for all three of the pilot landfills because the ETVs for ammonium are relatively low compared to the expected concentrations in the leachate after the period of active treatment. Including the breakdown of ammonium in groundwater under the landfills was opted during the process of deriving ETVs. It was however not possible to do so because of lacking information. To include the breakdown of ammonium under the landfill, it is recommended that during the period of active treatment further research on the breakdown of ammonium be performed at the pilot landfills. Furthermore, it is recommended that further research be performed on how the breakdown of ammonium could be accounted for in the modelling of the ETVs.

The conditions under the landfill are critical for arsenic. In the default modelling, arsenic would not arrive at $POC2_{gw}$ within 500 years. So the ETV was derived from the environmental criterion for soil as an alternative. This resulted in ETVs of 190 µg/L for the Braambergen and Wieringermeer landfills. However, if the conditions become anaerobic (reducing conditions), arsenic will become more mobile and the ETV of 190 µg/L will be insufficient and under-protective. At the moment it is difficult to predict how the conditions under landfills will develop. It is therefore recommended that these conditions are monitored during and after the period of active treatment.

Because of the complexity of the hydrological situation at the Wieringermeer landfill, a hydrological modelling of this situation was performed by order of the landfill operator. This proved to be a very informative exercise, which allowed an even more site-specific approach to the landfill. A modelling of the hydrological situation could be considered for the Kragge and Braambergen landfills as well. This suggestion also applies to the remaining landfills that are selected for active treatment in the future.

To date the measurement of mineral oil in separated aliphatic and aromatic TPH fractions is not a routine job for the analytical laboratories. There is currently discussion about measuring TPH fractions within the framework of contaminated soils. Although the final decision to enforce these ETVs is to be taken by the competent authority after the period of active treatment, it is recommended that during the period of active treatment oil fractions be reported by the laboratories as summed (aliphatic and aromatic) EC10-12, EC12-16, EC16-21 and EC21-35 fractions. This will provide insight into the distribution of the fractions in the landfills, but will not add to the costs of analysis.

No ETVs were calculated for phosphate because validation by measurements indicates that phosphate model predictions are still inadequate. Phosphate is, however, a substance that is frequently measured at landfills in order to comply with the landfill permit. Therefore, the leaching of phosphate from the pilot landfills will require monitoring after the period of active treatment. If the concentration becomes too high and effects on surface water are expected, action should be taken to prevent the leaching of phosphate from the landfill.

In the current model, the groundwater or surface water at POC2 next to the landfill is designated as a receptor that needs protecting. It is, however, possible that a vulnerable receptor (such as a nature conservation area) is present near the landfill at (the to be defined) POC3, requiring special attention. This receptor can be more sensitive than the environmental protection criterion at POC2. It is advised that in the final evaluation of the period of active treatment the possible presence of a vulnerable receptor at POC3 be determined and, if necessary, additional measures to prevent exposure taken. The competent authorities could consider revising the ETV to protect this receptor.

1 Introduction

1.1 The need for sustainable landfill management

The urrent Dutch landfill policy focuses on the completely watertight and airtight sealing of substances in landfills. The aim is to isolate the substances and thus minimize the risk of contamination of soil and groundwater. This method ensures that no rainwater can enter the landfill once the landfill site has been filled and closed to further waste acceptance. Sealing off the landfill changes the composition of the waste and the associated substances. The top cover of the landfill needs to be replaced regularly, at considerable cost, because of its limited life. Moreover, responsibility for the management of the landfill is passed on to future generations.

Since the 1990s, research has been carried out into sustainable landfill management. The aim of sustainable landfill management is to reduce the extent to which the consequences of landfilling are passed on to future generations. This is done via a source-focused approach. This approach focuses on reducing the emission potential of the waste by stimulating biological degradation processes and the immobilization of substances in the landfill. To this end, water is allowed to infiltrate the landfill and the waste is aerated. This procedure is called active treatment. The idea of active treatment is that the emission potential of the landfill is stabilized at a level at which the fitting of a top cover is no longer necessary and minimum aftercare is required.

There is currently no practical experience of sustainable landfill management in The Netherlands. In order to allow sustainable landfill management in the future, Dutch policy relating to landfills needs to be modified. An initial step in this direction is to allow a (demonstration) project to investigate whether the desired end result can be achieved in practice and within an acceptable period by active treatment of pilot landfills. This experiment will be performed at three landfills in The Netherlands (Braambergen in Almere, Kragge II in Bergen op Zoom and Wieringermeer in Middenmeer) and will look into the long-term processes involved.

The Ministry of Infrastructure and Environment (I&M) has indicated that it wants to stimulate the development of innovative techniques and has therefore launched the Introduction of Sustainable Landfill Management project (hereafter IDS project). The provinces involved have also indicated that they intend to support this research.

If the experiment is successful, the possibility of sustainable landfill management will be introduced via a modification to the Landfills and Landfill Bans (Bssa) (after 2023). If the outcome is successful, approximately 20 more landfills will be eligible for the application of sustainable landfill management, as they meet the sustainability requirements set by the ministry. These locations are called PDS locations. PDS stands for the Dutch for Potential Sustainable Landfill Locations.

In addition to sustainable landfill management, other possible solutions for landfills may be investigated (for example, waste mining). However, this report will focus exclusively on sustainable landfill management as a promising solution for future landfill management.

1.1.1 *Assessment framework*

To assess the effectiveness of sustainable landfill management, a framework is needed. This assessment framework can be used to determine whether, after the period of active treatment (and without the presence of the traditional top cover for a landfill), the risks to the soil and groundwater from substances emitted by the landfill are low enough to be acceptable with regard to the objectives of soil and groundwater protection policy. A list of emission testing values (ETVs) is therefore to be drawn up and endorsed by a ministerial decree for each pilot landfill. The ETVs represent the permissible soil and groundwater emissions from the pilot landfills. The permissible soil and groundwater emissions comprise a concentration and volume of each substance coming from the landfill. After the completion of this phase (Phase 2) of the IDS project, a list of landfill-specific ETVs will be available for each pilot landfill. After the completion of the period of active treatment (approximately ten years), these ETVs will provide an assessment framework. If, after the completion of the active treatment period, the emissions from the landfill do not meet the environmental criteria (presented in the form of the ETVs), the competent authority can still make a traditional top cover compulsory. The landfill operators must also demonstrate that the ETVs can be permanently met after the period of active treatment.

1.1.2 *Research question*

The Ministry of I&M has commissioned the National Institute for Public Health and the Environment (RIVM) and the Energy Research Centre of The Netherlands (ECN) to compile a proposal for the derivation of ETVs for each pilot landfill after the period of active treatment. A landfill-specific approach was chosen that makes it possible to take into account specific properties such as the area of the landfill and the local soil properties. This results in three landfill-specific lists of ETVs. The ETVs to be developed should fit in as much as possible with the other preventive policies for soil and groundwater protection, including the Soil Quality Decree (Bbk), the Decree on Landfills and Landfill Bans (Bssa), the Water Framework Directive (WFD), the Groundwater Directive (GWR) and the European Landfill Directive. Furthermore, the ETVs will apply to only the three pilot landfills that have undergone the period of active treatment.

1.2 Project history

In the IDS project several activities are carried out. In this section an overview of the various phases is given. For more specific information about each phase and the activities carried out in these, please refer to the literature on the phase in question.

1.2.1 *Phase 1*

In Phase 1 (2010) the Ministry of I&M asked the RIVM to draw up a report exploring the options for putting together and calculating a list of ETVs relating to the discharge of substances from landfills (Versluijs et al., 2011). Based on the (inter)national legislation and regulations, an indicative calculation was drawn up of possible ETVs. A conceptual model for establishing the ETVs was also drawn up. The report by Versluijs et al. (2011) thus formed the basis for the activities in Phase 2 (present report). For a better understanding of the starting points of this report, a summary of the results of Phase 1 is now given.

In Phase 1, an inventory of the existing relevant frameworks for assessing soil pollution was drawn up. The existing frameworks include the Decree on Landfilling and Soil Protection and the associated implementation regulations and directives (Stbo), the Bssa, the EU Landfill Directive and the Bbk.

During the project the desired assessment framework was established. This consists of:

- environmental criteria that indicate at what soil and groundwater quality there is sufficient protection. These environmental criteria are the fleshing-out of the environmental protection criterion.
- emission testing values that indicate which emissions to soil are deemed to be acceptable in mg/m^2/time, i.e. so that the environmental protection criterion is not exceeded.

Then a computational model was drawn up to convert the existing standards for the leaching of substances from waste (Bssa) and from building materials (Bbk) into a soil load to determine the effect this leaching has on soil and groundwater quality. The source-path-receptor model and the local situation of the landfill were used as bases for this conversion. The standards that were converted relate to inert waste, non-hazardous waste, non-shaped building materials and non-shaped building materials for which isolation, management and control measures are required (IBC building materials) (Versluijs et al. 2011).

The main conclusion drawn from Phase 1 is that in Phase 2 of the project, the receptor (the environmental target to be protected) should be used as the starting point, not the source (the landfill) of the source-path-receptor model. Backward calculation can then establish the permitted emissions from the landfill. From this starting point several choices need to be made. The Phase 1 report provides the initial details of these choices (Versluijs et al. 2011).

In addition, the Phase 1 report recommends that the background concentration (BC) in groundwater plus the maximum permissible addition (MPA)[1] should be chosen as the environmental protection criterion and as a generic starting point for metals and organic substances, unless drinking water standards prompt a choice of a lower addition than MPA. In most cases, the drinking water standard will not be the determining factor for the fleshing-out of the environmental protection criterion for groundwater, as the drinking water standard is often higher than the BC plus MPA. For the macroparameters (chloride, nitrate and the like) further details of the environmental protection criterion should be given because the MPA values are often lacking (Versluijs et al. 2011).

1.2.2 *Phase 2*

In Phase 2 (2011–2014; this report), the method for deriving the ETVs is worked out in more detail and three pilot landfill-specific lists of proposed ETVs are drawn up. The initial method for deriving the ETVs was presented to the Soil Protection Technical Committee (TCB), which was asked for advice on the assumptions and starting points of the proposed method. This resulted in some changes to the initial concept, of which the details are given in Appendix 1.

To summarize: the TCB's advice related to:

- the position of the POC depending on the hydrological situation;

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[1] For a more detailed explanation of the concept MPA please refer to the box in Section 2.3.1.

- protection levels based on the protection of surface water were relevant;
- the inclusion in the sensitivity analysis of increased DOC concentration coming from the landfill;
- not taking into account the absence of dilution over the entire thickness of the saturated zone under the landfill;
- taking into account density flow;
- the derivation of a testing value for organic matter;
- the inclusion in the sensitivity analysis of increased mobility of substances due to the release of iron oxides;
- the inclusion of a criterion to prevent increasing concentrations in the leachate;
- the assumption of increased emissions of ammonium;
- the performance of a benchmark study at the start of the period of active treatment.

A sensitivity analysis was carried out to verify the influence of several assumptions in the method for deriving the ETVs. The assumptions and the results of the analysis are described in detail in this report (see Chapter 6).

After the completion of Phase 2, the active treatment of the landfills will take place. In this phase, actions (i.e. infiltration and aeration) will be carried out by the landfill operators to stabilize parts or sections of the three selected landfills. This phase will last approximately ten years but can be extended if this is deemed to be necessary by the competent authority. Such would be the case if the ETVs are not met in ten years, but a declining trend can still be seen in concentrations in the leachate. After this, the ETVs proposed in this report will be used to assess whether the pilot landfills meet the environmental criteria.

1.3 Reader’s guide to the report

Chapter 2 provides an overview of the national and international legislation and regulations relevant to Phase 2. Chapter 3 discusses the starting points adopted in the compilation of a proposal for the ETVs. It also discusses the environmental protection criteria adopted. Chapter 4 discusses the computational model used. Chapter 5 presents the ETVs derived. Chapter 6 discusses the approach to and the results of a sensitivity analysis. Chapter 7 provides further reflection on the assumptions and the results of the sensitivity analysis. Finally, Chapter 8 presents conclusions and recommendations.

2 Relevant laws and regulations

2.1 General

There are various policy frameworks in the Environmental Management Act, the Soil Protection Act and the Water Act that are important for the IDS project (see Figure 2.1). Based on these policy frameworks, requirements are set with regard to the way in which waste can be landfilled and to what extent this is allowed to impact on soil and groundwater. These frameworks cover:

1. The landfilling of waste
In The Netherlands the Decree on Landfills and Landfill Bans (Ministry of VROM, 1997) applies to the landfilling of waste. This decree has adopted the policy set out in EU Directive 1999/31/EG on the landfilling of waste. In addition, the Decree on Landfilling and Soil Protection and the associated Implementation Directive apply (Ministry of VROM, 1993).

2. Soil
For soil, the Soil Quality Decree (Bbk) is of particular importance, including the section on building materials. This decree describes the policy and requirements for the re-use of (slightly contaminated) soil and the use of building materials in large-scale soil applications.

3. Groundwater
For groundwater, the European Water Framework Directive (WFD) and the daughter directive, the Groundwater Directive (GWR), apply. In The Netherlands the stipulations of the GWR have been adopted in the Water Act and the Water Decree. For the IDS project, article 6 of the GWR is of particular importance. This article describes the measures aimed at preventing or limiting the input of hazardous substances and pollutants into groundwater.

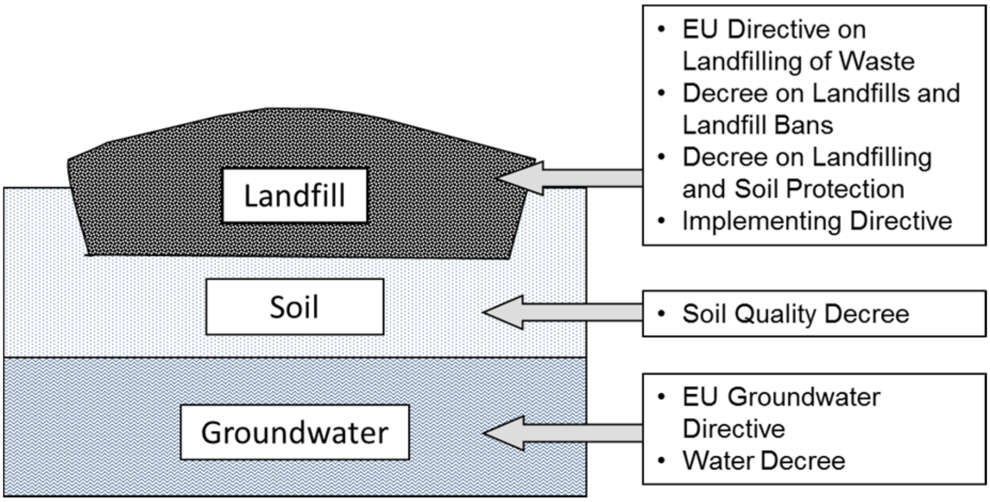


Figure 2.1: Schematic representation of the policy frameworks relevant to the IDS project.

This chapter looks at the relevance of these policy frameworks to the establishment of the ETVs. Similar information can be found in Chapter 2 of the

Phase 1 report (Versluijs et al. 2011); however, in this report the legislation relating to groundwater (the WFD/GWR) has been added.

2.2 Landfills

2.2.1 *European Directive on the Landfilling of Waste*

Directive 1999/31/EG describes European policy on the landfilling of waste. Article 1 states that the aim of the directive is: *to prevent negative environmental effects of waste landfilling, in particular the contamination of surface water, groundwater, soil and air.* The directive sets out regulations regarding: permits, construction, management, checks and the closure of landfills and a reduction in the landfilling of biologically degradable waste. The directive provides an introduction to: landfill classes, hazardous substances, non-hazardous substances, inert waste and associated acceptance conditions for waste by the landfill operaters.

Appendix I of this directive sets out the general regulations for all landfills, including requirements relating to the protection of soil, groundwater and surface water during the operational phase and after closure of the landfill. For example, contaminated water from the landfill should be collected and treated so that it meets acceptable discharge standards. Independent of the type of landfill, the bottom and the walls of the landfill should lined with a mineral layer of a certain thickness and permeability. Fitting a top cover is not compulsory but it can be prescribed if, after assessment of the environmental hazards by the competent authority, the formation of leachate is undesirable. In The Netherlands, landfills additionally have to be fitted with a top cover after 30 years.

Appendix II describes the procedure for determining the acceptability of waste at landfills, the acceptance criteria for each type of waste and the sampling and test methods that have to be used. Which type of waste is allowed to be landfilled at which type of landfill is determined by the leaching requirements (the so called emission limit values and composition values) set for the waste. The emission limit values and composition values that apply at an L/S (liquid/solid) ratio of 10 litres/kg are determined for inert and non-hazardous waste. The leaching of waste is compared with the emission limit values using standard leaching tests in the laboratory by measuring samples of the waste. There are leaching limit values and composition values for: most inorganic substances (metals), two macroparameters (sulphate and chloride), the degree of acidity (pH), the concentration of Dissolved Organic Carbon (DOC), the concentration of Total Organic Carbon (TOC) and Total Dissolved Substances (TDS). No values are included for organic substances. These leaching requierments are based on drinking water quality standards, among other things, and cannot be exceeded at POC (point of compliance) 2 and 3.

This European regulation has been converted in The Netherlands into national regulations in the Decree on Landfills and Landfill Bans and the Decree on Landfilling and Soil Protection. The relevant aspects of these decrees are discussed in the sections below.

2.2.2 *Decree on Landfills and Landfill Bans*

In the EU Landfill Directive and in the Dutch working-out of this (the Bssa), criteria are set for the acceptance of waste at landfills, which are aimed at limiting the risk of contamination of soil, groundwater and surface water. These concern the so-called leaching limits and composition values of the waste for hazardous, non-hazardous and inert waste. These acceptance criteria are included in Appendix II of the EU Landfill Directive (EC, 2003), the Decree relating to the establishment of criteria and procedures for the acceptance of waste at landfills. The leaching limits are tested against the measurement results of standard leaching tests carried out on waste in a laboratory and are based on criteria from the Water Framework Directive (WFD) and the WHO Drinking Water Directives.

Box 1: Explanation on the concept of point of compliance
To make it possible to determine whether an emission into groundwater is acceptable or not, the Guidance on Preventing and Limiting Direct and Indirect Inputs (EC, 2007) of the WFD introduced the concept of points of compliance (POC). POCs are one or more points in the soil/groundwater system that should comply with the specified environmental criteria (compliance values). Compliance values are values that, if not exceeded, ensure that an environmental objective at the receptor is not met. Model calculations or measurements should focus on these values.

The starting points for establishing these criteria are set out in a document entitled 'Development of acceptance criteria for landfilling' dated February 2003 (Miljøstyrelsen, 2003). This document was drawn up for the European Commission by the Danish Hydraulic Institute (DHI) and ECN. This research used POCs (see Box 1) at 20 m and 200 m downstream of the edge of the landfill.

2.2.3 *Decree on Landfilling and Soil Protection*

The Decree on Landfilling and Soil Protection (1993) contains rules for the landfilling of waste in accordance with the so-called IBC criteria: criteria for isolating, managing and controlling waste at landfills. For example, the Decree on Landfilling and Soil Protection prescribes the fitting of drainage pipes, the obligation to catch, collect, remove and purify the leachate and the obligation to sample the groundwater. The Implementation of Regulations for the Decree on Landfilling and Soil Protection (1993) states the parameters on the basis of which the leachate and the groundwater need to be monitored as well as the way in which exceedances of the testing value should be determined and the measures that should be taken to protect the environment against undesired effects. The test value for a substance is calculated by multiplying the so-called signal value of the relevant substance, measured at the reference measurement point, by 0.3 times the target value of this substance as stated in the Circular on Soil Remediation 2009 (VROM, 2009). The Implementation decree defines the signal value of a substance as follows:

- if fewer than 30 measurements are available at a measurement point: the signal value is equal to the arithmetical mean of the background values for groundwater measured at a reference measurement point, multiplied by 1.3;
- if more than 30 measurements are available at a measurement point: the signal value is equal to the value under which 98% of the observations lie (also called the 98th percentile or P98).

2.3 Soil

2.3.1 *Soil Quality Decree*

The Soil Quality Decree and the associated Soil Quality Regulation (Rbk) describe the policy framework and quality requirements for the re-use of building materials, soil and dredging material on or in the soil or in the surface water. For the re-use of building materials, so-called emission limit values have been derived that, like the ETVs for sustainable landfill management, should ensure that soil and groundwater standards are met. The emission limit values themselves are not relevant to sustainable landfill management (due to a different goal of the framework) but the way in which these values are derived is relevant. The sections below describe the starting points and the method for the derivation of emission limit values for building materials and large-scale soil applications. They also describe the method worked out in the Guidance for the Redevelopment of Deep Freshwater Pools. The redevelopment of deep freshwater pools is a specific form of large-scale soil application.

Re-use of building materials

For building materials, emission limit values are included in the Rbk for inorganic substances (metals) and macroparameters. No emission limit values have been drawn up for organic substances, as there are no suitable leaching tests available for these substances. Instead, limits are set for the composition of building materials regarding the amount of organic substances. A generic environmental protection target is adopted for building materials and a generic policy framework is chosen, as building materials can in principle be used all over The Netherlands. It is therefore not possible to apply site-specific factors such as dilution in groundwater.

The emission limit values for inorganic substances are linked to specific leaching tests. As the leaching behaviour of inorganic substances – and thus the risks from various types of building material – can differ greatly, a distinction is drawn in the Rbk between three categories of building materials:

- shaped building materials such as bricks, concrete paving blocks and asphaltic concrete;
- non-shaped building materials such as ashes and granulates;
- IBC building materials, i.e. non-shaped building materials that can be used only when isolation, management and control (IBC) measures are taken, in order to limit emissions.

For the Rbk, generic requirements have been drawn up for the use of these building materials that apply to the whole of The Netherlands and are not product-specific. For the derivation of ETVs for sustainable landfill management the derivation of leaching requirements for non-shaped and IBC building materials is particularly relevant, as these correspond closely to waste landfilling.

Method of establishing leaching requirements

IBC building materials and non-shaped building materials can be applied (respectively with or without isolation measures) in thinner or thicker layers on or in the soil. The derivation of the emission limit values for building materials is based on the transport of the substance by rainwater from the building material through the soil to the groundwater. In the generic scenario the soil layer under the building material is 1 m thick and the groundwater level is 1 m under the building material. POC1 is located in the first metre of the receiving groundwater (see Box 1). This was chosen because of the desire to use building materials nationally andby locating POC1 in the first metre of the receiving groundwater there is no need to take dilution and local conditions into account. This fits in with a generic application. Moreover, the calculation assumes a precipitation surplus of 300 mm/year and assumes that the layer of building material is 0.5 metre thick.

For IBC building materials an infiltration of 6 mm/year is assumed. The concentrations in soil and groundwater are calculated for a time frame of 100 years for various soil types and three types of binding capacity: low, medium and high. The emission limit values are established on the basis of the lowest concentration in either soil or groundwater. For groundwater the values are based on the (annual average) peak concentration in the top metre of the groundwater; for soil they are based on the average concentration in the soil after 100 years (Verschoor et al. 2006; Verschoor & Swartjes, 2008).

As a result of the calculation of these emission limit values, the TCB has recommended that the simulation period not be restricted to the first 100 years after the application of the building materials but be extended until the peak concentrations occur in the groundwater (TCB, 2006). Although the influence of a longer simulation time (1000 years) has been charted (Verschoor et al. 2006), the current emission limit values are based on 100 years. Moreover, the TCB has recommended that the thickness of the soil layer be restricted to the first 30 cm and that the maximum concentration, not the average concentration, be used as the basis for the derivation of the emission limit values.

The emission limit values are based on ecological risk limits, namely the maximum permissible risk for ecology (MPR_{eco}) for soil and the MPR_{eco} for groundwater. For metals and other inorganic substances, the added risk approach is used as the basis (Verschoor & Swartjes, 2008) (see Box 2). In addition, the drinking water standard is also taken into account (see Figure 2.1). The following risk limits are used in the establishment of the emission limit values for building materials:

Inorganic substances (metals)
- MPA_{eco} for soil at POC1;
- MPA_{eco} for groundwater or the drinking water standard (top metre of groundwater) at POC1 (Ministry of I&M, 2011b).

Macroparameters
- MPR_{eco} for groundwater or the drinking water standard (top metre of groundwater) at POC1.

Organic substances
- Composition value of the material to be used.

In practice, the MPA_{eco} for groundwater is often equal to or stricter than the drinking water standard. So for metals, the MPA_{eco} is chosen as the environmental criterion in the policy framework concerning the re-use of building materials.

Box 2: Explanation of use of MPR_{eco}, MPA_{eco} and NR_{eco} in soil and groundwater protection

Dutch policy on the protection of soil and groundwater is based on, amongst other things, ecological protection levels: negligible concentration (NR_{eco}), the maximum permissible addition (MPA_{eco}, only for metals), the maximum permissible risk (MPR_{eco}) and the serious risk level (SRC_{eco}).

Inorganic substances (e.g. metals) can occur naturally in the environment. The concentration naturally present in groundwater or soil (background concentration or BC) can have an effect on the ecosystem. The natural effect of this BC on the ecosystem is not taken into account in the risk assessment of contaminated soils. In these cases the so-called 'added risk approach' is used. For metals, the concentration in the soil associated with the selected risk level (MPA_{eco}) is added to the natural background levels in the soil. The MPA_{eco} is equal to the 95% level of protection, also referred to as HC5 (hazardous concentration). At this concentration level, 95% of organisms are protected against negative effects. The MPA_{eco} is determined by means of laboratory toxicity data. For inorganic substances, the MPA_{eco} is added to a natural BC, resulting in the MPR_{eco}. For inorganic substances, the MPR_{eco} is thus equal to the MPA_{eco} + BC.

Organic substances are usually of anthropogenic origin. PAHs are an exception in this respect; a further explanation of PAHs is given in Section 3.4.2. In Dutch soil policy, the added risk approach is not applied to anthropogenic substances. Therefore, the 95% level of protection based on the laboratory toxicity data is equal to the MPR_{eco}. It is possible that different substances have the same mode of action on receptors, enhancing the total negative effects on the ecosystem. This is called combination toxicology. In assessing combination toxicology the effects of the individual substances are summed to determine the overall risk. Due to the anthropogenic origin of organic substances and the occurrence of many compounds at the same location, the use of the MPR_{eco} as a standard for soil and groundwater is considered too flexible. Furthermore, the use of an MPR_{eco} for organic substances is considered to be too high to be consistent with the 'prevent and limit' principle of the Groundwater Directive (GWD) (see Box 3 in Section 2.4.1). For organic substances, therefore, the NR_{eco} is used as the standard.

The NR_{eco} (sometimes referred to as the target value) is obtained by dividing the 95% protection level by 100 (MPA_{eco}/100 and MPR_{eco}/100 for metals and organic substances, respectively). For inorganic substances, the natural BC is then added to the NR_{eco} (Verbruggen et al. 2001; Verschoor & Swartjes, 2008; Ministry of VROM, 2008). The use of the NR_{eco} as a standard for organic substances is considered conservative and fits with the 'prevent and limit' principle of the GWD.

For macroparameters (chloride, sulphate and nitrogen), the added risk approach can be used, as for inorganic substances, because macroparameters are also naturally present. Yet the standards for macroparameters are based on the MPR_{eco}.

Due to the physical-chemical properties of, for example, chloride, its solubility is so high that the BC should be considered fully available to the receptor (Verbruggen et al. 2008b). Therefore, it is assumed that for chloride both the added concentration and the BC are completely bioavailable and can cause negative effects on the ecosystem. For chloride, generally the same overall approach (MPR_{eco} without a BC) is applied as for substances of anthropogenic origin. This makes the assessment of chloride stricter than for inorganic substances.

For sulphate and nitrogen, an MPA_{eco} is generally used, but the scientific underpinning of MPA_{eco} could not be traced. It is, however, known that the MPA_{eco} for these compounds is mainly based on secondary effects on aquatic ecosystems (such as eutrophication) and not on the direct toxicity of the substances (Brand et al. 2008).

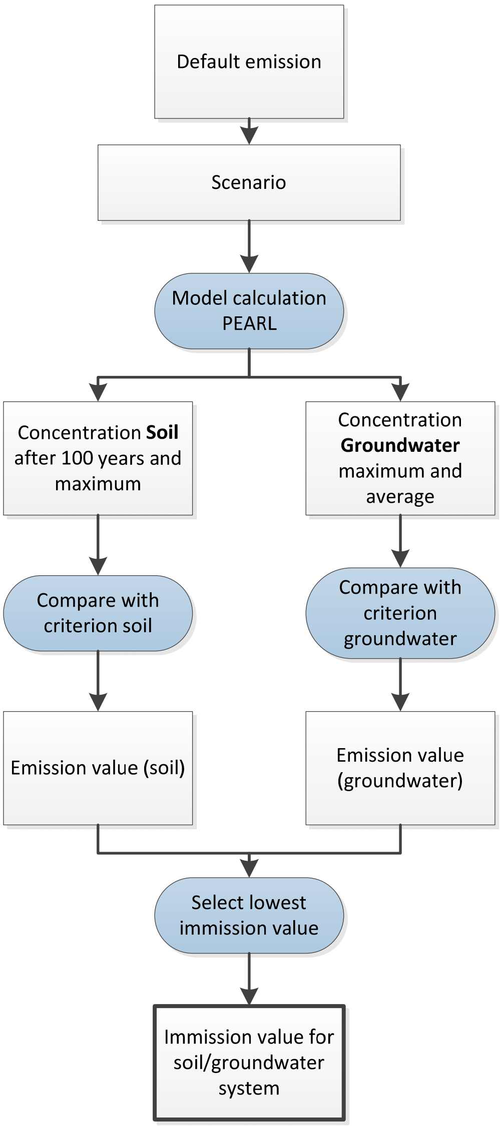


Figure 2.1: Flow chart of the derivation of emission limit values for the re-use of building materials (Verschoor et al. 2006).

Large-scale soil applications

Emission limit values that are specific to the application of soil and dredged material in large-scale soil applications such as dikes and noise barriers are set for metals (Ministry of VROM, 2008). Large-scale applications are those with a volume of at least 5000 m^3 and a height of at least 2 m.

The derivation calculation of these emission limit values is comparable to that for building materials: an open application (without isolation materials) and a net infiltration of 300 mm/year but with an application height of 5 metres (Verschoor et al. 2006). The emission limit value is such that the average concentration of a contaminated substance in the first metre of the soil and in the first metre of the groundwater over a period of 100 years does not exceed the MPA_{eco}.

Re-use of soil and sediment in deep freshwater pools

The Circular on the Redevelopment of Deep Freshwater Pools and the associated Guidance for the Redevelopment of Deep Freshwater Pools (Agentschap NL, 2010) provide the competent authority and other involved parties with information about the responsible use of soil and dredging material in the redevelopment of deep freshwater pools. This Circular was drawn up because of the social concerns and questions among people in the surrounding area and within the local authorities that resulted from the redevelopment of several deep freshwater pools at the beginning of 2009. The Verheijen Committee was set up to this end in order to '*reach agreement as far as possible on the support for policy for the redevelopment of deep freshwater pools*' in consultation with the parties involved. As part of this consultation, various elements were brought together by the RIVM, ECN and Deltares for assessing large-scale soil applications in deep freshwater pools (Lijzen et al. 2011). Parts of this assessment system are also relevant to the IDS project.

The use of large amounts of possibly slightly contaminated soil and dredged material in the redevelopment of deep freshwater pools can have an effect on the quality of groundwater and surface water. A method was developed for the assessment of large-scale soil applications that complies with the aims of the European Water Framework Directive and the Groundwater Directive. The underlying report by Lijzen et al. (2011) examines the use of testing criteria for groundwater and surface water, dilution factors and the effect of distance from vulnerable objects such as drinking water extraction wells and aquatic and terrestrial ecosystems.

In summary, there is a generic approach and a site-specific approach for the reuse of soil and sediment in deep freshwater pools.
Within these, various testing criteria are adopted that apply to metals and/or organic substances. The competent authority can choose the appropriate approach and criteria for each application.

Site-specific approach

In its recommendation 'Assessment Framework for the Redevelopment of Deep Freshwater Pools' the TCB notes the following:
'*The TCB believes that, because of the importance of protecting the often clean deep groundwater, the starting point should be that assessment criteria are related to a 'high' protection level in table 4 (page 27). After all, there are often also other sources that influence the groundwater and the joint influence can then be higher than the protection level concerning MPA and MPR' (TCB, 2010).*

The assessment criteria referred to in the quotation include the following environmental criteria for a site-specific application (Lijzen et al. (2011):
At the assessment point:
Inorganic substances (metals)

- $MPA_{eco}/10$ + generic BC in groundwater
  or
- $MPA_{eco}/10$ + local BC in groundwater
  or
- NR_{eco} (= target value for deep groundwater) (= $MPA_{eco}/100$ + generic BC in groundwater).

Organic substances

- $MPR_{eco}/10$
  or
- NR_{eco} (= target value for deep groundwater) (= $MPR_{eco}/100$).

For the time being macroparameters are not taken into account in the policy relating to the redevelopment of deep freshwater pools.

In addition, the competent authority can consider testing against the drinking water standard if, on the basis of the local groundwater flow, a drinking water extraction well can be reached during a period of 100 years.

The assessment point is located at POC2. This point is located where the flow paths around the deep freshwater pool in the groundwater come together. The flowpath around the freshwater pool has to do with the construction of the pools. The deep freshwater pools reach down into the groundwater (saturated zone) and thus directly influences the flow paths in the groundwater. The groundwater flows to a small extent through the pool, but mostly around the pool, as a result of which the flow paths deviate (see Figure 2.2). In practice, this means that POC2 is at a distance that is roughly the same as the width of the deep freshwater pool concerned. POC2 is thus established site-specifically during the redevelopment of deep freshwater pools.

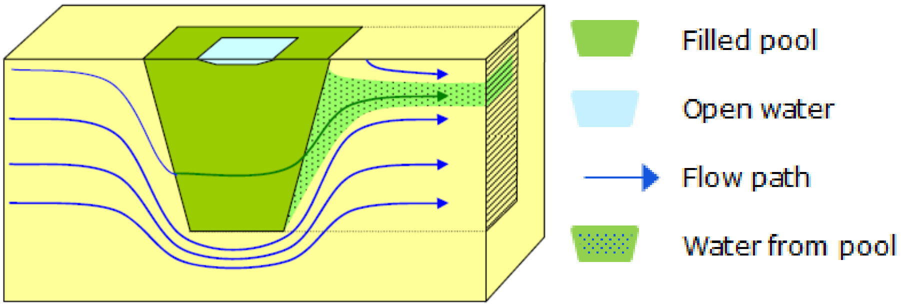


Figure 2.2: Schematic representation of the flow paths surrounding and passing through a semi-permeable deep freshwater pool. The applied filling materials in the pools have a lower permeability than the surrounding soil (the depth is not representative in relation to the horizontal plane) (Lijzen et al. 2011).

Generic approach
In the generic framework the TCB recommends the use of the MPR_{eco} for groundwater in the pore water of the dredged material or soil to be used. This then counts as a composition requirement for the soil or dredged material to be used. Additionally, a generic dilution factor in groundwater of 100 is applied.

This factor is applied as the groundwater usually flows *along* the deep freshwater pool and only to a limited extent *through* the deep freshwater pool (see Figure 2.2).

2.4 Groundwater

2.4.1 Water Framework Directive and Groundwater Directive

The European Water Framework Directive (WFD, 2000/60/EG) has been in force since the end of 2000. This directive aims to ensure that the quality of the surface water and groundwater in Europe is in order in 2015. In order to improve the quality of groundwater and surface water, catchment area management plans were drawn up in 2009 in which the current condition of groundwater and surface water is described as well as the measures that will be taken to improve their quality.

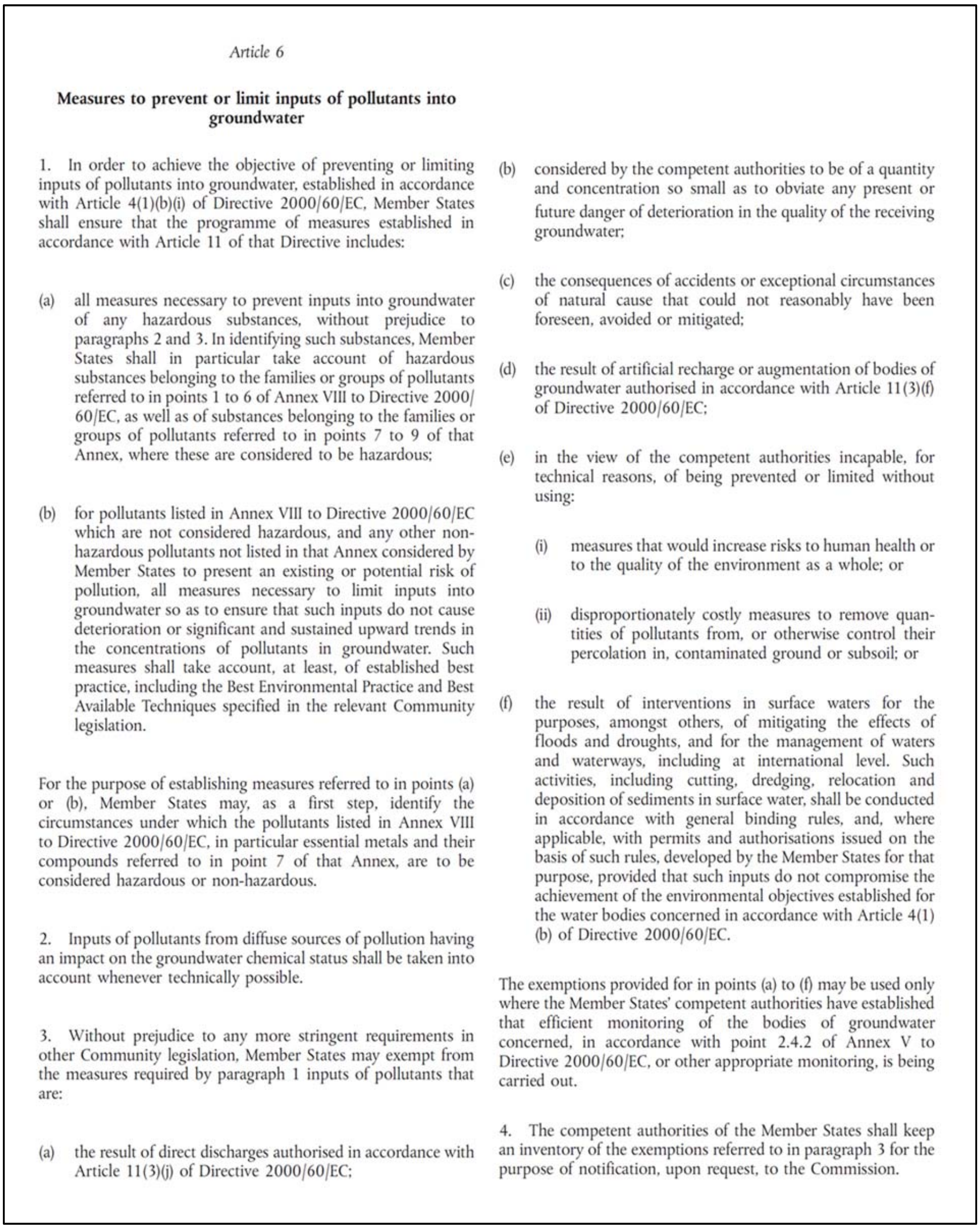
The WFD focuses on surface water and surface-water-dependent ecosystems and is not very specific with regard to groundwater. The aims for groundwater are worked out in further detail in the GWR (GWR, 2006/118/EG). The GWR describes the quality standards that groundwater has to meet and the measures that must be taken to protect groundwater. Article 6 of the GWR relates to the measures that aim to prevent or restrict the input of hazardous substances and pollutants into groundwater (see Box 3).

Article 6
Article 6 states that all (protective) measures should be taken to ensure that hazardous substances cannot get into the groundwater and that all measures should be taken to limit the input of pollutants into the groundwater (article 6.1.b). The measures should take into account the best techniques available. The application of article 6 is explained in further detail in the European Guidance on Preventing or Limiting Direct and Indirect Inputs (EC, 2007).

With regard to hazardous substances the Guidance states: '*Harm is deemed to have occurred when hazardous substances are present in the discharge in amounts that are discernible over and above the naturally occurring background concentrations in the receiving groundwater*'. It then immediately notes that article 6.3 of the GWR includes several exceptions with regard to the input of hazardous substances in certain cases. According to the Guidance (section 3.1), it is not acceptable in the assessment of new emissions (from, for example, a landfill or a sedimentation pond) to take into account the dilution of these substances by the groundwater flow.

For pollutants, the condition of the groundwater must not deteriorate and there should be no significant increasing trend (including on a local scale) (see section 3.4 of the Guidance and Box 4).

Box 3: Article 6 of the GWR with respect to ‘prevent and limit’



Article 6

Measures to prevent or limit inputs of pollutants into groundwater

1. In order to achieve the objective of preventing or limiting inputs of pollutants into groundwater, established in accordance with Article 4(1)(b)(i) of Directive 2000/60/EC, Member States shall ensure that the programme of measures established in accordance with Article 11 of that Directive includes:

(a) all measures necessary to prevent inputs into groundwater of any hazardous substances, without prejudice to paragraphs 2 and 3. In identifying such substances, Member States shall in particular take account of hazardous substances belonging to the families or groups of pollutants referred to in points 1 to 6 of Annex VIII to Directive 2000/60/EC, as well as of substances belonging to the families or groups of pollutants referred to in points 7 to 9 of that Annex, where these are considered to be hazardous;

(b) for pollutants listed in Annex VIII to Directive 2000/60/EC which are not considered hazardous, and any other non-hazardous pollutants not listed in that Annex considered by Member States to present an existing or potential risk of pollution, all measures necessary to limit inputs into groundwater so as to ensure that such inputs do not cause deterioration or significant and sustained upward trends in the concentrations of pollutants in groundwater. Such measures shall take account, at least, of established best practice, including the Best Environmental Practice and Best Available Techniques specified in the relevant Community legislation.

For the purpose of establishing measures referred to in points (a) or (b), Member States may, as a first step, identify the circumstances under which the pollutants listed in Annex VIII to Directive 2000/60/EC, in particular essential metals and their compounds referred to in point 7 of that Annex, are to be considered hazardous or non-hazardous.

2. Inputs of pollutants from diffuse sources of pollution having an impact on the groundwater chemical status shall be taken into account whenever technically possible.

3. Without prejudice to any more stringent requirements in other Community legislation, Member States may exempt from the measures required by paragraph 1 inputs of pollutants that are:

(a) the result of direct discharges authorised in accordance with Article 11(3)(j) of Directive 2000/60/EC;

(b) considered by the competent authorities to be of a quantity and concentration so small as to obviate any present or future danger of deterioration in the quality of the receiving groundwater;

(c) the consequences of accidents or exceptional circumstances of natural cause that could not reasonably have been foreseen, avoided or mitigated;

(d) the result of artificial recharge or augmentation of bodies of groundwater authorised in accordance with Article 11(3)(f) of Directive 2000/60/EC;

(e) in the view of the competent authorities incapable, for technical reasons, of being prevented or limited without using:

   (i) measures that would increase risks to human health or to the quality of the environment as a whole; or

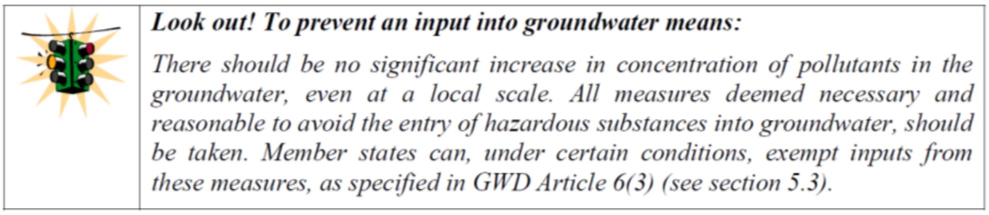
   (ii) disproportionately costly measures to remove quantities of pollutants from, or otherwise control their percolation in, contaminated ground or subsoil; or

(f) the result of interventions in surface waters for the purposes, amongst others, of mitigating the effects of floods and droughts, and for the management of waters and waterways, including at international level. Such activities, including cutting, dredging, relocation and deposition of sediments in surface water, shall be conducted in accordance with general binding rules, and, where applicable, with permits and authorisations issued on the basis of such rules, developed by the Member States for that purpose, provided that such inputs do not compromise the achievement of the environmental objectives established for the water bodies concerned in accordance with Article 4(1)(b) of Directive 2000/60/EC.

The exemptions provided for in points (a) to (f) may be used only where the Member States’ competent authorities have established that efficient monitoring of the bodies of groundwater concerned, in accordance with point 2.4.2 of Annex V to Directive 2000/60/EC, or other appropriate monitoring, is being carried out.

4. The competent authorities of the Member States shall keep an inventory of the exemptions referred to in paragraph 3 for the purpose of notification, upon request, to the Commission.

Box 4: Guidance on prevention or limitation in the GWD



Look out! To prevent an input into groundwater means:

There should be no significant increase in concentration of pollutants in the groundwater, even at a local scale. All measures deemed necessary and reasonable to avoid the entry of hazardous substances into groundwater, should be taken. Member states can, under certain conditions, exempt inputs from these measures, as specified in GWD Article 6(3) (see section 5.3).

Hazardous substances and pollutants

According to the WFD, hazardous substances are toxic, persistent and bioaccumulative substances, and other substances that give cause for concern. Pollutants are substances that are not hazardous but could cause contamination. They comprise generic groups of substances listed in appendix VIII of the GWR (such as halogen compounds, phosphorus compounds, heavy metals, carcinogenic substances and pesticides). The Guidance, which describes how the various inputs from the GWR should be assessed, does not make any further suggestions about which substances in particular belong to either the group of hazardous substances or the group of pollutants. The member states should themselves establish which substances are regarded as non-hazardous and which substances should be regarded as pollutants in this context. At the moment there is no clear definition of hazardous substances and pollutants in The Netherlands.

Heldring & Van Zwam (2008) give a proposal for division of compounds into hazardous substances and pollutants in four directives: the Activities Decree, the Decree on Discharges and Soil Protection, the Decree on Infiltration and Soil Protection and the Soil Quality Decree. They indicate whether the substances listed in these regulations are hazardous substances or pollutants and thus whether their input into groundwater needs to be prevented or limited. In the Heldring & Van Zwam (2008) report, as in the Claessens et al. (2010) and the Lijzen et al. (2011) reports, it is recommended that it be established in an administrative or ministerial directive which substances should be regarded as hazardous substances and which should be regarded as pollutants in The Netherlands.

Exemptions

Section 3 of article 6 of the GWR describes exemptions to the measures outlined in the directive. According to this article, the member states can decide that the prescribed measures are not applicable to the input of substances that are, for example, the result of disasters or if the amount of pollutants is so small that the receiving groundwater does not deteriorate.

According to the GWR, these exemptions should be based on transparent criteria that are described in detail in catchment area management plans. For comparable activities a general description in the catchment area management plan may be sufficient. Exemption can be made if:

- it is not in conflict with stricter European regulations; for example, the input does not have any negative consequences for a Natura 2000 area or for drinking water production (article 6.3);
- the competent authority ensures suitable monitoring that verifies whether the effects of the exemption are acceptable (article 6.3); and
- an inventory of exemptions is kept up to date (article 6.4).

The European Guidance provides a large number of examples of exemptions. One of these exemptions regards landfills. The Guidance states: '*The minimis provision may also apply to residual insignificant inputs from landfills. Landfills have to meet certain requirements aimed at minimising leaching. In the course of time a small flux of pollutants into groundwater may occur, but if the impact is assessed as being insignificant (e.g. by modelling) and validated through monitoring, then the exemption applies.*'

3 Principles and assumptions

The following sections describe the starting points adopted for the derivation of the proposals for ETVs. The computational model used is described in Chapter 4.

3.1 Selection of pilot landfills

The three pilot landfills where active treatment will be tested were selected by the landfill sector before Phase 1 of the IDS project in consultation with the Ministry of I&M.

During the selection, the variability of Dutch landfills was taken into account: for example, the geohydrological situation and the composition of the landfill. Only those landfill compartments were selected where the waste had sufficient potential for stabilization. Landfill compartments and landfills with certain types of hazardous waste (related to emission limit values and composition values for waste) were excluded. Landfills were also excluded where the bottom liner was missing, did not meet the soil protection requirements or had too short a life span (less than 10 year remaining). Finally, the selection took into account the potential for limiting emissions to the air (landfill gases) and the economic feasibility of sustainable landfill management. The result was that landfills or landfill compartments that were not suitable for active treatment were not included in the ministerial decree.

The three pilot landfills that were selected for active treatment are:

- the Braambergen landfill in Almere;
- the Kragge II landfill in Bergen op Zoom;
- the Wieringermeer landfill in Middenmeer.

The chosen landfill compartment at the Braambergen landfill is dominated by inorganic materials such as contaminated soil and soil purification residue. The chosen landfill compartment at the Kragge landfill is dominated by domestic waste. The chosen compartment of the Wieringermeer landfill is dominated by industrial, building and demolition waste. For all the pilot landfills the feasibility of sustainable landfill management was studied in a feasibility study.

The three chosen landfills have different geohydrology. At Braambergen and Kragge there is an infiltration situation. That means that there is leaking of the leachate to the deeper aquifer. The leachate then disperses further in this aquifer. The Wieringermeer pilot landfill is in an area where the surrounding groundwater levels are higher than the level of the adjacent surface water (the ditch around the landfill). Also, the rise height in the underlying aquifer is greater than in the surface layer, as a result of which seepage occurs under and in the area surrounding the landfill.

For a more detailed description of the landfills, refer to the action plans that were drawn up as part of Phase 3 of the IDS project.

3.2 Conceptual landfill model

3.2.1 *In general*

In Phase 1 of the IDS project (Versluijs et al. 2011) a proposal for build-up of a landfill and the surrounding area was defined. The conceptual model adopted complies with the generally accepted source-path-receptor approach. This approach is adopted within other policy frameworks (e.g. the policy framework for the re-use of building materials and the redevelopment of deep freshwater pools) and describes the exposure of a receptor (for example, a drinking water extraction well) via an exposure route (soil and groundwater) from a source (landfill).

In the conceptual model the landfill is considered to be a black box filled with material that leaches substances when water flows through it. Even with the provisions that have to be made in accordance with the current legislation and regulations (for example a top and bottom liner), a certain amount of rainwater flows into the landfill. In accordance with current legislation and regulations, the extent of this infiltration flow is limited by this construction after the landfill has been sealed. Immediately above the bottom liner are leachate drains that collect and remove the leachate. The amount that escapes from the landfill at the bottom is determined by the quality of the top and bottom liners and the removal of leachate by the leachate drains above the bottom liner.

In the future situation (after the period of active treatment) the assumptions made are that the bottom liner has completely failed and that there is no top cover. The protection of the groundwater nevertheless needs to be sufficient.

A generic infiltration flow of 300 mm/year in the landfill is chosen, corresponding to the average net infiltration in The Netherlands and consistent with other policy frameworks such as the re-use of building materials and large soil applications in the Soil Quality Decree. This infiltration of 300 mm/year is assumed to flow out at the bottom across the entire width of the landfill. In reality, the infiltration rate of the leachate into the groundwater may deviate from this assumption, depending on vegetation on the top, the quality of the liner system, and geographical and hydrological factors.

3.2.2 *Location of the points of compliance (POCs)*

To establish the environmental protection target a choice has to be made about distances and times used in the modelling. The framework of the European GWD is used for this. The relevant distances are called POCs. These are points that are relevant for the calculation of the model. In the conceptual model adopted, three POCs are used, namely:

- POC0 = in the landfill;
- POC1 = in the saturated zone of the groundwater or in the surface water;
- POC2 = on the path of the leachate (for example, in the groundwater).

The emissions and concentrations at POC0 provide information about the substances present in the landfill but cannot yet be related to the effect of substances on the groundwater. The concentrations at POC1 and POC2 are more directly related to the risk of groundwater contamination. The concentrations found at POC2 are therefore used to assess whether emissions from the landfill are acceptable. In order to specify where the various POCs should be located, a study was carried out into how the relevant policy frameworks deal with this question.

In accordance with the proposal by Versluijs et al. (2011), a decision was taken to make the environmental protection target equal to the protection of the groundwater quality at POC2 (hereafter referred to as $POC2_{gw}$) for landfills where there is a discharge of leachate to the groundwater. In other words, the the groundwater next to the landfill is protected. For landfills where a discharge of leachate to the surface water occurs, POC2 (hereafter referred to as $POC2_{sw}$)

is located in the receiving surface water. Then a decision was made to translate the environmental protection target into an ETV that applies at POC0 using a (backwards) model calculation. The final testing of the leachate concentration against the ETVs will be carried out at POC0 after the period of active treatment.

It should be mentioned that by policy decision, there is no additional environmental protection criterion for the soil directly under the landfill (see section on exceptions below). This is contrary to the derivation of emission limit values for building materials in the Soil Quality Decree (Verschoor et al. 2006). In the Soil Quality Decree for building materials and large soil applications, an environmental protection criterion for soil was taken into account, namely $MPA_{eco\ soil}$. This criterion is the average concentration that is allowed to accumulate in the first metre of the soil underneath the application during 100 years (see Section 2.3).

Infiltration situation – Leaching to groundwater (Braambergen and Kragge pilot landfills)

In an infiltration situation the POC0 is located directly above the failing bottom liner of the landfill (see Figure 3.1). This has two advantages. First, there are leachate drains at this spot that can be used to monitor the concentrations of the leachate. Second, the concentrations are monitored before they can disperse into the groundwater so that action can be taken in good time if necessary.

The POC0 is also a good place to monitor the trend in the leachate concentrations during the active treatment of the landfills, as the concentration at POC0 is entirely determined by the processes in the landfill and the infiltration of water into the landfill.

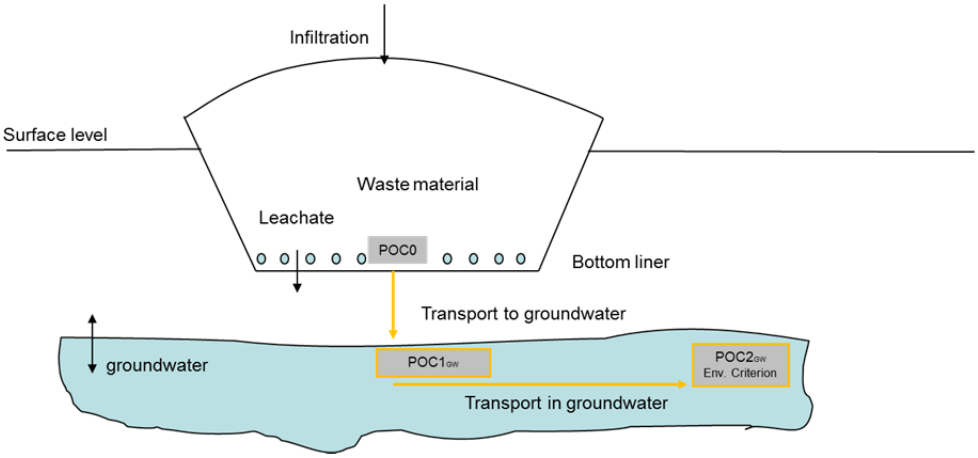


Figure 3.1: Conceptual model of the landfill and its surroundings in an infiltration situation. The yellow arrows indicate the conceptual pathway of the leachate towards groundwater.

At the end of the period of active treatment an assessment will be made on the basis of the concentrations at POC0 as to whether the treatment has been successful.

Under the landfill the soil consists of an unsaturated layer (1 m) with, under that, a layer saturated with groundwater (1 m). The generic thickness of the unsaturated layer under the landfill is chosen to be 1 metre, consistent with the framework for the re-use of building materials in the Soil Quality Decree. In practice, the thickness of the unsaturated zone may vary for the default value of 1 metre (see Chapter 6). The location of POC1 in groundwater (hereafter referred to as $POC1_{gw}$) is located in the first metre of the saturated zone of the groundwater, just as in the building materials framework. The location of $POC1_{gw}$ is included in the testing or monitoring of the environmental protection target if the leachate drains at POC0 stop working. In addition, the location of $POC1_{gw}$ is relevant to the modelling of the ETVs, as this is where the transition point lies between the vertical dispersion from the landfill and the horizontal dispersion in the groundwater.

The location of $POC2_{gw}$ is in agreement with the EU Landfill Directive and the Bssa. For the modelling, the location of $POC2_{gw}$ is established at 20 metres downstream of the edge of the landfill in the groundwater. This choice assumes that a minimum part of the groundwater next to the landfill is influenced by the leachate coming from the landfill and it offers the possibility of taking into account site-specific parameters such as dilution.

So $POC2_{gw}$ is not in keeping with the conceptual model of the policy concerning deep freshwater pools. A major difference from the conceptual model for deep freshwater pools is that landfills are located *above* the aquifer and thus do not influence the flow of the groundwater, as deep freshwater pools do.

Seepage situation – leaching towards surface water (Wieringermeer pilot landfill)
The Wieringermeer landfill is located in a seepage situation. In addition to leaching to the groundwater it is possible that leachate disperses directly into surface water (see Figure 3.2). In this case the leachate does not reach the groundwater and there is only shallow run-off via a passage through soil (20 metres) in the unsaturated zone between the landfill and the surface water. The $POC1_{sw}$ for surface water is located just before the point where the leachate enters the surface water (green line in Figure 3.2).

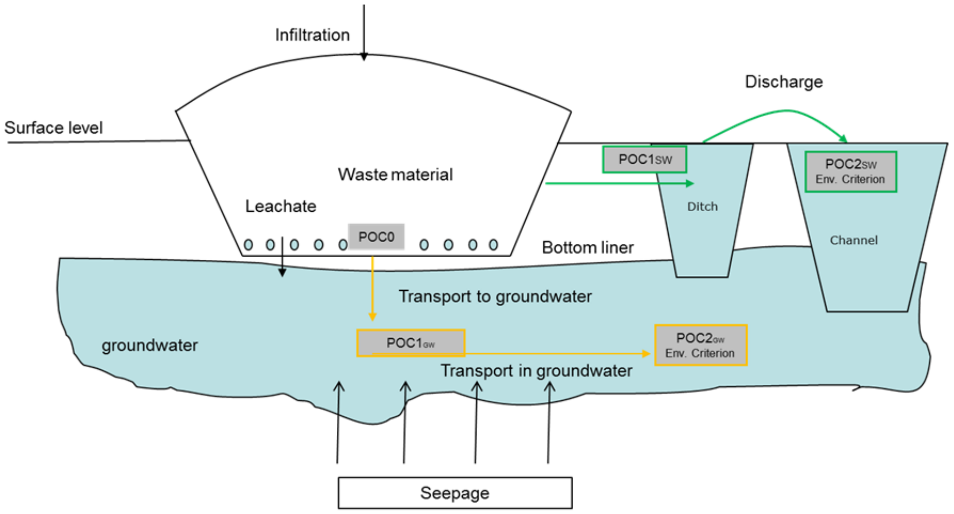


Figure 3.2: Conceptual model of the landfill and its surroundings in a seepage situation. The yellow arrows indicate the conceptual pathway of the leachate towards the groundwater. The green arrows indicate the conceptual pathway of the leachate towards surface water.

Depending on the hydrological situation at a site, different surface waters are classified as the receptor. At the Wieringermeer pilot landfill, the water board, HHNK (Hoogheemraadschap Hollands Noorderkwartier), classified a nearby channel (called Westfriesche Vaart) as the receptor and not the adjacent ditch (see Figure 3.2). The ditch is considered to be part of the construction of the landfill. Water from the ditch is discharged into the channel if the water level in the ditch rises above a certain level. Therefore, $POC2_{sw}$ is located in the channel. This is in line with the WFD and this conceptual model was therefore adopted for the Wieringermeer pilot landfill.

Research by Van Someren (2013) stated that, in addition to leaching to surface water, leaching to groundwater can be relevant on a local scale. So for the Wieringermeer pilot landfill it was decided that both pathways, groundwater and surface water, should be assessed to derive ETVs. The most stringent ETV of both scenarios was chosen as the proposed ETV. Hereafter the two pathways are referred to as the groundwater scenario and the surface water scenario at the Wieringermeer landfill. For a more detailed description of the Wieringermeer pilot landfill, see Section 4.6 and Appendix 7.

Exceptions – leaching into soil

In the conceptual models for an infiltration situation and a seepage situation, a passage through soil (20 metres) is included, either under the landfill (in the saturated zone in the groundwater) or next to the landfill (in the unsaturated zone towards the surface water). During their passage through soil, substances can bind to soil. For some substances, this process is so strong that they never reach the groundwater/surface water at the designated POC. It was therefore not possible to derive a feasible ETV for these substances based on protection targets for groundwater.

In these exceptional cases it was investigated whether the environmental criterion based on soil and the average concentration in the total soil volume under or next to the landfill (in total 20 m^3 of soil) could present an ETV (see Figure 3.3).

This criterion determines the maximum amount of a substance that is allowed to enter the soil within the chosen time frame. This amount is then translated into a concentration in the leachate (µg/L). This principle is also applied in the policy for the re-use of building materials, with the difference that for building materials the most stringent criterion for both soil and groundwater is key to the emission limit value.

In general, the locations of POC0, POC1 and POC2, as described in the previous sections, were selected for deriving the ETVs. How the monitoring of the leachate concentrations will be performed in practice is not part of this report but is described in a guide on the use of the ETVs. Choices made in the final monitoring after the treatment period are of major importance to the end result of the project.

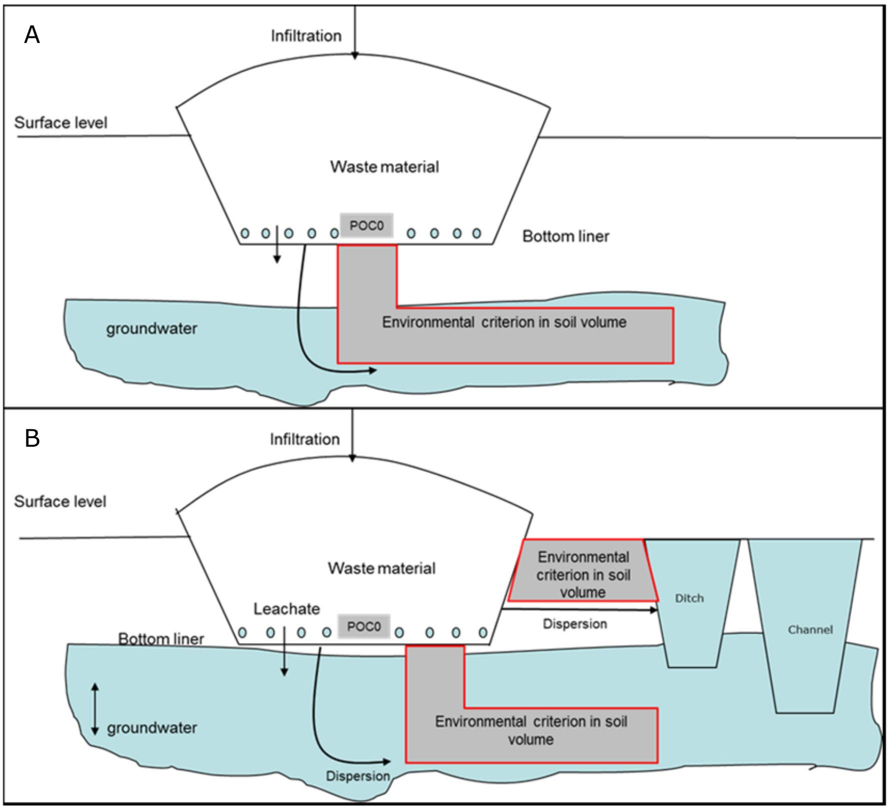


Figure 3.3: If substances do not reach the designated POC2 in surface water or groundwater, the environmental criterion for soil might be used over the total soil volume of 20 m^3 next to or under the landfill. Figure A shows an infiltration situation; Figure B shows a seepage situation.

3.3 Selection of relevant substances

3.3.1 In general

A selection was made of relevant substances for the project. Given the landfill-specific approach within the IDS project, three lists of ETVs were drawn up (one for each pilot landfill).

In the selection of relevant substances the generic list of substances in the Bssa was used as a starting point. The Bssa is the most important policy for landfills and contains a list of substances that landfill operators are obliged to include in their monitoring reports. The substances from the Bssa are so-called trigger substances. If these substances are found in the groundwater under a landfill, investigation is required. The list of substances in the Bssa is based on the substances for which valid A, B and C values existed at the moment the policy was written. The A, B and C values are the precursors of the current target values and intervention values in the soil and groundwater legislation. This list was then supplemented with relevant substances on the basis of expert judgement.

For the IDS project several landfill-specific substances were added to the generic list of substances from the Bssa. These are substances that the landfill operators must monitor in addition to the substances listed in the Bssa. As a result, the list of substances adopted for the ETVs is not the same as, for example, the list of substances used within groundwater policy. However, using the substances from the Bssa as a basis meant that the substances most relevant substances to landfills were selected.

It is recommended that for substances commonly measured in groundwater but which were not selected for this project, to assess whether these are found at the sites of the pilot landfills. This can be done, for example, during the benchmark study before the start of the treatment phase. If this is the case, the establishment of an ETV can still be considered for these substances. During the active treatment phase of approximately ten years additional information can be gathered, such as background concentrations in groundwater.

3.3.2 *Generic Bssa list of substances*

The Bssa includes a generic list of substances that have to be measured in the leachate and the groundwater at each landfill. Table 3.1 provides an overview of the substances that must be monitored. The list of substances from the Bssa forms the basis of the list of substances for which ETVs will be derived. Landfill-specific substances will be added to this generic list of substances.

Table 3.1: Overview of the generic list of substances from the regulatory framework on landfills.

Metals	**Organic substances**	
Arsenic	*VOX*	*PAH*
Cadmium	Vinylchloride	Naphtalene
Chrome	Dichloromethane	Phenantrene
Copper	1,1 dichloroethane	Antracene
Mercury	1,2 dichloroethane	Fluoranthene
Lead	1,1 dichloroethene	Chrysene
Nickel	1,2 dichloroethene (cis,trans)	Benzo(a)antracene
Zinc	Dichloropropane (1,2)	Benzo(a)pyrene
Macroparameters	Dichloropropane (1,3)	Benzo(k)fluoranthene
Chloride	Trichloromethane (chloroform)	Indeno(1,2,3cd)-pyrene
Sulphate	1,1,1 trichloroethane	Benzo(ghi)perylene
N-Kjeldahl/ ammonium[1]	1,1,2 trichloroethane	PAH sum 10
	Trichloroethene (tri)	*Mineral oil*
	Tetrachloromethane (tetra)	Sum EC10-EC40
	Tetrachloroethene (per)	

[1] *N-Kjeldahl/ammonium is the sum concentration of organic nitrogen, ammonia (NH_3) and ammonium (NH_4^+) in the groundwater sample. These are determined by means of a sulphuric acid digestion. N-Kjeldahl/ammonium is expressed in the total amount of ammonium ions generated during the acid digestion.*

Mineral oil

The Bssa prescribes that the total EC10-EC40 fraction has to be measured for mineral oil. There is a discussion within the framework for soil protection about the use of the so-called fraction approach for mineral oil (TPH method). With the fraction approach, every partial mineral oil fraction is tested against a specific protection value for that particular fraction. Allocating risk limits to partial fractions makes it possible to test oils of different compositions against the standard. For measurements in the field, the measurements have to be related to the risk limits for the partial fractions (measurements/risk limit). The total of these risk indices for the partial fractions must not be greater than 1. This is called the ‘toxic unit’ approach. The RIVM has presented both human (Lijzen et al. 2001) and ecological values (Verbruggen, 2004; Verbruggen et al. 2008a), which can be used for the fraction approach.

Policy-related decisions about how to handle the fraction approach within soil protection still need to be taken. To facilitate these decisions, research into the consequences of applying the TPH approach was carried out (Pinedo et al. 2013). As the IDS project has a minimum duration of approximately ten years, a decision was made to take the fraction approach for mineral oil into account when deriving the ETVs. It has not yet been established whether the pilot landfills will actually be assessed against these values in the long term. The following fractions for mineral oil were added to the list of ETVs in addition to the determination of the total EC10-EC40 fraction:

Mineral oil aliphatic:
- EC5-EC6
- EC6-EC8
- EC8-EC10
- EC10-EC12
- EC12-EC16
- EC16-EC21

Mineral oil aromatic:
- EC5-EC7
- EC7-EC8
- EC8-EC10
- EC10-EC12
- EC12-EC16
- EC16-EC21
- EC21-EC35

Organic matter (DOC, TOC and COD)
A frequent problem with landfills is the release of organic matter and oxygen from the landfill. The presence of DOC (dissolved organic carbon) has an influence on the concentrations and the transport speed of other substances. For example, arsenic and cyanide leach quickly in the presence of DOC. So, in addition to the substances themselves, the DOC content, the TOC (total organic carbon) content and the COD (chemical oxygen demand) are measured. There is currently no standard for assessing DOC, TOC or COD directly. For the time being, it is not possible to derive an ETV for organic matter, although the influence of DOC on the mobility of other substances is being studied in a sensitivity analysis (see Chapter 6).

3.3.3 *Additional (landfill-specific) substances per pilot*

In order to gain an insight into the landfill-specific substances that should be included in the derivation of ETVs, the monitoring reports of the pilot landfills were consulted. These contained all the substances that the landfills have to report by virtue of their permits. The site-specific substances were added to the list of substances.

At the Wieringermeer pilot landfill, the leachate was also analysed for phosphate, bicarbonate and BTEX. At the Braambergen pilot an additional analysis was carried out for phosphate, BTEX and phenols. At the Kragge pilot an additional analysis was carried out for cyanide and BTEX.

In summary, the list of substances for Phase 2 of the IDS project consists of the substances presented in Table 3.2. This list can be modified for the remaining

Landfills which are selected for the possible active treatment in the future if this experiment is successful.

Table 3.2: List of substances that are relevant for Phase 2 of the project, based on the generic list of substances from the regulatory framework for landfills and the landfill-specific additions based on the requirements of the landfill permits.

Metals	**Organic substances**	
Arsenic Cadmium Chrome Copper Mercury Lead Nickel Zinc	*VOX* Vinylchloride Dichloromethane 1,1 dichloroethane 1,2 dichloroethane 1,1 dichloroethene 1,2 dichloroethene (cis,trans) Dichloropropane (1,2) Dichloropropane (1,3) Trichloromethane (chloroform) 1,1,1 trichloroethane 1,1,2 trichloroethane Trichloroethene (tri) Tetrachloromethane (tetra) Tetrachloroethene (per)	*Mineral oil* Sum EC10-EC40 Aliphatic EC5-EC6 Aliphatic EC6-EC8 Aliphatic EC8-EC10 Aliphatic EC10-EC12 Aliphatic EC12-EC16 Aliphatic EC16-EC21 Aromatic EC5-EC7 Aromatic EC7-EC8 Aromatic EC8-EC10 Aromatic EC10-EC12 Aromatic EC12-EC16 Aromatic EC16-EC21 Aromatic EC21-EC35
Macroparameters		**Site-specific additions**
Chloride Sulphate N-Kjeldahl/ammonium	*PAH* Sum PAH10 Naphtalene Phenantrene Antracene Fluoranthene Chrysene Benzo(a)antracene Benzo(a)pyrene Benzo(k)fluoranthene Indeno(1,2,3cd)-pyrene Benzo(ghi)perylene	Phosphate Cyanide Fenols *BTEX* Benzene Xylene Toluene Ethylbenzene

3.4 Environmental protection target

3.4.1 General goals

The environmental protection target for the Braambergen and Kragge pilot landfills is the protection of the groundwater under and next to the landfill. Because of the specific situation around the Wieringermeer pilot landfill (seepage), both the protection of the surface water next to the landfill and the protection of the groundwater under and next to the landfill were defined as environmental protection targets. The strictest environmental protection target (either groundwater or surface water) is key for Wieringermeer.

To derive the ETVs at POC0, environmental criteria in the groundwater/ surface water at POC2 are necessary for the relevant substances. Chapter 2 describes the environmental criteria used in other policy frameworks for groundwater. Appendix 2 gives an overview of the various environmental criteria in groundwater and surface water that are valid for the substances that are relevant in the context of sustainable landfill management.

In order to reach an unambiguous environmental policy, the environmental protection target requires that (a) the quality of the groundwater has to be adequate for the preparation of drinking water and (b) the ecological risks have to be acceptable.

To establish the environmental criteria for landfills, the following starting points are adopted:

1. The possible effects on ecosystems and people (drinking water) are taken into consideration. The strictest criterion (either human or ecologogy) of these is used in establishing the environmental criteria.

In addition, the following applies for the macroparameters:

2. If the local background concentration in groundwater or surface water is higher than the environmental criterion (ecology or drinking water), the leachate can in principle contain a concentration which is equally high as the local background concentration.

In addition, the following applies for the inorganic substances:

3. If for inorganic substances the environmental criterion for drinking water is the strictest but the local background concentration in groundwater or surface water is higher than the environmental criterion for drinking water, the leachate can in principle contain a concentration equally high as the local background concentration.

Note to 1: As groundwater is a source of drinking water, the drinking water standard is taken into consideration for landfills where the groundwater is in principle suitable for drinking water preparation. This applies to locations where the chloride and sulphate concentrations are lower than 150 mg/L, as stated in the Decree on Drinking Water (Ministry of I&M, 2011b). For drinking water, only the standards included in the Decree on Drinking Water are taken into consideration. These standards are based on a precautionary principle and therefore do not include any risk basis (Ministry of I&M, 2011b). There is a drinking water standard available for only a limited number of substances. The substances for which a drinking water standard is available are metals, macroparameters, vinyl chloride, 1,2 dichloroethylene, trichloroethylene, tetrachloroethylene, total PAH, benzene and cyanide. For the substances for which there is no drinking water standard available, the possible effect on the ecosystem is adopted.

Note to 2 and 3: In some places in The Netherlands the natural background concentrations in groundwater exceed the standards for groundwater. For these areas the derivation of the ETVs takes into account this background concentration. This avoids a situation in which low ETVs are derived for the leachate on the basis of standards for ecology or drinking water. At $POC2_{gw}$, if the local background concentration is greater than the standard for ecology or drinking water, the local background concentration becomes the environmental criterion (standstill). For chloride and ammonium, an additional approach is necessary. This will be explained in Section 3.4.3.

3.4.2 *Environmental criteria*

Environmental criterion for groundwater

The following environmental criteria were adopted for the protection of the groundwater at POC2$_{gw}$ (see Figure 3.3). These starting points apply to the infiltration scenario at the Braambergen, Kragge and Wieringermeer pilot landfills.

- For inorganic substances the environmental criterion is: MPA$_{eco}$ + (local) background concentration (= (local) MPR$_{eco}$). Only if the landfill is in an area with fresh groundwater are inorganic substances also tested against the drinking water standards. The stricter of these two is selected (this complies with the policy frameworks relating to building materials and the redevelopment of deep freshwater pools and the European Landfill Directive). Brackish groundwater is not tested against the drinking water standard, as this water is not used for the preparation of drinking water. If the local background concentration is higher than the drinking water standard, the environmental criterion in the groundwater at POC2$_{gw}$ is made equal to the local background concentration (a concentration equal to the local background concentration is then allowed to leach from the landfill).

- For macroparameters the environmental criterion is: MPR$_{eco}$. If the drinking water standard is lower than the MPR$_{eco}$, the drinking water standard is chosen. If the local background concentration is higher than the MPR$_{eco}$ and the drinking water standard, the environmental criterion in the groundwater at POC2$_{gw}$ becomes equal to the local background concentration (a concentration equal to the local background concentration is then allowed to leach from the landfill). Chloride is an exception to this; for chloride the environmental criterion in the groundwater at POC2$_{gw}$ is equal to the MPR$_{eco}$ + local background concentration (see Section 3.4.3 for an explanation).

- For organic substances the environmental criterion is: NR$_{eco}$. If the drinking water standard is lower than the NR$_{eco}$, the drinking water standard is chosen. The principle of substances that occur naturally (metals) and therefore have a natural background concentration in groundwater or substances that do not occur naturally (organic substances) justifies this choice. In addition, there is an increased risk on negative effects if the MPR$_{eco}$ is used as a criterionbecause of a possible combined effect of individual substances on a receptor (combination toxicology). This choice is also in line with a former TCB recommendation regarding the environmental criterion for organic substances in the redevelopment of deep freshwater pools.

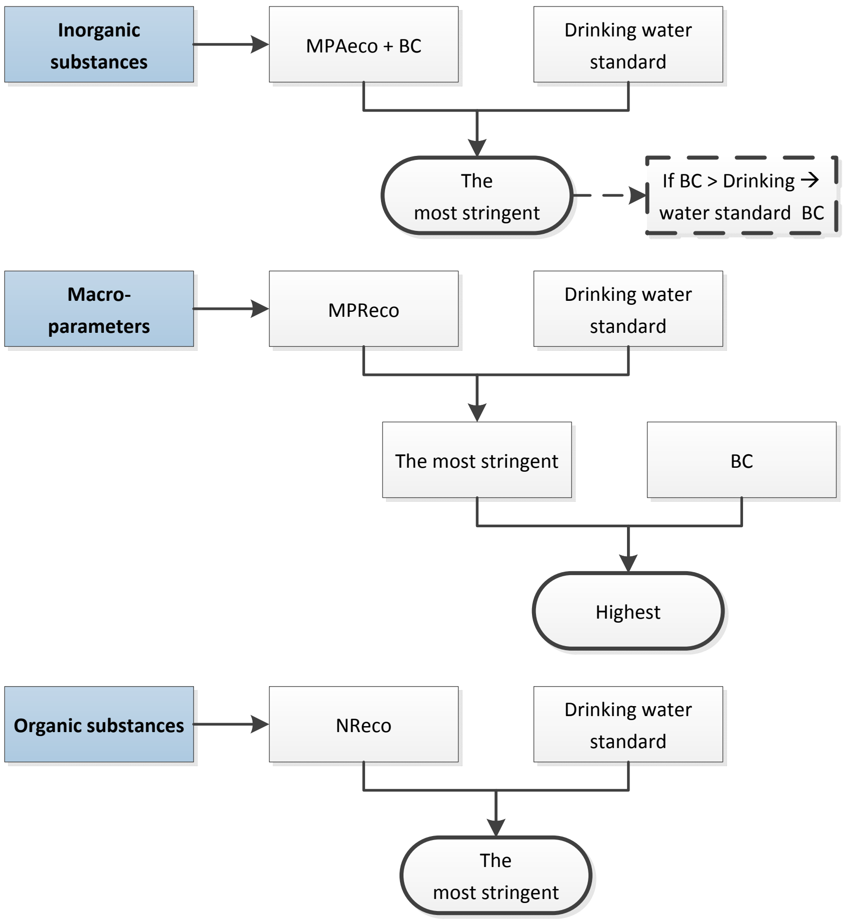


Figure 3.3: Overview of the environmental criterion for inorganic substances, macroparameters and organic substances for the protection of groundwater. Chloride forms an exception to the macroparameters, because for this compound the environmental criterion is always MPR_{eco} + BC in groundwater.

Environmental criterion for surface water

The scenario in which seepage occurs applies only to the Wieringermeer landfill. This scenario is more complex than the infiltration scenario. Within the existing legislation (Ministry of VROM, 2009) the ditch surrounding the landfill is regarded as independent surface water. In addition, the ditch is not a closed system but is freely accessible to fauna and is in direct contact with the surrounding soil and the adjacent channel. This implies that the ditch has to meet the standards for surface water.

At the same time, the ditch has been designated by the competent authority, the HHNK, as being part of the construction of the landfill. Although the assessment point was not established in the permit, the HHNK uses the surface water in the adjacent channel Westfriesche Vaart as an assessment point.

Taking the local situation into account, it was decided that $POC2_{sw}$ should be placed in line with the assessment points adopted by the water board (Ministry of I&M, 2011a; Kleissen, 2012), meaning that the ditch is part of the construction of the landfill and that the effects on the surface water are tested in the channel Westfriesche Vaart.

For the assessment of the effects on the surface water, the concentration of the substances in the surface water is compared with the surface water quality standard, which is called the annual average environmental quality standard (JG-MKN). If there is no JG-MKN the MPR_{eco} for surface water is used as an alternative. For several substances, the HHNK adopts the standstill principle based on the values measured locally in the channel. These values have not been formally established. The definitive values will probably be established in 2014 when the discharge permit for Wieringermeer is reviewed. The WFD's immission/emission test will have to be used for this purpose (see Figure 3.4).

In view of the progress of this project it was decided not to wait for this revision and to adopt the standstill principle for these substances for the time being. The concentrations measured in the channel are, according to the HHNK: chloride 460 mg/L; Phosphor total 0.57 mg/L (= approx. 1.74 mg/L PO4); ammonium 0.65 mg/L.

The WFD immission test was used to establish what concentration is permitted in the ditch according to the environmental criteria in the channel. In addition, the leachate from the landfill in the ditch will be diluted. Appendix 7 gives an overview of the input parameters, starting points and results of the immission test and the dilution factors.

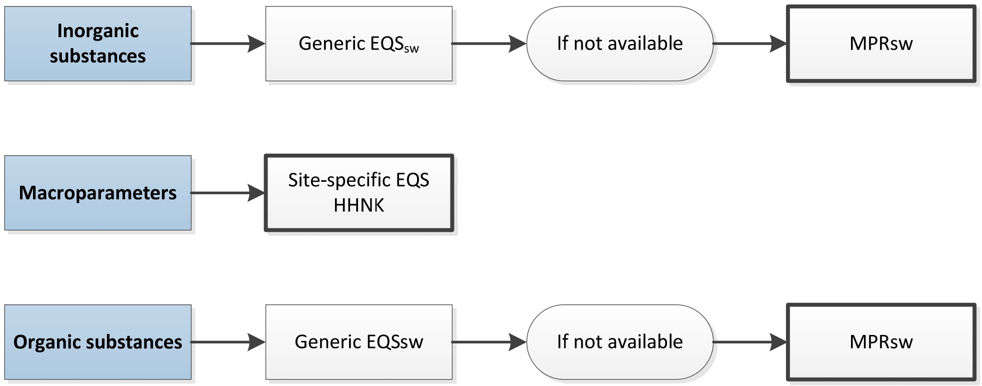


Figure 3.4: Overview of the environmental quality standard (EQS) for inorganic substances, macroparameters and organic substances for the protection of surface water (HHNK stands for Hoogheemraadschap Hollands Noorderkwartier).

Exceptions – environmental criterion for soil

If a substance does not reach the groundwater because of strong binding in the unsaturated zone under the landfill, no unambiguous ETV can be established on the basis of an environmental criterion for groundwater or surface water, and an additional criterion is necessary. Analogous to the building materials situation, it was investigated whether an additional criterion in soil would provide an ETV. The environmental criterion adopted was based on an MPA_{eco} criterion for soil (mg/kg, values from Verschoor et al. 2006). This criterion determines the maximum amount of a substance that is permitted to enter the receiving soil from the landfill within the time frame adopted. The MPA_{eco} value is in this case related to a volume of 20 m^3 soil, i.e. the soil volume across the entire transport distance between POC0 and POC2.

3.4.3 *Special considerations for macroparameters and PAHs*

Chloride
The MPR_{eco} for chloride applies to fresh groundwater. However, the Wieringermeer and Kragge pilot landfills are located in brackish groundwater areas. Locations where in the future active treatment might be applied may also be located in brackish groundwater areas. For these landfills the MPR_{eco} for chloride has no significance, as the emission from the landfill should be lower than the chloride concentration naturally present in the surrounding groundwater.

The derivation of an MPR_{eco} for brackish groundwater is technically not possible because of the lack of data on toxicity for organisms. This is to do with the fact that the organisms living in brackish groundwater have adapted to the conditions. These organisms are either freshwater organisms with a high tolerance of saltwater or saltwater organisms with a high tolerance of freshwater. So it is not possible to establish a protection level for chloride for these organisms on the basis of toxicity studies. It is generally accepted that the exposure of organisms to chloride in brackish groundwater systems may be higher than the background concentrations naturally present, because of the a higher tolerance of the organisms due to adaptation. It cannot be established on the basis of scientific knowledge what this additional exposureload may be.

Within the IDS project various options were considered for assessing chloride. A balanced evaluation was made between protection of the ecosystem, the practicability of the experiment and current environmental policy. Appendix 3 shows the different options that were considered for establishing an environmental criterion.

For chloride a decision was taken to always assess the MPR_{eco} + local BC, even in the case of fresh groundwater. This option contrasts with the generic groundwater policy of an additional chloride concentration above the MPR_{eco} for organisms in fresh groundwater. It should be noted here that the generic (ground)water policy is based on an 'old ' MPR_{eco} for chloride of 200 mg/L. The present report is based on the new, scientifically established MPR_{eco} of 94 mg/L (Verbruggen et al. 2008b). It is assumed that any effects on freshwater organisms are mainly local and minimal.

Nitrogen
This report adopts the current environmental criteria for groundwater and drinking water, as well as for nitrogen in the form of ammonium, as the starting point for the ETVs. For the time being, degradation or binding to soil particles of substances in the aquifer are not taken into account in the derivation of the ETVs. However, hydrological dilution in the saturated zone of the groundwater (between $POC1_{gw}$ and $POC2_{gw}$) is taken into account (see Section 4.4).

During the project the possibilities were explored to include the degradation and/or binding of ammonium to soil in the derivation of ETVs. Appendix 4 provides an overview of this study. As a result of this study the binding of ammonium, but not the degradation of ammonium, could be implemented in the model used for the derivation of the ETVs. However, the binding of ammonium has scarcely any effect on the magnitude of the ETVs, as the substance is mobile in groundwater and therefore reaches $POC2_{gw}$ quickly.

Polycyclic aromatic hydrocarbons (PAHs)
PAHs are substances that have both natural and anthropogenic origins. They can be formed as a result of combustion in industries but also by natural phenomena such as forest fires (Verbruggen, 2012). The background concentrations of PAHs in soil are mostly caused by atmospheric deposition (74%) (Lijzen & Ekelenkamp, 1995). The background concentration can be of PAHs of natural and anthropogenic origins. The national background concentration of PAHs in soil is 1.5 mg/kg (Dirven Van Breemen et al. 2007). Information on PAH concentrations in groundwater is not available because PAHs are not measured in the Dutch National Monitoring Network Groundwater (LMG). Although PAHs can be of natural origin, the contribution of naturally formed PAHs in background concentrations is lower than that of PAHs formed as a result of anthropogenic processes.

Because in the field a difference between naturally formed PAHs and PAHs formed as a result of anthropogenic processes cannot be determined and the contribution of natural causes is limited, all PAHs are considered to be of anthropogenic origin and are assessed as such. For the assessment of the leachate of the three pilot landfills, this means that the environmental criterion for PAHs is equal to the NR_{eco} value at $POC2_{gw}$.

3.4.4 *Time frame*

In addition to the environmental protection target itself, it is necessary to determine a time frame within which the target must be complied to. In the IDS project, a period of 500 years was chosen, primarily because (1) landfills have a long life and are in general not remediated, (2) landfills have a relatively thick waste layer (around 10 metres) that can cause emissions over a long period and (3) with a time frame of 500 years the magnitude of the calculated ETVs is less dependent on the chosen time frame. The use of a time frame of 500 years in this application is also necessary because the valid environmental criterion is 20 metres downstream of the landfill and it therefore takes some time for the substances to reach POC2 (because of binding to soil particles).

This choice of time frame deviates from that chosen for the derivation of the emission limit values for building materials in the Soil Quality Decree, for which a period of 100 years was used (Verschoor et al. 2006). A period of 100 years was also chosen for building materials from a policy point of view, partly because used building materials can be removed and re-used again within 100 years. Also, calculated emission limit values for moderately mobile substances were particularly sensitive to the choice of time frame (Verschoor et al. 2006). The TCB pointed out in a recommendation that the emission limit values should preferably be established for a longer periods of time so that they are less dependent on the time frame chosen. This was done in the assessment of landfills.

3.5 Background concentrations and dilution in groundwater

3.5.1 *Available methods*

In addition to its influence on the environmental protection target, the local background concentration plays a role in the hydrological dilution in the saturated zone of the groundwater. For all the naturally occurring substances (metals and macroparameters) a concentration-dependent dilution is calculated from $POC1_{gw}$ to $POC2_{gw}$. On the basis of the data supplied for each landfill a study is carried out into the extent to which background values can be determined for heavy metals and macroparameters according to the generic approach.

Within current soil and groundwater policy there is no uniform prescribed method for establishing local background concentrations in groundwater. Moreover, there is no unambiguous definition of what exactly a background concentration is. The establishment of background concentrations and the determination of a definition are relevant not only for sustainable landfill management but also within the context of soil and groundwater remediation and the WFD. The Ministry of I&M should consider devoting attention in a wider framework to the determination of local background concentrations in groundwater. It is not possible to pursue this discussion within the scope of this project. So a study was carried out for the IDS project to ascertain how local background concentrations can be determined in compliance with existing methods.

Fraters et al. (2001) define three types of background concentration: natural, semi-natural and regional. For this project a decision was taken to use the natural background concentration. This is the concentration that would be measured at locations that are not affected by human influences. As these locations rarely exist in The Netherlands, only an estimate of natural background concentration can be made. Two methods were considered for this. These are explained below.

Method based on signal values
Within the context of the Bssa, so-called signal values are derived for each landfill in order to make it possible to establish any deterioration of the groundwater quality downstream (signal function) of the landfill. The signal value is established on the basis of measurement points upstream of the landfill, where it may be assumed that the groundwater has not been influenced by the landfill, and can therefore be used as a background value. Depending on the number of available measurements, the signal values are equal to either the 98th percentile (P98) of the data set (with over 30 measurements) or the average of the data set x 1.3 (with fewer than 30 measurements).

Method based on the Water Framework Directive
As an alternative to the use of signal values, the method used to derive national background values as part of the WFD was investigated (EC, 2009; De Nijs et al. 2011). The measurement data required for this were supplied by the landfill managers. These data were collected over the last few years during the monitoring rounds in groundwater filters. A median was calculated of the filters upstream of the landfill and for each substance over the period measured. Then the 90th percentile of these medians was calculated with a 95% confidence interval (hereafter called P90/95) (see Figure 3.5). The 95th or 97th percentile can also be chosen instead of the 90th percentile. In the modelling for this project the 90th percentile was chosen, as the pilot landfills have a limited number of filters. The smaller the number of filters, the larger the uncertainty in the background concentrations established. If there were more measurements available, the P90/95 would ultimately be equal to the P90.

This method deviates from the Bssa, where P98 is used over the entire dataset of the upstream measurement points for deriving a signal value (at >30 measurements).

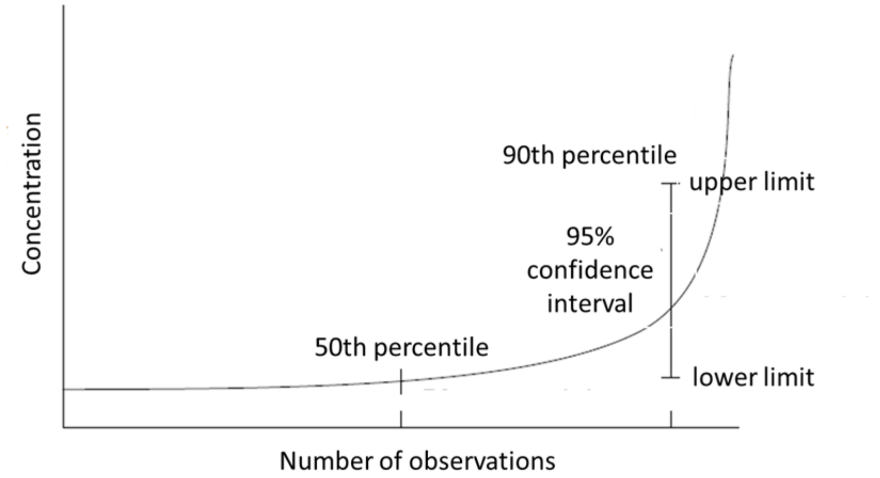


Figure 3.5: Principle of the P90 value with a 95% confidence interval. The greater the number of samples, the smaller the confidence interval will become, resulting in a more realistic background concentration.

3.5.2 *Pros and cons of methods of determining background concentration*

Several marginal notes can be made as to the usefulness of each method to the IDS project. The following marginal notes can be made relating to the derivation of background concentrations based on signal values:

- The method dates from the 1990s and it is unclear how this method arose at that time and why P98 was chosen. It is known that this method is used to indicate the need for (remediation) action. This is a different objective from that intended for the IDS project, namely the protection of the groundwater under and next to the landfill.
- It is unclear why with fewer than 30 measurements the signal value is based on the average x 1.3.
- Using all the measurement data means that the various measurement locations weigh differently in the percentile value depending on the number of filters per location. For example, if there are two filters and there are 10 measurement data at location 1 and 100 measurement data at location 2, then location 2 weighs ten times as heavily in the percentile value. This can be countered by taking the median for each filter. Both locations will then weigh just as heavily in the background value.
- After evaluation of the signal values supplied by the landfill managers for the pilot landfills, it appeared that different starting points were adopted per landfill. As a result it was not possible to establish background concentrations in a uniform manner. Sometimes the signal values were equal to the target value for groundwater and were not in fact based on measured data upstream. In addition, the choice also varies per landfill between the target value for deep groundwater (>10 m) and the target value for shallow groundwater (<10 m). The use of target values has the following restrictions:
  - Target values are not site-specific.
  - Target values are exceeded naturally in lots of places in The Netherlands.
  - The current target values are now outdated (though formally still in force).

A marginal note should also be made relating to the method for establishing national background values within the framework of the WFD, namely that the number of filters that are used for establishing the background value affects the value obtained. Where there are a few filters (<30), the established background value will be closer to the lowest limit of the confidence interval. This can produce an underestimation of the actual background concentration. If more filters are available (>30), the lowest limit of the confidence interval will be nearer to the 90th percentile and the calculated background value will thus better reflect the actual local background value. Table 3.4 illustrates this principle.

Table 3.4: Illustration of the influence of the number of filters on the value for the background concentration of chloride at the Kragge pilot landfill. By the use of randomly selected fictitious filters with concentrations of chloride between 32 and 275 mg/L, the number of filters is increased from 3 (real filters) to 20 (17 fictitious filters and 3 real filters). The average and standard deviation remain equal to the 3 real filters.

Kragge pilot landfill Chloride	**Number of measuring points (filters)**	**Background concentration in groundwater (mg/L)**
Real scenario Based on: – Filters 1: 32 mg/L – Filters 2: 230 mg/L – Filters 3: 270 mg/L	3	32
Fictitious scenario Based on: – Randomly selected fictitious filters with concentrations between 32 and 275 mg/L based on the same average and standard deviation as the 3 real filters	5	135
	7	127
	10	218
	20	230

3.5.3 Conclusion

Taking the above-mentioned marginal notes into account, it decided to use the WFD method. The method that is used in the Bssa has greater uncertainty with regard to the establishment of background concentrations than the WFD method. Moreover, the method based on signal values has a different purpose from that intended in the IDS project, namely signalling remediation actions to be taken.

The WFD method is one of the accepted methods in groundwater policy in The Netherlands for obtaining background concentrations. This method provides a safe, reliable background value. In addition, this method is actually intended for the establishment of local background concentrations.

Finally, in the case of new projects such as the IDS project the preference is to use state-of-the-art methods. The WFD method is an example of these.

3.5.4 Limits of quantification (LOQ)

To establish background concentrations in groundwater, information is used that is supplied by the managers of the pilot landfills. A point of special attention in the use of these datasets is that sometimes all the measurements and/or measurement points reported are smaller than the detection limit (<DL). These concern mainly inorganic substances (metals). There are often sufficient data available for the macroparameters.

In general the term detection limit is often used. Using this term can be confusing as this includes both the limits of quantification (LOQ) and the limits of detection (LOD). This report therefore discusses the LOQ, which are the measurement limits that a laboratory can achieve under certified quality management. The LOD is the lowest demonstrable limit of detection that a laboratory can achieve, but these detections are often not within the quality management and are therefore not certified. As a result, the LOD can vary with each analysis.

If a laboratory reports that the concentration of the substance in question is below the LOQ, this does not always mean that the substance is not present in the sample. In particular, if a high LOQ applies, it may be that the substance is indeed present in the sample.

The LOQs shown in the datasets supplied for pilot landfills are high in relation to what is measurable in practice and in relation to the background values expected in groundwater. In view of the desired progress of the IDS project, it is not possible to carry out additional measurements for the pilot landfills for the background concentrations.

In order to estimate background concentrations for the pilot landfills, a decision was taken to make the reported LOQ value equal to a previously established and accepted LOQ (Table 3.5). These accepted LOQs are lower than the reported LOQs for the pilot landfills. As these are background values, it is not desirable to adopt the high LOQs from the datasets.

First of all, the LOQs from the Aquo Parameter List[2] were assessed. This list was approved by the Groundwater Working Group (consisting, amongst others, of provinces and the national government) and is a formal list that states how groundwater data should be reported within The Netherlands.

However, the LOQs on the Aquo Parameter List are not yet achievable in practice by regular laboratories for all the substances. On the basis of an enquiry at three randomly selected laboratories, a list was drawn up of achievable LOQs (see Table 3.5). The LOQs that were the closest to the value of the Aquo Parameter List were chosen. Where the LOQs from the Aquo Parameter List are not achievable in practice, the values originating from the laboratories were adopted as usable LOQs for deriving local background concentrations (see Table 3.5 and Figure 3.6).

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[2] http://www.aquo.nl/aquo-standaard/aquo/

Table 3.5: LOQs according to the Aquo Parameter List and LOQs based on the information given by three randomly selected laboratories.

Substances	**Aquo Parameter List**	**Achievable in Practice**
Arsenic	0.1 ug/l	1 ug/l
Cadmium	0.01 ug/l	0.1 ug/l
Chrome	0.1 ug/l	1 ug/l
Lead	0.5 ug/l	1 ug/l
Nickel	0.5 ug/l	1 ug/l
Copper	0.1 ug/l	5 ug/l
Mercury	0.01 ug/l	0.1 ug/l
Zinc	1 ug/l	5 ug/l
Chloride	0.1 mg/L	1 mg/L
Sulphate	0.1 mg/L	1 mg/L
N-Kjeldahl	0.02 mg/L	0.3 mg/L

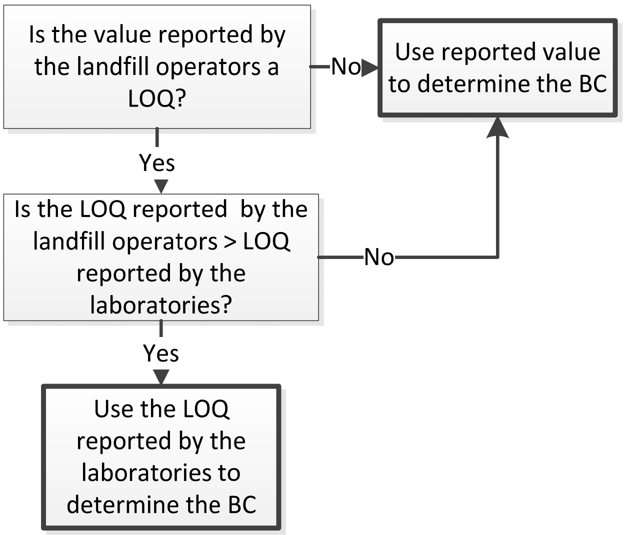

Figure 3.6: If the Limit of Quantification (LOQ) reported by the landfill operators is larger than the LOQ provided by the laboratories, then the LOQ from the laboratories will be chosen to determine the background concentration.

If the background values that were established according to the WFD method (Section 3.5.1), using both the LOQs and actual measurement data (supplied by the landfill operators), result in lower concentrations than the LOQs provided by the laboratories, the LOQs of the laboratories are adopted as background concentrations in the groundwater (see Figure 3.7). If a dataset supplied for the pilot landfills consists completely of LOQs, the background values will be based on the LOQs reported by the laboratories. In the long term it is preferred to adopt the LOQs from the Aquo Parameter List for measuring concentrations in groundwater for determining the local background concentrations in groundwater.

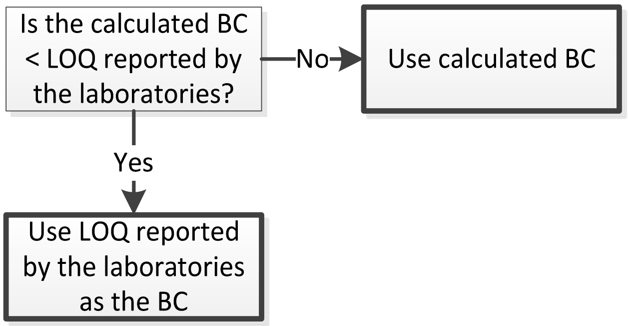

Figure 3.7: If the calculated background concentration in groundwater is below the LOQ provided by the laboratories, then the latter is used as the background concentration in the groundwater.

4 Modelling of transport in soil and groundwater

4.1 Overview and generic model assumptions

Chapter 3 of this report describes the conceptual model and environmental criteria regarding the transport of substances from the landfill into the soil, groundwater and surface water based on the source-path-receptor approach.

It should be noted that the purpose of the modelling is to derive sufficiently protective ETVs, based on a number of generic concepts and assumptions on the one hand and the available site-specific information on the other hand. Hence, the modelling approach should not be seen as an attempt to provide a detailed chemical, physical and hydrological representation of reality.

Figure 4.1 shows the conceptual model used for the dispersion of substances from landfills. The ETVs are calculated for the leachate that infiltrates the soil at POC0 and will ultimately arrive in the groundwater. If the landfill is located in an area where the movement of the groundwater is upwards to the surface (seepage situation), the water from the landfill will also disperse towards the surface water. The ETVs set limits to the concentrations in the landfill leachate at POC0, in such a way that the environmental criteria at $POC2_{gw}$ and $POC2_{sw}$ are not exceeded.

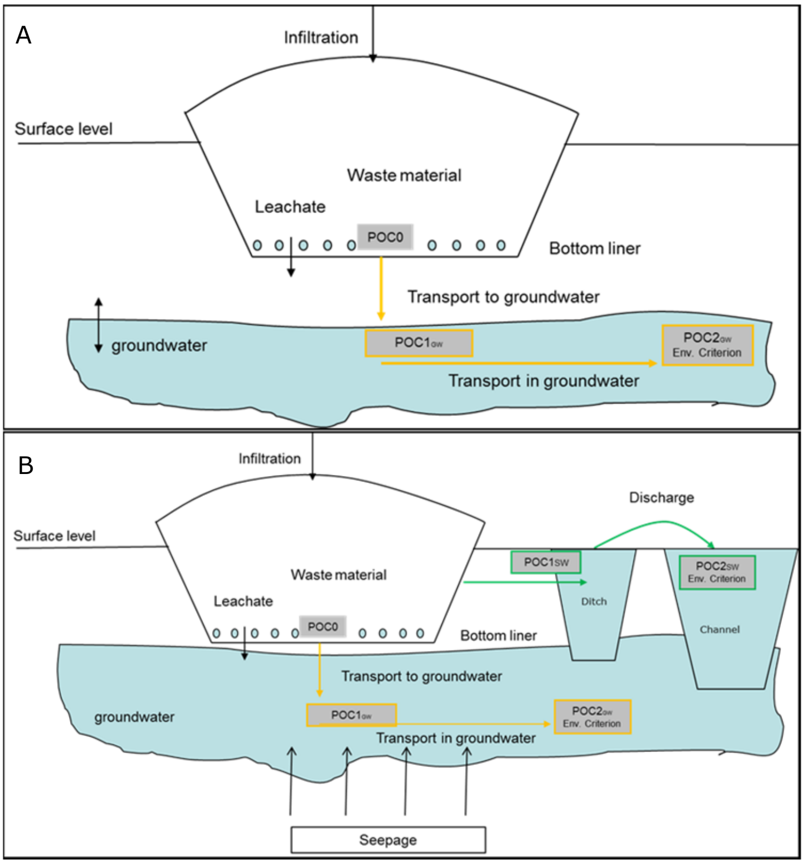

Figure 4.1: Conceptual model of the landfill and its surroundings. Figure A shows an infiltration situation. The yellow arrows indicate the conceptual pathway of

the leachate towards groundwater. The green arrows in Figure B indicate the conceptual pathway of the leachate towards the surface water in a seepage situation. See also Figures 3.1 and 3.2.

The conceptual model as depicted in Figure 4.1 is 'translated' into a model that is used for calculating the ETVs. The approach is based largely on the approach used to derive emission limit values for the re-use of building materials in the Soil Quality Decree (see Section 2.3). The approach consists of three model 'components', which are depicted in Figure 4.2 and will be explained in detail in the sections below:

1. *A description of the 'source term' (POC0) (Section 4.2).* The source term in the model is the assumed contaminated landfill leachate that enters the underlying soil and to which the ETVs apply in reality (given in units of µg/l). The definition of the source term requires assumptions on whether infiltration rate and concentrations are constant or declining over time, and to some extent the landfill-specific composition is taken into account. In this approach, the concentrations in the source term are assumed to be constant over time, and the source term that enters the underlying soil is of a magnitude of 300 mm/year, corresponding to the average net infiltration in The Netherlands.

2. *A description of transport and retardation in the soil (Section 4.3).* In the soil, substance-specific geochemical processes take place, which cause retardation (slowing down) of the transport of substances that originate from the landfill and move through the soil towards the groundwater. In the model, the conservative assumption is made that sorption (binding) and retardation take place only in the unsaturated zone and in the upper metre of the saturated zone directly under the landfill (see Figure 4.2 and Section 4.3). From the upper metre of the saturated zone ($POC1_{gw}$) to $POC2_{gw}$, only dilution is taken into account (see Figure 4.2). The downward reactive transport through the unsaturated and saturated zones is calculated in one dimension using the geochemical and transport model ORCHESTRA (Meeussen, 2003). The model for sorption processes is state-of-the-art and takes into account the geochemical processes that determine the substance-specific sorption and retardation in soils (Dijkstra et al. 2004; 2009). This model was used previously to derive the emission limit values for the re-use of building materials in the Soil Quality Decree (Verschoor et al. 2006). The soil properties of the underlying soil required in this model are site-specific (taken from the STONE database; see Section 4.3.2) and determine the degree of substance-specific sorption at each site.

3. *An estimate of the dilution of the leachate between the water in the upper metre of the saturated zone ($POC1_{gw}$, model output from the previous step) and $POC2_{gw}$ or $POC2_{sw}$ (section 4.4).* This is based on a 3D estimate of the hydrology of the landfill site and is therefore also site-specific.

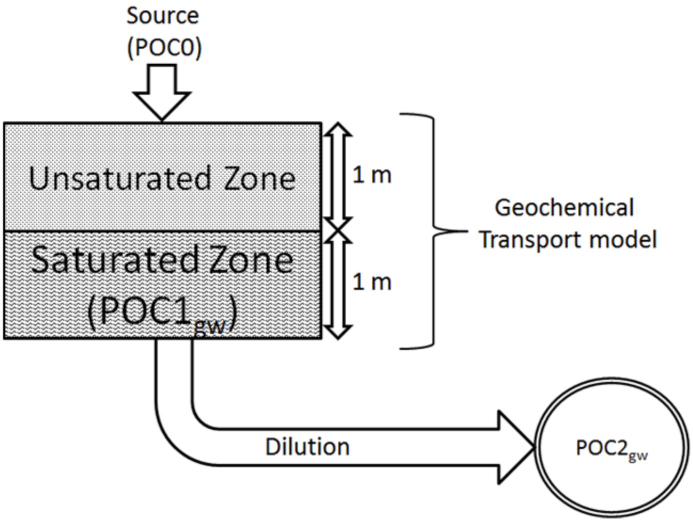

Figure 4.2: Schematic overview of the calculation of the concentration at *$POC2_{gw}$*.

Depending on the substance and the local soil properties, the simulated concentrations in the first metre of the saturated zone ($POC1_{gw}$) will change over time and ultimately become equal to the concentrations in the source term of the model. Substances that show no retardation will arrive at $POC1_{gw}$ quickly. Substances that are strongly retarded may not arrive at $POC1_{gw}$ even within the period of 500 years. Their concentrations are after 500 years lower than the source term of the model. In order to determine the concentration at $POC2_{gw}$, the concentration at $POC1_{gw}$ is diluted according to the site-specific dilution factor, taking background concentrations in groundwater into account (see Figure 4.2).

The ETVs are calculated on the basis of the concentrations that will occur over 500 years in the groundwater at $POC2_{gw}$ or surface water at $POC2_{sw}$, in accordance with the principles outlined in Section 3.4.3. The model simulation time is extended to 1000 years in order to illustrate how concentrations change in the period after 500 years.

In the following sections, the above-mentioned model components are explained in detail separately.

4.2 The source term

The source term in the geochemical model describes the intensity (infiltration) and composition (concentration) of the leachate that infiltrates the soil directly under the landfill (see Figure 4.2). Note that the ETVs are actually derived as concentrations (units of µg/l) in the source term, i.e. the landfill leachate.

For the calculation of the ETVs for the landfills, a constant, infinite source term with respect to both infiltration rate and composition is assumed, based on the following considerations:

1. Infiltration rate: It is assumed that during the entire period of 500 years after the landfill has been actively treated, there is no top cover present and the bottom liner is completely failing (see Section 3.2.1). This means that throughout the entire simulation a constant infiltration rate of 300 mm/year is assumed in the calculations. The infiltration rate of 300

mm/year is chosen because it represents the average net precipitation surplus in The Netherlands and corresponds also to the infiltration rate assumed in the derivation of emission limits for the re-use of building materials and large soil applications in the Soil Quality Decree (Verschoor et al. 2006). It is assumed that the precipitation enters the landfill at the top and leaves the landfill at the bottom, entering the soil directly underneath. In reality, the infiltration rate of the leachate into the groundwater may deviate from this assumption depending on vegetation and geographical and hydrological factors.

2. Composition: Concentrations in the source term, i.e. the landfill leachate entering the soil, are assumed to be constant over time. The application height of the material in the landfill is much higher than in other applications such as for the re-use of building materials. At the assumed infiltration rate of 300 mm/year and a height of up to several tens of metres, the percentage reduction in concentrations of substances in the source term is expected to be very limited. For comparison, in the scenario for the re-use of building materials (with decreasing concentrations in the source term) the application height was only half a metre (Verschoor et al. 2006). The influence of a decreasing source term (instead of the constant source term assumed in this project) was studied quantitatively in the sensitivity analysis in order to determine the possible effect (see Section 6.1).

The source term of the model includes all the selected substances (see Section 3.3) along with several major chemical elements (such as sodium and calcium). The composition of the major elements is important for calculating the sorption of substances in the soil using the geochemical transport model (see Section 4.3), as they influence the geochemical processes that take place in the unsaturated and saturated zones. Sodium and calcium are needed to create a neutral solution to balance the high concentrations of chloride and sulphate, respectively, and influence the sorption of other cationic substances through exchange processes and competition for sorption sites. This is a plausible assumption because monovalent cations generally complete the balance for chloride (as a single charged anion), and calcium and sulphate in waste materials are often related to each other through the gypsum equilibrium (see Dijkstra et al. 2002).

4.3 Reactive transport and retardation in the unsaturated and saturated zones

4.3.1 Overview and general aspects

A reactive transport model is used to describe the transport of substances influenced by sorption and retardation in the unsaturated and saturated zones based on site-specific soil properties (see Figure 4.2). In the model, the assumption is made that sorption and retardation take place only in the unsaturated zone and in the upper metre of the saturated zone directly under the landfill (see Figure 4.2).

The model that is used for this purpose calculates numerically the transport of water through the unsaturated and saturated zones (see Figure 4.2), in combination with the main geochemical (sorption) processes that emitted substances undergo in soil and groundwater. This model is implemented in the model framework ORCHESTRA (Meeussen, 2003) and has been used in the same manner for the derivation of emission limit values for the re-use of building materials and large soil applications in the Soil Quality Decree (Verschoor et al. 2006).

For a detailed description and demonstration of the predictive capabilities of the model for a wide range of substances, both under batch conditions (as a function of pH) and under (laboratory and field) transport conditions, as well as an evaluation of uncertainties and sensitivities, the reader is referred to Dijkstra et al. (2004, 2009, and references therein) and to Comans et al. (2014).

The model will be explained below with reference to the local soil properties from the STONE database (see Section 4.3.2), the transport of water (see Sections 4.3.3 and 4.3.4) and the geochemical approach to the calculation of sorption and retardation (see Section 4.3.5).

4.3.2 *STONE database*

The reactive transport model uses a site-specific approach as much as possible, by including the local properties of the soil and groundwater taken from the landfill monitoring reports. However, additional information is needed, which is not present in the monitoring reports. For this purpose, the STONE database is also used. The STONE database is an internally consistent source of information that contains the remaining soil properties needed in this modelling approach. Originally developed for nutrient balance purposes by Alterra, the STONE database contains the required data about dry bulk density, clay percentages, organic matter content (total), amorphous and crystalline iron and aluminium (hydr)oxides and pH for 6405 plots of 1 m^2 all over The Netherlands, measured for each 10-centimetre layer to a depth of 2 metres. These geochemical properties, which are otherwise usually not known in detail (as was the case at the pilot landfills), are needed in the geochemical part of the model that calculates sorption processes. This is the main reason why this database was used. The STONE database has the additional advantage that it has a very high geographic coverage and hence includes the main representative soils of The Netherlands. For each landfill site, the soil properties are based on the closest STONE plot (the greatest distance is approximately 2 km) (see Appendix 5).

4.3.3 *Transport of water in the unsaturated and saturated zones*

In the model, sorption and retardation take place only in the unsaturated zone and in the upper metre of the saturated zone (POC1$_{gw}$) directly under the landfill (see Figure 4.2). This conservative assumption is made because the relevant information on the geochemical properties of the soil below a depth of 2 metres (the maximum depth of the STONE soil profiles) is not sufficiently available. Between POC1$_{gw}$ and POC2$_{gw}$, only dilution is taken into account, based on a three-dimensional estimate of the local hydrology (see Section 4.4). Hence, the reactive transport model describes only the transport and sorption processes in these two soil compartments of 1 metre thickness, as shown in Figure 4.2.

The unsaturated and saturated zones, as shown in Figure 4.2, are represented in the model by two homogeneous layers, each 1 metre thick. The upper layer is the ‘unsaturated’ layer and the lower layer is the ‘saturated’ layer. Note that the ‘saturated’ layer, i.e. the upper metre of the saturated zone, is also regarded as POC1$_{gw}$. The properties of both these layers are derived from the average chemical and physical properties of the ten layers of 0.1 m thickness recorded in the STONE database.

One-dimensional, saturated, stationary downward transport through these layers is calculated numerically on the basis of an assumed infiltration of 300 mm/year. This implies that for each m^2 of the landfill, 300 litres of landfill leachate per year (300 mm/y) enter 1 m^2 of underlying soil. The 300 mm/y infiltration rate is a fixed parameter in the model. With a saturated porosity (volume fraction or pores) of approximately 0.3 m^3/m^3 (it varies somewhat for the different soil types considered in this study), the resulting net downward velocity (pore water velocity) in the soil is approximately 1 metre per year in this approach. Substances that do not bind to the soil (such as chloride) will have the same downward velocity as water. Substances that bind strongly to the soil will show a velocity that is substantially lower than 1 metre a year.

4.3.4 *Remarks on preferential flow, unsaturated flow and dependence on time frame*

The spatial discretization (i.e. subdivision into one layer of 1 metre) of the unsaturated zone is different from the discretization used for the derivation of emission limit values for the re-use of building materials and large soil applications in the Soil Quality Decree (Verschoor et al. 2006). In the latter work, the unsaturated zone was subdivided into ten layers each of 0.1 metre thickness, as in the STONE database. The reason for deviating from this approach and choosing a homogeneous layer of 1 metre thickness originates from uncertainty and sensitivity analyses that were performed on the advice of the TCB, which are reported in Comans et al. (2014). The TCB pointed out in 2006 that the model used in Verschoor et al. (2006) did not sufficiently reflect the potential effects of preferential flow of the leachate through the soil; the TCB also recommended that the model be developed to enable the derivation of emission limit values that were less sensitive to variations in the time frame (in the Soil Quality Decree, the time frame was set at 100 years; see Chapter 2).

An approach based on a homogeneous layer of 1 metre thickness results in larger numerical dispersion (mixing), which is more conservative with respect to the likely occurrence of preferential flow paths, their change over time, and the mixing that occurs due to bioturbation over long time scales (Comans et al. 2014). Because of the homogeneous layer, part of the substances arrive earlier at POC in groundwater, while still the complete reactivity of the unsaturated zone is taken into account. The coarser spatial discretization has a relatively small effect on concentration levels (see comparison of different modelling approaches in Comans et al. 2014).

In addition, mixing of concentrations in the unsaturated zone makes the results less dependent on the chosen time frame. The time frame was also the subject of a sensitivity analysis in this project, on the advice of the TCB. Instead, when the unsaturated zone is subdivided into ten layers of 0.1 metre, as in the approach to the re-use of building materials (Verschoor et al. 2006), concentration fronts become very sharp, and the derived model results are determined by the exact moment of arrival at $POC2_{gw}$ (see discussions in Verschoor et al. 2006,and in Comans et al. 2014). Due to the layer of 1 metre, the transport of substances is described more disperse (spread out) and results become less dependent to the time frame.

With respect to the upper metre of the saturated zone, it should be noted that concentrations were also mixed in the derivation of emission limit values for building materials using the PEARL and ORCHESTRA models (Verschoor et al. 2006). In Verschoor et al. (2006) concentrations in the lower ten layers of the model were explicitly mixed. The difference between explicit mixing in these layers and the use of a homogeneous layer of 1 m thickness has been shown to be very small (Comans et al. 2014).

Diffusion as a transport mechanism is not taken into account because its contribution to the overall (fixed) transport velocity (i.e. a pore water velocity of approximately 1 metre per year, due to advection of substances) is low. Also, the contribution to the concentration front spreading is negligible considering the already substantial numerical dispersion due to the homogenization of the unsaturated zone.

The fixed downward infiltration of 300 mm/y and the condition of a saturated pore space in the unsaturated zone is a simplification compared with reality. However, in Verschoor et al. (2006) a direct comparison was made for the re-use of building materials scenario in the Soil Quality Decree (similar to the scenario used in this work) between the calculated transport of linear sorbing substances under unsaturated, non-stationary conditions in a sandy soil from the STONE database (with the model PEARL-SWAP) and a scenario using stationary saturated conditions (with ORCHESTRA), using the same fine grid (layers of 0.1 m) in both models and for a period of 100 years. Under those conditions, the differences appeared very small (see Figure 4.6 in Verschoor et al. 2006). A similar comparison was made with ORCHESTRA coupled to SWAP between unsaturated, non-stationary transport and a scenario using stationary saturated conditions, both in combination with non-linear sorption processes (Comans et al. 2014). The differences were again small, illustrating that adding complexity to the transport approach is not deemed necessary. For clay soils, the fixed downward infiltration rate of 300 mm/y may be conservative, as in reality water flows around semi-permeable layers.

4.3.5 *Geochemical processes in the model that determine sorption and retardation*

The reactive transport model calculates sorption of substances to soil based on the main contributing processes for which reliable generic binding parameters are available in the scientific literature (see Dijkstra et al. 2004; 2009; similar approaches have been followed, e.g., by Weng et al. 2001; Schröder et al. 2005; Bonten 2011). This fundamental geochemical approach to the calculation of substance-specific sorption has been shown to be more accurate for the prediction of inorganic substances than simple partition functions such as a linear ‘Kd’. The latter approach is used in many other transport models but has limited predictive capabilities outside the range to which they apply (see discussion in Groenenberg et al. 2012). Hence, geochemical models based on fundamental properties are more generally applicable and are therefore expected to provide more accurate site-specific predictions of sorption and retardation.

The substances are subject to chemical reaction, including solution complexation and sorption to dissolved and particulate organic matter, iron and aluminium (hydr)oxides and clay using dedicated sub-models with generic reactions and parameter sets (Dzombak & Morel, 1990; Kinniburgh et al. 1999). The substances can also be precipitated through the formation of minerals. The required inputs of pH, organic matter, iron oxides, aluminium oxides and clays for each landfill site are taken from the STONE database.

As the substances influence each other in these processes, due to competition for sorption sites, complexation, etc., the concentrations (and ETVs) for all the substances are calculated simultaneously. This process is highly non-linear and several iterations are therefore necessary.

Compounds-specific aspects

The soil and the groundwater in the unsaturated zone and the top metre of the saturated zone of the model are at the start of the modelling phase 'clean' with respect to the substances. However, background concentrations of important main elements (that are important for competition for sorption sites) are imposed in the model.

For iron and aluminium, the dissolved and adsorbed concentrations in the soil are regulated by the presence of a sufficiently large amount of ferrihydrite ($Fe(OH)_3(s)$) and gibbsite ($Al(OH)_3(s)$) (as included in the STONE database, see Appendix 5), which dissolve under the influence of soil parameters such as pH and organic matter.

Calcium

Calcium can precipitate in the soil as calcite if the pH and carbonate content are sufficient. The background concentrations of calcium are:

- At the start of the calculation, the soil solution at the Kragge pilot landfill contains 0.001 M Calcium, which is displaced by calcium from the assumed source term. The CO_2 pressure is established at $10^{-2.5}$ bar (ten times atmospheric, representative for most soils; Lindsay, 1979). Nevertheless calcium does not precipitate as calcite, as the pH is too low.
- In the soil at Braambergen and Wieringermeer, calcite is present according to the STONE data (see Appendix 5), which is consistent with the high pHs that are reported there. In the model, calcium from the source term is allowed to precipitate in the soil profile as calcite; dissolved carbonate has the value associated with a CO_2 pressure of $10^{-2.5}$ bar.

Phosphate

The background concentration of phosphate in the soils is initially $5x10^{-7}$ M (0.05 mg PO4/l), derived from laboratory experiments (see Appendix 6). The concentration of phosphate in the leachate of all the landfills is 0.4 mg PO4/L ($4x10^{-6}$ M). This concentration will gradually displace the background concentration in the soil to $POC1_{gw}$ during the calculation.

For phosphate itself it is not possible to derive an unambiguous ETV. Validation has shown that phosphate is only moderately predictable using the ORCHESTRA model (for details see Dijkstra et al. 2009). Also, because of the very strong adsorption of phosphate in the soil predicted by the model, phosphate will not in most cases reach $POC2_{gw}$. If phosphate is included in the model, an unrealistic(ally high) ETV will be calculated, because of the strong sorption. The high phosphate concentration in the model will also occupy a lot of adsorption sites at the top of the soil profile of the model and thus influence other substances, as a result of which unrealistic(ally low) ETVs would be derived for substances such as sulphate and arsenic.

Because it is not possible to determine an ETV sufficiently reliably, phosphate is designated as a substance for special attention (see Chapters 7 and 8). Certainly for the Wieringermeer landfill, phosphate is a relevant substance because of the seepage route to the surface water, and phosphate has a eutrophying effect.

pH and redox potential

The pH is fixed in the model as a function of depth in the soil, in accordance with the soil properties from the STONE database. The consequence of the model design is that no account is taken of possible change in the pH in the soil as a result of an acidic or alkaline source term. The model possibilities for dynamic change in the pH are still in the development stage (particularly validation) and require input data that are not available.

In the model, a mild oxidative redox state is used that corresponds to pH + pe = 15, as a result of which the redox state in the soil is dependent on the pH and changes with depth (Lindsay, 1979). The same problem as outlined for the pH applies to the redox state and redox buffer capacity in the soil, namely that the model possibilities for dynamic redox change are still in the development stage (in particular validation) and require input data that are not available.

DOC

DOC (dissolved organic carbon) is a very important parameter that determines the mobility of other substances, particularly heavy metals that bind to reactive fractions of DOC (humic acid and fulvic acids). It is assumed (from the results of the sensitivity analysis, based on the advice of the TCB) that DOC from the landfill leachate will infiltrate the soil and replace the soil-native DOC.

The DOC concentrations that are assumed in the pore water of the soil are derived from the specific landfill leachates. To determine what the DOC concentration will become after active treatment of the landfills, an estimate is made. First, the median (50th percentile) values of DOC concentrations measured in the landfill leachates[3] for each location are taken from a database of leachate data available at the ECN. The expected DOC reduction after ten years of aeration and recirculation is a factor of 3–5, given the observed proportionality with N-Kjeldahl[4]. The median DOC concentration in the landfill leachate is therefore divided by 4 (the average of a factor of 3–5). This is, however, total DOC and not the reactive fraction that causes increased metal mobility. Therefore, 50% of the DOC is assumed to be reactive for metal binding, on the basis of measurements on organic matter in a bioreactor (ANVM 247 project; Comans et al. 2014).

At the time of writing there are still no models that can properly predict the concentration and transport of DOC in soils. The approach that is applied here, and also in the calculation of emission limit values for the re-use of building materials (Verschoor et al. 2006), is to keep the DOC concentrations in the pore water solution constant for each layer in the transport model. During the transport calculations the DOC concentrations as a function of depth were kept constant, but the DOC-bound fraction was used for the calculation of the transport of substances. Although the transport of DOC itself is not described in this way (the formation, adsorption and degradation of DOC in the soil are, as it were, in balance with each other), the effect of the DOC present on the transport of other substances is calculated correctly.

Ammonium

For ammonium (NH_4^+) there are no binding parameters available for specific adsorption to iron and aluminium (hydr)oxides and organic matter. However, on

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[3] Directly as DOC or via Chemical Oxygen Demand (COD) via a relation derived from a large data set, DOC = COD/2.51.

[4] Information gained from Mr H. Oonk, email dated 18/01/2013.

the basis of the transport velocities referred to in the scientific literature (including Böhlke et al. 2006) it can be assumed that ammonium is a relatively mobile substance, concentrations will break through within several decades (in any case within the time frame of 500 years) into the groundwater (see Appendix 4). The ETV is based on this transport behaviour (see Figures 4.1a and 4.1b). For the time being no account has been taken of degradation and/or conversions, which can limit the transport of ammonium. The occurrence or non-occurrence of this transformation is highly dependent on the local conditions present (see Appendix 4). There is still insufficient information available on the degradation or conversion of ammonium.

4.4 Dilution in the saturated zone between $POC1_{gw}$ and $POC2_{gw}$

In order to be able to make an estimate about the extent of mixing/dilution in the saturated zone and of the substance concentrations to be expected, two (pragmatic) assumptions were made (see conceptual model described in Section 3.2):

- There is a stationary situation with a net infiltration of 300 mm/year in the first aquifer.
- The bottom liner is no longer present and therefore has no influence on the amount of leachate that comes out of the bottom of the landfill.

The dilution of the water from the landfill was calculated on the basis of site-specific information such as the geometry of the landfill, the thickness and porosity of the aquifer and the difference in potential in the saturated zone (as supplied by the landfill operators). It was assumed that the leachate disperses uniformly at Kragge and Braambergen over the first aquifer. If the dilution, the background value and the concentrations at $POC1_{gw}$ are known, based on the law of the conservation of mass, the concentration at $POC2_{gw}$ is calculated as:

$$C2 = [C1 + (w-1)AW] / w$$

where:
C2 = the concentration at $POC2_{gw}$
C1 = the concentration at $POC1_{gw}$
AW = the background concentration in the groundwater used for the dilution
w = the hydrological dilution factor.

Appendix 7 describes in more detail how the dilution factors at Braambergen and Kragge were established for the groundwater. For Braambergen the dilution factor between $POC1_{gw}$ and $POC2_{gw}$ is 4.7 and for Kragge 1.4.

At Wieringermeer the dilution factor in groundwater was made equal to 1 (no dilution) because, although substances will infiltrate the groundwater locally as a result of pressure from the landfill, at a short distance from the landfill, a maximum of 750 m, it will seep to the surface (Van Someren, 2013). In this situation the leachate at 20 m from the landfill will be practically undiluted.

4.5 Calculation of site-specific emission testing values

Figure 4.3 shows schematically how the site-specific ETVs for landfills are calculated. All the substances will influence each other; therefore the ETVs are derived for all substances simultaneously. Because the sorption processes are non-linear, several model iterations are required to derive a set of ETVs. After each model iteration, the concentration at 500 years at $POC2_{gw}$ is compared with the groundwater criterion. If the concentration is not equal to the groundwater criterion, the concentration at POC0 is adapted for the next iteration by linear scaling and a new model run is generated. This iterative process continues until the ETVs in the source term for all substances result in concentrations at POC2$_{gw}$ at 500 years that meet the environmental protection criteria for groundwater, with a tolerance of 1% (i.e. the predicted concentration is allowed to deviate by a maximum of 1% from the target value; this is a result of the iterative and non-linear calculation process). If this criterion is met, the site-specific ETVs are established.

The same approach has been followed to develop emission limit values for the re-use of building materials in The Netherlands (Verschoor et al. 2006; Verschoor & Swartjes, 2008) and a similar approach was followed for recycled construction products in Germany (Dijkstra et al. 2013; Susset & Grathwohl, 2010).

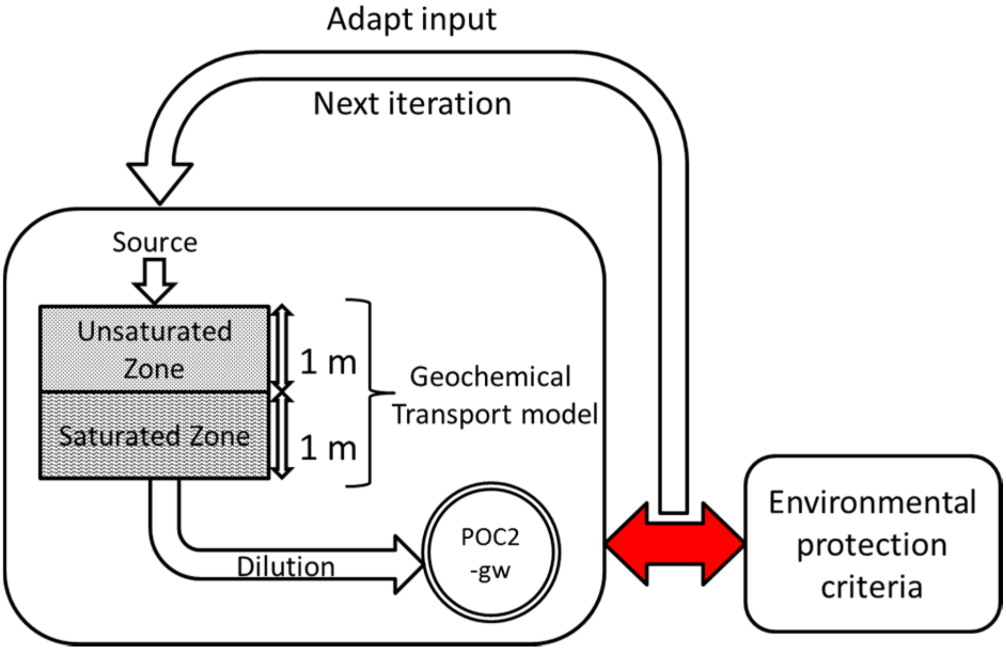

Figure 4.3: Methodology for calculating site-specific ETVs for the three pilot landfills. The resulting concentrations of the geochemical and dilution model at POC2$_{gw}$ are compared with the environmental protection criteria. If one or more of the substance concentrations is not equal to the environmental protection criteria, *the source term is adapted and a new iteration is started until the concentrations in the source term results are equal to the* environmental protection criteria.

Although the outflow of leachate and the concentration in the leachate from the landfill are assumed to be constant over 500 years, the concentrations at POC1$_{gw}$ are not because of the retardation of the substances in the soil as a result of binding processes (between POC0 and POC1$_{gw}$).

The concentrations at POC1$_{gw}$ increase over time until they are equal to the concentrations in the source term; a situation known as 'complete arrival' or 'complete breakthrough'. Most substances that bind strongly to the soil only partially arrive at POC1$_{gw}$, as a result of which the ETV for a lot of substances is higher than the environmental protection target for groundwater. This is substance-specific and also depends on the local soil properties. However, this effect is always time-dependent, i.e. dependent on the choice of time frame. Moreover, depending on the location, the dilution between $POC1_{gw}$ and $POC2_{gw}$ also plays a role in the magnitude of the ETVs.

4.5.1 *Calculation of ETVs for organic substances*

Because of the large number of organic substances, the ETVs for these substances are derived in a simplified manner, using a linear distribution coefficient. This distribution coefficient depends on two factors: (1) the distribution of a substance over the water phase and the natural content of organic matter in the soil (this is expressed in the log Koc) and (2) the distribution of natural organic matter itself in dissolved organic matter and solid organic matter. The latter distribution was calculated from the organic matter content in the soil per location (STONE database) and the assumed concentration of dissolved organic matter in the soil, derived from the landfill leachate (see Section 4.3.5). The overall distribution of a substance between the dissolved and solid phases is expressed as Kd (linear distribution coefficient), which is composed of the above two factors, here referred to as Kd1 and Kd2:

Kd1 is the distribution between substance that is bound to natural organic matter and substance dissolved in the water phase according to Appelo & Postma (2005):

$$Kd1 = Koc \times foc \qquad (L/kg)$$

where Koc is the reported Koc value for each substance, and foc is the fraction of organic (carbon) substance in the soil. The Koc is derived from the assessment model for human risks from contaminated soils called CSOIL 2000 (Brand et al. 2007).

Kd2 is the distribution of solid and dissolved natural organic matter between the solid phase and the water phase:

$$Kd2 = SOC\ (kg/kg)/DOC \qquad (kg/L)$$

The values for SOC and DOC (solid and dissolved natural organic matter, respectively) arise from the organic matter content in the soil per location (STONE database) and the assumed concentration of dissolved organic matter in the soil, derived from the landfill leachate.

The overall Kd (distribution coefficient) arises from Kd1 and Kd2 for the transport of organic substances in the soil:

$$Kd\ overall = Kd1 \times Kd2/Kd1 + Kd2$$

As there are a lot of organic substances, with different transport velocities, these substances are divided on the basis of the log Koc per location into classes with approximately the same transport velocity. This has the advantage that the use of an reactive transport model is limited. To this end, first of all the transport velocity per location was calculated for several substances across the whole range of Koc values. On the basis of these results the substances were divided into four classes of substance as follows:

- Class 1: Substances showing complete arrival (100% of the input concentration) within the time frame of 500 years.

- Class 2: Substances showing arrival of between 25% and 75% of the input concentration within the time frame.
- Class 3: Substances showing arrival of between 15% and 25% of the input concentration within the time frame.
- Class 4: Substances showing arrival of less than 15% of the input concentration within the time frame.

Then the ETV was calculated for each class per location, taking into account the extent of the arrival at the $POC2_{gw}$ after 500 years. If there is no dilution, the ETV for substances from Class 1 is equal to the environmental criterion in groundwater, as there is (virtually) complete arrival. For Class 2 the ETV is equal to twice the groundwater criterion, as this class of substances arrives at $POC2_{gw}$ after 500 years with approximately half of the source term concentration. If there is also dilution by a factor of 2, then the ETV is not twice but four times the groundwater criterion.

For Classes 3 and 4 the ETV is equal to four times and eight times the groundwater criterion, respectively (excluding dilution). In the determination of the ETV values for organic substances the following processes were not taken into account:

- biological degradation (which can further limit transport);
- gas phase transport (which can greatly speed up transport as it is an important transport route particularly for volatile substances);
- floating layers or subsidence layers (which can have an accelerating or a decelerating effect). Floating layers and subsidence layers are often formed by pure products such as mineral oil. The thinking behind sustainable landfill management is that emissions are reduced after the treatment period in such a way that there is no more pure product present as a result of which floating layers and subsidence layers could form.

4.6 Transport towards surface water

The dispersion route to surface water is of importance only at Wieringermeer because of the seepage situation present. The hydrological model simulations of Van Someren (2013) show that if the bottom liner is no longer functional, both the ditch and the groundwater are influenced by the leachate from the landfill. The surface water of the channel is, according to these simulations, scarcely influenced by the direct inflow of leachate (carried via the groundwater) (see Figure 3.2). The transfer of surplus water from the ditch will, on the other hand, influence the concentrations in the channel. It is not known how great this transfer is per year, as this is not monitored.

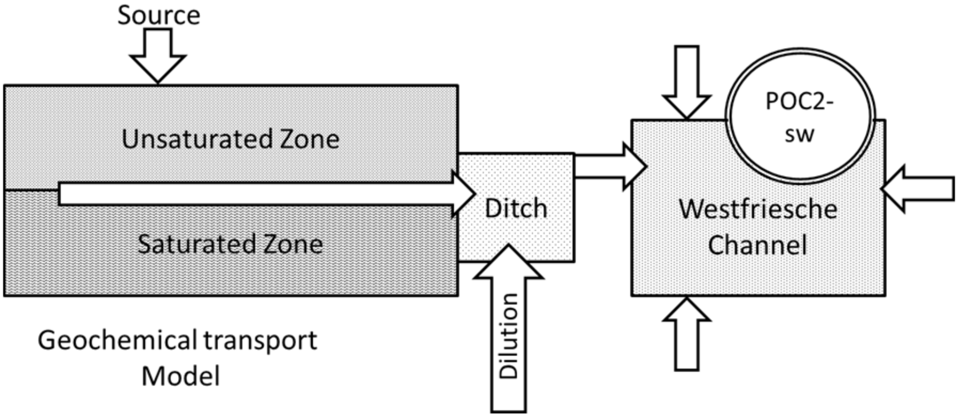

Figure 4.4: Schematic overview of the calculation of the concentration in surface water at $POC2_{sw}$.

Figure 4.4 shows schematically how the concentrations in the channel are calculated. The water from the landfill flows via the dike surrounding the landfill to the ditch (see also green arrow in Figure 4.1B). This means that there is a soil passage and shallow run-off via groundwater. No assessment is carried out in this ditch as this is regarded by the competent authority as part of the construction of the landfill. The water from the landfill is diluted in the ditch and then the diluted water is discharged via a transfer into the Westfriesche Vaart, where the calculated concentration is tested against the valid local surface water quality criteria for macroparameters and the JG-MKN or MPR_{eco} for surface water in the absence of a JG-MKN (metals and organic substances).

The concentrations that occur in the groundwater are calculated, as with the dispersion via the infiltration situation, using the same geochemical transport model as is used for the dispersion to groundwater, but with a few small modifications.

The transport of substances for which geochemical reactions in the seepage situation of the Wieringermeer pilot landfill are taken into account is calculated across the full distance of 20 metres to $POC1_{sw}$. Instead of two homogeneous sections of 1 metre thickness, the transport in this situation was calculated over five homogeneous sections of 4 metres length (in total 20 metres). The model design thus complies with the rule of thumb that the extent of 'mixing' on the path as a result of processes such as dispersion is approximately a 1/10 part of the distance covered. The average soil properties of the STONE plot of this location are assumed for the whole path. These properties are comparable with the soil data for the location supplied by the landfill operator. The transport was calculated with the same infiltration rate as with the groundwater scenarios (300 mm/year, effective pore water velocity 1 metre/year) without dilution on the path between POC0 and $POC1_{sw}$. Dilution effects for salts on the path to $POC1_{sw}$ were discounted in the calculation of the permitted concentration in the ditch at $POC1_{sw}$.

4.6.1 *Dilution in the ditch*

The leachate from the landfill that gets into the ditch is diluted by water that seeps under the landfill and the neighbouring arable land (see Figure 4.5). In addition, the water in the ditch is diluted by:

- infiltrating rainwater from the various parts of the landfill in the form of purified leachate;
- infiltrating rainwater from the West landfill (not built);
- infiltrating rainwater from the business park and run-off from the business park;
- infiltrating rainwater from a small piece of surrounding arable land.

The concentration of the water in the ditch can be calculated according to:

$$c_{R,j} = \frac{\sum_i (c_{i,j} * Q)_i}{\sum_i Q_i}$$

where:

$c_{R,j}$ = resulting concentration in the ditch (µg/L)

$c_{i,j}$ = concentration of substance j in source i (or the various sections) (µg/L)

Q_i = flow rate from the various sources (or sections) (m^3).

Figure 4.5: Aerial view of the Wieringermeer pilot landfill.

The dilution factor is defined as the quotient of the substance concentration in the leachate or drain from the landfill for substance *j* from source *i* and the resulting substance concentration in the water from the ditch.

$$D_j = \frac{c_{i,j}}{c_{R,j}}$$

where:

$c_{i,j}$ = substance concentration of substance j in source i (µg/L)

$c_{R,j}$ = resulting substance concentration of substance j in ditch (µg/L).

Table 4.2 provides an overview of the flow rates that were used to calculate the dilution factor in the ditch.

Table 4.2: Contribution of different surfaces at Wieringermeer to total the total water flow into the ditch. Surface area is estimated on the basis of GIS data.

Different surfaces at Wieringermeer pilot landfill	**Surface area (ha)**	**Precipitation (mm/year)**	**Contribution precipitation to dilution in ditch**	**Flow (m^3/year)**
Seepage	75.4	36.5	100%	27,521
Arable land	17.9	300	100%	44,700
Business park run-off (built)	21.7	650	67%	94,504
Business park infiltration		300	33%	21,483
Wieringermeer East (built)	20	300	40%	27,600
Wieringermeer West (built)	10.8	300	100%	32,400
Wieringermeer West (not built)	5	300	100%	15,000
Total water flow into ditch	**75.4**			**263,208**

It was assumed for the calculation of the dilution in the ditch that the concentration in the seepage water is equal to the background concentration in the groundwater. The water from the business park, the run-off and infiltration from the not-built part of the landfill is designated as not polluted. The water from the arable land contains a certain amount of ammonium, but how much this is, is not known.

The concentrations in the leachate from the various landfill sections that are built are made equal to the average concentrations in the effluent from the water purification plant during the last four years, as reported to the competent authority. This leachate is purified only for ammonium, as a result of which the concentration of ammonium in the unpurified leachate will be higher than reported. For substances that are not monitored (often organic substances) or for which no good measurement values are available, no dilution factor can be established. Ultimately, it was only possible to establish an indicative dilution factor for chloride and ammonium of 1.8 and 10.3, respectively. For the other substances the dilution factor cannot be properly estimated and was set as 1 (no dilution).

The dilution factors can become higher in the future if the concentrations in the leachate fall during the experiment. It is recommended that the dilution factors be re-established shortly before the end of the period of active treatment based on the measurement data then known.

4.6.2 *Dilution in the Westfriesche Vaart*

The water from the ditch will be further diluted when it gets into the Westfriesche Vaart. From a policy point of view this is a discharge from the landfill to the Westfriesche Vaart that has to be assessed in accordance with the WFD immission test (Ministry of I&M, 2011a; Kleissen, 2012). The ditch itself is regarded as part of the construction. This test ascertains whether the water quality objectives set are exceeded in the surface water, at POC2$_{sw}$, as a result of the discharge.

In this test the concentration is calculated for a certain substance near the discharge point, the dilution in the mixing zone behind the discharge point being taken into account. The test includes the current national water quality standards (JG-MKN/MPR_{eco}) so that it can be directly determined whether the standard is exceeded. The regional water quality standard can be filled in manually.

The immission test is used to determine the dilution factor in the Westfriesche Vaart and the maximum permitted concentrations in the ditch. Appendix 7 shows the results of the immission test.

5 Results

The following sections present the model results. First, a general explanation is given of how to interpret the output of the model. Thereafter the pilot-specific lists of ETVs are presented. The results are reflected upon in more detail in chapter 7.

5.1 General interpretation of results and examples of different concentration/time behaviour at POC2$_{gw}$

The mobility of a substance, and therefore the time at which it reaches POC2$_{gw}$, depends on the modelled binding strength of the substance to the solid phase of the soil. Soluble salts will travel quickly through the soil, while many heavy metals are strongly delayed due to strong binding on soil particles. However, it is important to note that regardless of the degree of binding of a substance, a constant concentration in and amount of the leachate coming from the landfill implies that sooner or later all substances will reach the same concentration at POC2$_{gw}$ as in the leachate (only if dilution in groundwater takes place can concentrations be lower at POC2$_{gw}$ than in the leachate).

Therefore, to understand the results, examples are shown below. It is important to note that calculation of an ETV also requires a time frame within which the (modelled) concentration at POC2 must be below a certain environmental criterion (in groundwater, surface water or soil). If no time frame is used (implying an infinite or eternal time frame), and if no hydrological dilution is taken into account, the ETVs from the landfill will be equal to the groundwater protection criteria, as all substances will eventually arrive at the designated POC2.

In this study, sorption is taken into account only in the unsaturated zone and the upper metre of the saturated zone; between the saturated zone and POC2$_{gw}$ no sorption is taken into account, only hydrological dilution. Given the boundary conditions of this study, the site-specific ETVs will usually be different from the environmental criteria at POC2$_{gw}$ because of:

- retardation (binding) of the substances in the unsaturated zone and the first metre of the saturated zone in such a way that the maximum concentration at POC2$_{gw}$ is not reached within the time frame. For substances that are not subject to binding to soil (e.g. salts), the maximum will be reached within a few years. Substances that show strong binding will only partially arrive at POC2$_{gw}$, or in some cases will not arrive at POC2$_{gw}$ at all, within the set time frame of 500 years.
- hydrological dilution of the leachate coming from the landfill (dilution factor).

In general, the results allow a distinction to be made between four types of substance behaviour at POC2$_{gw}$. They are described as Behaviours A, B, C and D for the infiltration scenario (Figure 5.1). In Figure 5.1, the (constant) volume and concentration of leachate coming from the landfill is in all cases calculated such that the concentration in the groundwater at POC2$_{gw}$ meets the groundwater criterion within the period of 500 years, i.e. the concentration in the leachate represents the site-specific ETV and no further iterations are necessary. In these examples the model calculations were performed for up to 1000 years to indicate the effects of arrival of the substances at POC2$_{gw}$ after the chosen time frame of 500 years.

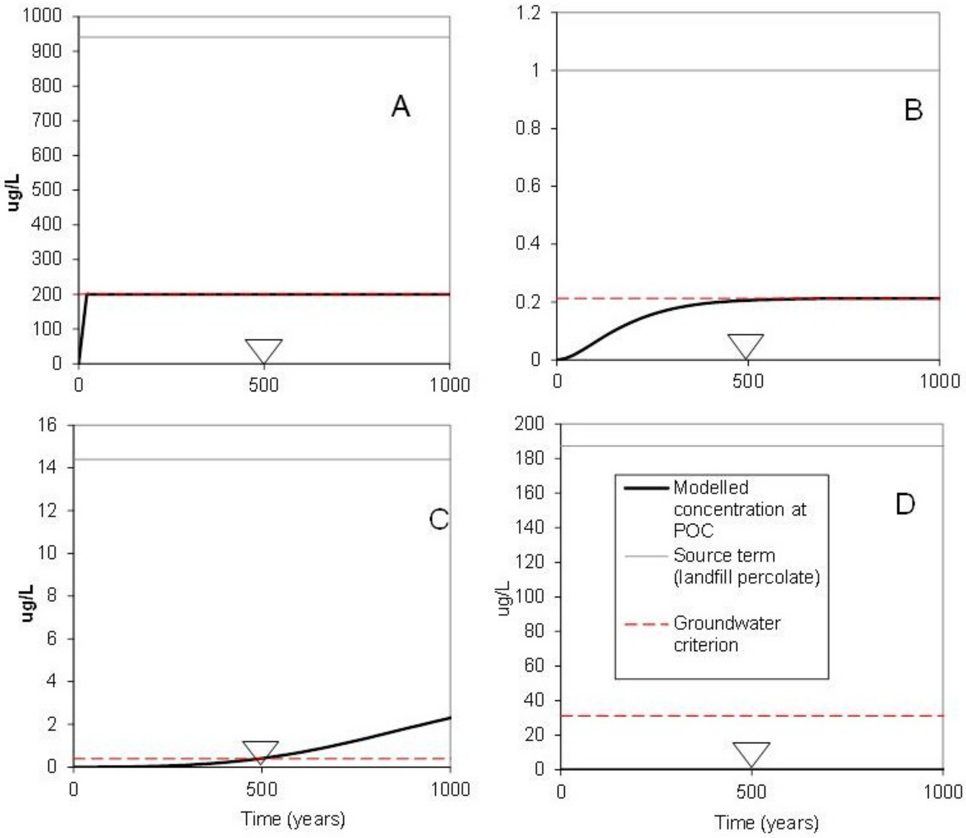

Figure 5.1: Concentration/time profiles of four substances with different retardation behaviours at POC2$_{gw}$*. The triangles indicate the end of the time frame (500 years) during which the concentration at POC2*$_{gw}$ *must be below the groundwater criterion. Behaviour A: mobile substances such as soluble salts arrive rapidly at POC2*$_{gw}$*; Behaviour B: substances that show mild retardation still arrive at POC2*$_{gw}$ *within the time frame of 500 years; Behaviour C: strong retardation: substances only partially arrive at POC2*$_{gw}$ *within 500 years; Behaviour D: extremely strong retardation: substances do not arrive at POC2*$_{gw}$*.*

Behaviour A is representative of mobile substances such as soluble salts (e.g. chloride). These substances travel with the same velocity as water and reach their maximum concentration in groundwater after a few years (Figure 5.1A). Substances that show mild retardation according to Behaviour B may also reach their maximum within 500 years.

In the case of both Behaviour A and Behaviour B, the difference between the constant concentration in the leachate and the maximum concentration in the groundwater equals the dilution factor at this location. In other words, the maximum tolerable concentration coming from the landfill (= ETV at POC0) can be higher than the environmental protection criteria for groundwater and equal to the dilution factor. In the case of Behaviours A and B, retardation is not strong enough to affect the calculation of the site-specific ETV.
On the contrary, substances that show Behaviour C are strongly delayed, resulting in only partial arrival at POC2$_{gw}$ after 500 years. In this case, too, the leachate concentration is calculated such that the concentration after 500 years meets the groundwater criterion. In this case, the difference between the concentration at POC2$_{gw}$ after 500 years and the maximum tolerable

concentration coming from the landfill consists of a dilution factor and a remaining factor due to retardation and partial arrival at $POC2_{gw}$.

From Figure 5.1C it is clear that the ETV is very sensitive to the chosen time frame. It should be stressed that in the case of substances showing Behaviour C, the concentrations of substances at $POC2_{gw}$ will continue to increase after 500 years to values well above the groundwater criterion. Behaviour C is commonly found in ‘moderately mobile’ substances, which include heavy metals such as copper and chromium. The effect of the time frame on the calculated ETVs also depends on factors such as the assumed thickness of the (un)saturated zone in which the sorption of substances is assumed to occur, the groundwater flow rate, and the distance between the source and $POC2_{gw}$. The findings with respect to the time frame dependency of ETVs for ‘moderately mobile’ substances are not unique to this study, but were identified previously in the calculation of emission limit values for building materials for the Dutch Soil Quality Decree (Dijkstra et al. 2013; Verschoor et al. 2006) and of emission limit values for the German recycling decree (Dijkstra et al. 2013; Susset & Grathwohl, 2010).

Behaviour D is characteristic of substances that do not reach $POC2_{gw}$ at all within the calculation period. This behaviour is sometimes found in substances that are predicted to form a precipitate in the soil below the landfill, such as $Pb(OH)_2(s)$ at high pH. Arsenic is in some cases also predicted to be sorbed so strongly to the soil (as arsenate) that it does not reach $POC2_{gw}$. Under these conditions the substances are predicted to accumulate below the landfill and will not be transported towards $POC2_{gw}$. This behaviour implies that it is not possible to determine a site-specific ETV based on a groundwater criterion at $POC2_{gw}$, because the concentration in the leachate would become infinitely high. In the case of Behaviour D, it was investigated whether the environmental quality criterion of total content in the soil (mg/kg) would be useful for deriving an ETV, as was done for the Soil Quality Decree for building materials and large-scale applications of (excavated) soil and sludge (Verschoor et al. 2006; Verschoor & Swartjes, 2008).

Whether a specific substance shows Behaviour A, B, C or D depends on the intrinsic chemical properties of the substance (chemical reactivity) and on site-specific properties, such as the pH and the amounts of reactive surfaces present (organic matter, Fe/Al hydroxides and clay content). Therefore, a general division of substances into the above categories is not possible, as they are site-specific. Exceptions are soluble salts, which are chemically non-reactive and will show in all scenarios mobility according to Behaviour A. The next section will present the calculated ETVs.

5.2 Calculated emission testing values

The following sections present the modelled ETVs. For various reasons it may be necessary to deviate from these calculated values when defining the final list. For example, when ETVs are below the LOQ. However, for the sake of traceability the calculated values are presented in the following sections. Appendix 8 presents the corresponding modelling curves for each pilot landfill.

5.2.1 Emission testing values for the Braambergen pilot landfill

Table 5.1 presents the background concentrations in groundwater, the environmental criterion at $POC2_{gw}$ and the calculated ETVs for the Braambergen pilot landfill. In general the ETVs were derived from the environmental criterion at $POC2_{gw}$. Where this is not the case, it is indicated in the column ‘Remarks’.

Table 5.1: Background concentrations in groundwater, environmental criterion at POC2$_{gw}$ and the calculated emission testing values for the Braambergen pilot landfill.

Substance	Background concentration	Environmental criterion at POC2$_{gw}$	ETV	Remarks
Inorganic substances (µg/L)				
Arsenic	1	10	190	Based on soil criterion due to binding with HFO
Cadmium	0.1	0.44	6.4	
Chromium	1	9.7	210	
Copper	5	6.1	50	
Mercury	0.1	0.33	5.8	
Lead	1	10	60,000	Based on soil criterion due to deposition of Pb(OH)$_2$
Nickel	1	2.9	21	
Zinc	5	12.3	160	
Free cyanides	n.a.	5	61	
Macroparameters (mg/L)				
Chloride (mg/L)	8.1	102	450	
N-Kjeldahl/ ammonium (mg/L)	1.88	1.8	1.8	
Sulphate (mg/L)	1	150	700	
Phosphate (mg/L)	n.a.	n.a.	n.a.	
Organic substances (µg/L)				
Mineral oil aliphatic				
EC5-EC6	n.a.	0.17	0.8	Calculated ETV below LOQ
EC6-EC8	n.a.	0.039	0.37	
EC8-EC10	n.a.	0.005	0.047	
EC10-EC12	n.a.	0.00127	0.00127	
EC12-EC16	n.a.	0.00071	0.00071	
EC16-EC21	n.a.	-	-	
Mineral oil aromatic (µg/L)				
EC5-EC7	n.a.	1.23	4.7	Calculated ETV below LOQ
EC7-EC8	n.a.	0.83	3.9	
EC8-EC10	n.a.	0.55	2.6	
EC10-EC12	n.a.	0.32	1.5	
EC12-EC16	n.a.	0.14	1.3	
EC16-EC21	n.a.	0.038	0.36	
EC21-EC35	n.a.	0.0032	0.06	
Mineral oil sum EC10-EC40	n.a.	50	470	
VOX (µg/L)				
Vinylchloride	n.a.	0.01	0.047	Calculated ETV below LOQ
Dichloromethane	n.a.	0.01	0.047	
1,1 dichloroethane	n.a.	1	4.7	
1,2 dichloroethane	n.a.	3	14	
1,1 dichloroethene	n.a.	0.01	0.047	Calculated ETV below LOQ
1,2 dichloroethene (cis,trans)	n.a.	0.01	0.047	
Dichloropropane (1,2)	n.a.	0.8	3.8	
Dichloropropane (1,3)	n.a.	0.8	3.8	
Trichloromethane	n.a.	1	4.7	

Substance	Background concentration	Environmental criterion at $POC2_{gw}$	ETV	Remarks
(chloroform)				
1,1,1 trichloroethane	n.a.	0.01	0.047	Calculated ETV below LOQ
1,1,2 trichloroethane	n.a.	0.01	0.047	
Trichloroethene (tri)	n.a.	10	47	
Tetrachloromethane (tetra)	n.a.	0.01	0.047	Calculated ETV below LOQ
Tetrachloroethene (per)	n.a.	0.01	0.047	
PAH (µg/L)				
Naftalene	n.a.	0.01	0.047	
Phenantrene	n.a.	0.003	0.028	
Antracene	n.a.	0.0007	0.0066	Calculated ETV below LOQ
Fluoranthene	n.a.	0.003	0.056	
Chrysene	n.a.	0.003	0.056	
Benzo(a)antracene	n.a.	0.0001	0.0019	Calculated ETV below LOQ
Benzo(a)pyrene	n.a.	0.0005	0.0094	
Benzo(k)-fluoranthene	n.a.	0.0004	0.0075	
Indeno(1,2,3cd)-pyrene	n.a.	0.0004	0.0075	
Benzo(ghi)perylene	n.a.	0.0003	0.0056	
PAH (sum10)	n.a.	0.1	1.9	
BTEX (µg/L)				
Benzene	n.a.	0.2	0.94	
Xylene	n.a.	0.2	0.94	
Toluene	n.a.	1	4.7	
Ethylbenzene	n.a.	1	4.7	
Other (µg/L)				
Phenols	n.a.	0.2	0.94	

n.a. = not applicable

- = not available

5.2.2 *Emission testing values for the Kragge pilot landfill*

Table 5.2 presents the background concentrations in groundwater, the environmental criterion at $POC2_{gw}$ and the calculated ETVs for the Kragge pilot landfill. In general the ETVs were derived from the environmental criterion at $POC2_{gw}$. Where this is not the case, it is indicated in the column 'Remarks'.

Table 5.2: Background concentrations in groundwater, environmental criterion at POC2$_{gw}$ and the calculated ETVs for the Kragge pilot landfill.

Substance	Background concentration	Environmental criterion at POC2$_{gw}$	ETV	Remarks
Inorganic substances (µg/L)				
Arsenic	1	10	100	
Cadmium	0.1	0.44	3.6	
Chromium	2.65	11.35	140	
Copper	5	6.10	64	
Mercury	0.1	0.33	4.1	
Lead	1	10	130	
Nickel	5.15	7.05	47	
Zinc	9.4	16.7	120	
Free cyanides	n.a.	5	6.8	
Macroparameters (mg/L)				
Chloride (mg/L)	32	126	160	
N-Kjeldahl/ ammonium (mg/L)	1.1	1.1	1.1	
Sulphate (mg/L)	6.2	150	200	
Phosphate (mg/L)	n.a.	n.a.	n.a.	
Organic substances (µg/L)				
Mineral oil aliphatic (µg/L)				
EC5-EC6	n.a.	0.17	0.23	Calculated ETV below LOQ
EC6-EC8	n.a.	0.039	0.11	
EC8-EC10	n.a.	0.005	0.054	
EC10-EC12	n.a.	0.00127	0.014	
EC12-EC16	n.a.	0.00071	0.0077	
EC16-EC21	n.a.	-	-	
Mineral oil aromatic (µg/L)				
EC5-EC7	n.a.	1.23	1.4	Calculated ETV below LOQ
EC7-EC8	n.a.	0.83	2.3	
EC8-EC10	n.a.	0.55	1.5	
EC10-EC12	n.a.	0.3	0.87	
EC12-EC16	n.a.	0.14	0.38	
EC16-EC21	n.a.	0.038	0.21	
EC21-EC35	n.a.	0.0032	0.035	
Mineral oil sum EC10-EC40	n.a.	50	270	
VOX (µg/L)				
Vinylchloride	n.a.	0.01	0.014	Calculated ETV below LOQ
Dichloromethane	n.a.	0.01	0.014	
1,1 dichloroethane	n.a.	1	1.4	
1,2 dichloroethane	n.a.	3	4.1	
1,1 dichloroethene	n.a.	0.01	0.014	Calculated ETV below LOQ
1,2 dichloroethene (cis,trans)	n.a.	0.01	0.014	
Dichloropropane (1,2)	n.a.	0.8	1.1	
Dichlorpropane (1,3)	n.a.	0.8	1.1	
Trichloromethane (chloroform)	n.a.	1	1.4	
1,1,1 trichloroethane	n.a.	0.01	0.014	Calculated ETV below LOQ
1,1,2 trichloroethane	n.a.	0.01	0.014	

Substance	Background concentration	Environmental criterion at POC2$_{gw}$	ETV	Remarks
Trichloroethene (tri)	n.a.	10	14	
Tetrachloromethane (tetra)	n.a.	0.01	0.014	Calculated ETV below LOQ
Tetrachloroethene (per)	n.a.	0.01	0.014	
PAH (μg/L)				
Naftalene	n.a.	0.01	0.014	Calculated ETV below LOQ
Phenantrene	n.a.	0.003	0.016	
Antracene	n.a.	0.0007	0.0038	
Fluoranthene	n.a.	0.003	0.033	
Chrysene	n.a.	0.003	0.033	
Benzo(a)antracene	n.a.	0.0001	0.0011	Calculated ETV below LOQ
Benzo(a)pyrene	n.a.	0.0005	0.0054	
Benzo(k)-fluoranthene	n.a.	0.0004	0.0044	
Indeno(1,2,3cd)-pyrene	n.a.	0.0004	0.0044	
Benzo(ghi)perylene	n.a.	0.0003	0.0033	
PAH (sum10)	n.a.	0.1	1.1	
BTEX (μg/L)				
Benzene	n.a.	0.2	0.27	
Xylene	n.a.	0.2	0.27	
Toluene	n.a.	1	1.4	
Ethylbenzene	n.a.	1	1.4	
Other (μg/L)				
Phenols	n.a.	0.2	0.27	

n.a. = *not applicable*
\- = *not available*

5.2.3 *Emission testing values for the Wieringermeer pilot landfill*

Table 5.3 presents the background concentrations in groundwater, the environmental criterion at POC2$_{gw}$ and the calculated ETVs for the two scenarios (groundwater and surface water) of the Wieringermeer pilot landfill. In general the ETVs in the groundwater scenario were derived from the environmental criterion at POC2$_{gw}$. Where this was not the case, it is indicated in the column 'Remarks'.

It was decided that for Wieringermeer the lower value of the two scenarios would be the calculated ETV. Table 5.4 presents the final list of calculated ETVs for the Wieringermeer pilot landfill.

Table 5.3: Background concentrations in groundwater, environmental criterion at $POC2_{gw}$ and the calculated ETVs for the Wieringermeer pilot landfill for both surface water (SW) and groundwater (GW) scenarios.

Substance	**Background concentration GW**	**Environmental criterion at POC2**		**ETV**		**Remarks**
		SW	**GW**	**SW**	**GW**	
Inorganic substances (µg/L)						
Arsenic	17.05	44.9	17.05	190	190	ETV**gw** and **sw** are based on a soil criterion
Cadmium	0.1	0.38	0.44	160	1.3	ETV**sw** is based on a soil criterion
Chromium	3.05	7.8	11.75	79	37	
Copper	5	9.7	6.1	700	19	
Mercury	0.1	0.089	0.33	390	1	
Lead	1	10.8	10	11,000	25,000	ETV**gw** and **sw** are based on soil criterion due to deposition of $Pb(OH)_2$
Nickel	19.75	30.8	20	54	21	ETV**sw** is based on a soil criterion
Zinc	5	20.6	12.3	3,300	39	
Free cyanides	n.a.	0.3	5	2.1	35	
Macroparameters (mg/L)						
Chloride (mg/L)	2,300	1,934	2,394	1,900	2,400	
N-Kjeldahl/ Ammonium (mg/L)	50	14.4	50	14	50	
Sulphate (mg/L)	1,400	860	1,400	860	1,400	
Phosphate (mg/L)	n.a.	3.9	n.a.	*	n.a.	
Organic substances (µg/L)						
Mineral oil aliphatic (µg/L)						
EC5-EC6	n.a.	23.1	0.17	180	0.17	Calculated ETV**gw** below LOQ. ETV**sw** based on a soil criterion
EC6-EC8	n.a.	7.6	0.039	61	0.039	
EC8-EC10	n.a.	0.97	0.005	7.8	0.01	
EC10-EC12	n.a.	0.25	0.0013	2	0.0025	
EC12-EC16	n.a.	0.13	0.00071	1.1	0.0014	
EC16-EC21	n.a.	-	-	-	-	
Mineral oil aromatic (µg/L)						
EC5-EC7	n.a.	159.9	1.23	1300	1.2	Calculated ETV**gw** below LOQ. ETV**sw** based on a soil criterion
EC7-EC8	n.a.	107.9	0.83	860	0.83	
EC8-EC10	n.a.	0.97	0.55	570	0.55	
EC10-EC12	n.a.	43.5	0.32	350	0.32	
EC12-EC16	n.a.	19.0	0.14	150	0.28	
EC16-EC21	n.a.	7.4	0.038	59	0.076	
EC21-EC35	n.a.	0.62	0.0032	5	0.0064	
Mineral oil sum EC10-EC40	n.a.	68	50	540	100	ETV**sw** based on a soil criterion
VOX (µg/L)						
Vinylchloride	n.a.	0.17	0.01	0.17	0.01	Calculated emission testing value **gw** below LOQ. Emission testing value sw based on a soil

Substance	Background concentration GW	Environmental criterion at POC2		ETV		Remarks
		SW	GW	SW	GW	
						criterion
Dichloromethane	n.a.	27.2	0.01	27	0.01	Calculated ETV**gw** below LOQ. ETV**sw** based on a soil criterion
1,1 dichloroethane	n.a.	910	1	910	1	ETV**sw** is based on a soil criterion
1,2 dichloroethane	n.a.	13.6	3	14	3	
1,1 dichloroethene	n.a.	12.2	0.01	12	0.01	Calculated ETV**gw** below LOQ. ETV**sw** based on a soil criterion
1,2 dichloroethene (cis,trans)	n.a.	9.2	0.01	9.2	0.01	
Dichloropropane (1,2)	n.a.	364	0.8	360	0.8	ETV**sw** is based on a soil criterion
Dichloropropane (1,3)	n.a.	98.8	0.8	99	0.8	
Trichloromethane (chloroform)	n.a.	4.9	1	4.9	1	
1,1,1 trichloroethane	n.a.	28.6	0.01	29	0.01	Calculated ETV**gw** below LOQ. ETV**sw** based on a soil criterion
1,1,2 trichloroethane	n.a.	29.9	0.01	30	0.01	
Trichloroethene (tri)	n.a.	13.6	10	14	10	ETV**sw** based on a soil criterion
Tetrachloromethane (tetra)	n.a.	16.3	0.01	16	0.01	Calculated ETV**gw** below LOQ. ETV**sw** based on a soil criterion
Tetrachloroethene (per)	n.a.	13.6	0.01	14	0.01	
PAH (µg/L)						
Naftalene	n.a.	4.7	0.01	19	0.01	Calculated ETV**gw** below LOQ. ETV**sw** based on a soil criterion
Phenantrene	n.a.	0.58	0.003	4.7	0.006	
Antracene	n.a.	0.19	0.0007	1.6	0.0014	
Fluoranthene	n.a.	0.19	0.003	1.6	0.006	
Chrysene	n.a.	1.7	0.003	14	0.006	
Benzo(a)antracene	n.a.	0.057	0.0001	0.46	0.0002	
Benzo(a)pyrene	n.a.	0.095	0.0005	0.76	0.001	
Benzo(k)-fluoranthene	n.a.	0.057	0.0004	0.46	0.0008	
Indeno(1,2,3cd)-pyrene	n.a.	0.0038	0.0004	0.03	0.0008	
Benzo(ghi)perylene	n.a.	0.0038	0.0003	0.03	0.0006	
PAH (sum10)	n.a.	-	0.1	-	0.2	ETV**sw** based on a soil criterion
BTEX (µg/L)						
Benzene	n.a.	13.6	0.2	14	0.2	ETV**sw** based on a soil criterion
Xylene	n.a.	4.7	0.2	4.7	0.2	
Toluene	n.a.	96.2	1	96	1	

Substance	Background concentration GW	Environmental criterion at POC2		ETV		Remarks
		SW	GW	SW	GW	
Ethylbenzene	n.a.	481	1	960	1	
Other (µg/L)						
Phenols	n.a.	130	0.2	130	0.2	ETV**sw** based on a soil criterion

n.a. = *not applicable*

\- = *not available*

* = *For phosphate no reliable ETVs can be calculated (see also Section 4.3.5). From the sensitivity analysis it can be concluded that as long as the concentrations of phosphate in the leachate remain below 150µg/L, phosphate will probably not reach the surface water. This value should, however, not be interpreted as an emission testing value of any kind.*

Table 5.4: Selected background concentrations in groundwater, environmental criterion at $POC2_{gw}$ and the calculated ETVs for the Wieringermeer pilot landfill.

Substance	Background concentration GW	Environmental criterion at POC2	ETV	Remarks
Inorganic substances (µg/L)				
Arsenic	17.05	17.05	190	
Cadmium	0.1	0.44	1.3	
Chromium	3.05	11.75	37	
Copper	5	6.1	19	
Mercury	0.1	0.33	1	
Lead	n.a.	10	11,000	ETV based on criterion for soil
Nickel	19.75	20	21	
Zinc	5	12.3	39	
Free cyanides	n.a.	5	35	
Macroparameters (mg/L)				
Chloride (mg/L)	n.a.	1,934	2,400	ETV based on criterion for surface water
N-Kjeldahl/ Ammonium (mg/L)	n.a.	14.4	50	
Sulphate (mg/L)	n.a.	860	1,400	
Phosphate (mg/L)	n.a.	3.9	*	
Organic substances (µg/L)				
Mineral oil aliphatic (µg/L)				
EC5-EC6	n.a.	0.17	0.17	Calculated ETVgw below LOQ
EC6-EC8	n.a.	0.039	0.039	
EC8-EC10	n.a.	0.005	0.01	
EC10-EC12	n.a.	0.00127	0.0025	
EC12-EC16	n.a.	0.00071	0.0014	
EC16-EC21	n.a.	-	-	
Mineral oil aromatic (µg/L)				
EC5-EC7	n.a.	1.23	1.2	Calculated ETVgw below LOQ
EC7-EC8	n.a.	0.83	0.83	
EC8-EC10	n.a.	0.55	0.55	
EC10-EC12	n.a.	0.32	0.32	
EC12-EC16	n.a.	0.14	0.28	
EC16-EC21	n.a.	0.038	0.076	
EC21-EC35	n.a.	0.0032	0.0064	
Mineral oil sum EC10-EC40	n.a.	50	100	
VOX (µg/L)				
Vinylchloride	n.a.	0.01	0.01	Calculated ETVgw below LOQ
Dichloromethane	n.a.	0.01	0.01	
1,1 dichloroethane	n.a.	1	1	
1,2 dichloroethane	n.a.	3	3	
1,1 dichloroethene	n.a.	0.01	0.01	Calculated ETVgw below LOQ
1,2 dichloroethene (cis,trans)	n.a.	0.01	0.01	
Dichloropropane (1,2)	n.a.	0.8	0.8	
Dichloropropane (1,3)	n.a.	0.8	0.8	
Trichloromethane	n.a.	1	1	

Substance	Background concentration GW	Environmental criterion at POC2	ETV	Remarks
(chloroform)				
1,1,1 trichloroethane	n.a.	0.01	0.01	Calculated ETV$_{gw}$ below LOQ
1,1,2 trichloroethane	n.a.	0.01	0.01	
Trichloroethene (tri)	n.a.	10	10	
Tetrachloromethane (tetra)	n.a.	0.01	0.01	Calculated ETV$_{gw}$ below LOQ
Tetrachloroethene (per)	n.a.	0.01	0.01	
PAH (µg/L)				
Naftalene	n.a.	0.01	0.01	Calculated ETV$_{gw}$ below LOQ
Phenantrene	n.a.	0.003	0.006	
Antracene	n.a.	0.0007	0.0014	
Fluoranthene	n.a.	0.003	0.006	
Chrysene	n.a.	0.003	0.006	
Benzo(a)antracene	n.a.	0.0001	0.0002	
Benzo(a)pyrene	n.a.	0.0005	0.001	
Benzo(k)-fluoranthene	n.a.	0.0004	0.0008	
Indeno(1,2,3cd)-pyrene	n.a.	0.0004	0.0008	
Benzo(ghi)perylene	n.a.	0.0003	0.0006	
PAH (sum10)	n.a.	0.1	0.2	
BTEX (µg/L)				
Benzene	n.a.	0.2	0.2	
Xylene	n.a.	0.2	0.2	
Toluene	n.a.	1	1	
Ethylbenzene	n.a.	1	1	
Other (µg/L)				
Phenols	n.a.	0.2	0.2	

n.a. = not applicable

- = not available

** = For phosphate no reliable ETVs can be calculated (see also Section 4.3.5). From the sensitivity analysis it can be concluded that as long as the concentrations of phosphate in the leachate remain below 150µg/L, phosphate will probably not reach the surface water. This value should, however, not be interpreted as an emission testing value of any kind.*

6 Sensitivity analysis

A sensitivity analysis was carried out in order to study the influence of several important parameters on the magnitude of the ETVs. The following sections describe the specific choices that were made for the parameters to be used in the sensitivity analysis and the way in which the result of the analisys was used in the final geochemical transport model used to derive the ETVs.

It is important to note that the sensitivity analysis was performed with a parametization of the model that was similar to but not exactly the same as (particularly, lower DOC and higher PO_4) that of the final version, which was described in chapter 4 and used to derive the ETVs as presented in chapter 5.

The final parameter choices were derived from discussions within the project team and the points of special attention in the recommendations of the TCB (as described in Appendix 1).

Factors that were studied in the sensitivity analysis are: the effect of the pH of the receiving soil, a decreasing concentration in leachate from the landfill (instead of a constant concentration) and the time frame (shorter and longer than 500 years). Another aspect studied was the effect that the increased emission of phosphate from the landfill can have on the mobility and magnitude of the ETVs of the other substances that are considered relevant to the current framework. Finally, the sensitivity analysis studied the effects of the local thickness of the unsaturated zone of the receiving soil (instead of the standard 1 metre used in accordance with the policy on building materials) and of the effects of variations in the background concentrations in groundwater'.

The sections below explain in further detail for each aspect how the model was adapted. The modifications were always used for all inorganic substances because of the possible interactions between them. Table 6.1 shows a summary of the selected parameters as well as the reason why each parameter was chosen for the sensitivity analysis.

Other factors, such as the effects of the preferential flow of substances in soil and a non-stationary water flow from the landfill in the unsaturated zone, had already been investigated and reported with reference to the setting-up of the model for the re-use of building materials, for which the same reactive transport model was used (Verschoor et al. 2006; Comans et al. 2014).

Table 6.1: Summary of the factors relevant to the sensitivity analysis and why they were chosen.

	Factor	**Why relevant?**
1	Increased DOC concentration in leachate after the period of active treatment	The concentration of DOC plays an important role in the mobility of substances. An over- or underestimation of this value can directly influence the ETVs. This parameter was chosen on the basis of the advice given by the TCB and after discussion within the project team.
2	Variation in reducing conditions under the pilot landfills	The reducing conditions under the landfill influence the mobility of certain substances (e.g. arsenic, cyanide and phosphate). It cannot be predicted how the conditions under the landfills will be after approximately ten years of active treatment. If these conditions vary too much from the modelling described in this report, the ETVs might be over- or underprotective. To determine this influence, this parameter was chosen for further analysis.
3	Variation in pH under the landfill	pH influences the mobility of certain substances. It cannot be predicted how the pH under the landfills will be after approximately ten years of active treatment. If these conditions vary from the modelling described in this report, the ETVs might be over- or underprotective. To determine this influence, this parameter was chosen for further analysis.
4	Decreasing leachate concentrations	In the current modelling a constant concentration of substances in the leachate coming from the landfill is assumed. In reality, however, the concentrations in the leachate will most likely decrease over time. How this will take place cannot be predicted. To determine the influence, an analysis is necessary.
5	Variation in time frame (100, 500 and 1000 years)	For the re-use of building materials the time window used was 100 years. This proved to be too short for this project, because many substances do not reach POC1 within this time window. Therefore, a larger time window had to be selected. To determine the influence; the sensitivity analysis studied the effect of reducing the time window from 500 to 100 years as well as that of increasing it to 1000 years.
6	Increased phosphate concentrations in leachate	Phosphate has a large influence on other substances, such as arsenic, cyanide and sulphate. It is therefore difficult to derive a reliable ETV for phosphate. However, phosphate is present in the leachate of the pilot landfills; therefore, the influence of increased phosphate concentrations in the leachate was determined.
7	Variation in thickness of unsaturated layer	In the model used to derive the ETVs the saturated zone/layer has a default thickness of 1 m in analogy with the re-use of building materials. However, in reality this thickness varies per pilot landfill. To determine the influence, this parameter was selected for analysis.
8	Variation in the natural background concentration in groundwater	During the project, it sometimes proved difficult or even impossible to derive natural background concentrations in groundwater. If no data were present, the LOQ was used to derive an alternative background concentration. To determine the influence of variations in background concentration on the ETVs this parameter was included in the sensitivity analysis.

6.1 Starting points for each parameter

1) DOC

In the original model design the DOC content in the underlying soils was based on an estimate of the naturally present DOC concentration in the soil. In its recommendations (see Appendix 1) the TCB expressly referred to the possible invalidity of the assumption that DOC in the landfill leachate would no longer cause an increased DOC concentration in the receiving soil after the period of active treatment. According to the TCB, it was more likely that even after this period there would still be a substantial amount of DOC present in the leachate, which would have an influence on the mobility of substances. The TCB also regards the fact that ammonium is still being released from the landfill at the end of the active treatment period as a sign that the mineralization of organic matter and DOC formation are still taking place. It therefore recommended that the effect of increased DOC concentrations on the ETVs be included in the sensitivity analysis and that the assumption relating to DOC be reconsidered as a result of this.

The point of special attention is therefore the possible underestimation of relatively high (compared with the receiving soil) reactive DOC emissions from the landfill. Instead of using the naturally present DOC concentration from the STONE database (the original starting point), an assumption was made about the DOC content in the leachate after the period of active treatment.

It is expected that the current DOC content in the leachate will reduce by a factor of 3–5 after approximately ten years of aeration/recirculation[5]. N-Kjeldahl also reduces by a comparable factor in this period. The leachate data from the LeachXS database also indicate a reduction of DOC in leachate by a factor of 3–5 in approximately ten years.
In the model, the 95th percentile of the DOC content in the relevant pilot landfills is divided by a factor of 4 (the average of a factor of 3–5). The new assumption for the DOC content is derived from this. Only half of this DOC content is 'reactive' (ANVM 247 project; Comans et al. 2014). This results in the following values for each landfill:

- Braambergen: 95% DOC value (852 mg/L) divided by 4 = 213 mg DOC/L; 50% of this is reactive. Humic acid consists of 50% carbon, so the value in the model is 213 mg of humic acid/L. This value is constant over time and over the depth of the soil profile.
- Kragge: the same approach generates 721 mg of humic acid/L.
- Wieringermeer: the same approach generates 173 mg of humic acid/L.

The median DOC concentrations, instead of the 95th percentile values, were also investigated (after the same corrections for degradation, and the same assumption of 50% reactive DOC for metal binding). This produced the following concentrations of humic acid:

- Braambergen: 80 mg humic acid/L. This value is constant over time and over the depth of the soil profile.
- Kragge: 264 mg humic acid/L.
- Wieringermeer: 123 mg humic acid/L.

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[5] Information gained from Mr Hans Oonk, email dated 18/01/2013, ECN.

2) Reducing conditions

The TCB indicated that it had no objection to the ('worst case') assumption that the possible immobilization of metals as a result of the precipitation of metals with sulphides was not included in the derivation of the ETVs. However, it did point out that under reducing conditions, iron oxides, to which some metals and anions (particularly arsenic) can bind, dissolve. So reducing conditions can result in both an increase and a decrease in the mobility of substances.

In order to assess the influence of reducing conditions, calculations were carried out on the basis of the dissolving of 50% of the assumed amount of iron oxides (HFO) in the receiving soil. The complete dissolution of iron oxides is improbable (only in the case of extremely low pH and extremely low redox potential).

No calculations were carried out on the basis of precipitation of (metal) sulphides because of a lack of data and a lack of validation of model calculations.

3) pH

pH is an important parameter in the model in that it is key to the mobility of substances in the soil. So the effects of increasing and decreasing the pH by half a unit compared with the value included in the STONE database were studied in the sensitivity analysis.

4) Decreasing concentrations released from the landfill

In the model, a constant concentration of substances in the leachate over 500 years is assumed. In reality, the leachate concentrations will decrease to an unknown extent over 500 years. In order to determine what influence a decrease has on the ETVs, the sensitivity analysis uses a scenario in which the emissions decrease as a function of the achieved solid substance/liquid ratio of the landfill. This decrease takes place in accordance with an exponential function (Verschoor et al. 2006):

$$C(t) = C(0) \text{ x } e^{-k \text{ x } t}$$

Here k is a substance-specific leaching rate constant (expressed in the unit day^{-1}) that is related to the substance-specific leaching constant 'kappa' (kg/l; Verschoor et al. 2006), the height of the landfill h (m), the dry bulk density of the material ρ (kg/m^3, default choice 1550 kg/m^3) and the net precipitation N (mm/year) according to Verschoor et al. (2006):

$$k\ (\text{day}^{-1}) = [\text{kappa}/h] \text{ x } [(N/365)/\rho]$$

The values for kappa are generic for waste and building materials and are derived from laboratory leaching tests carried out on waste and building materials (Verschoor et al. (2006). To estimate the effect of decreasing concentrations in leachate, the factor used is the maximum value that occurs at $POC2_{gw}$ between 0 and 500 years, divided by the environmental criterion ($=C_{max}$(0-500j)/environmental criterion). The functions are derived for the application height of the waste of the specific pilot landfills (15, 20 and 12 metres for Braambergen, Kragge and Wieringermeer, respectively[6]).

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[6] In accordance with the email by Mr H. Scharff dated 18/02/2013, ECN.

5) Time frame

The time frame within which the concentrations at $POC2_{gw}$ have to remain below the groundwater criterion is set at 500 years. The sensitivity analysis looked at the effects on the derived ETVs if this period were 100 or 1000 years. The derived emission limit values for building materials were very sensitive for several substances to the choice of 100 or 1000 years (Verschoor et al. 2006; Dijkstra et al. 2013). This was the reason that the effects of this choice were included in the sensitivity analysis.

6) Phosphate

Despite the fact that it is not possible to derive an unambiguous ETV for phosphate (see chapter 4), there was a desire amongst the project team to investigate the influence of a certain amount of PO4 in the sensitivity calculations in order to study what consequences the increased presence of phosphate could have on the ETVs of substances such as arsenic and cyanide (see Appendix 6).

However, data relating to the phosphate concentrations in the landfill leachate of the three locations are very scarce. For the Kragge landfill there were no measurement data available. For the Braambergen and Wieringermeer landfills the data are incomplete and based on total P concentrations. Total P concentrations are an overestimation of the actual amount of phosphate (as organic matter also contains P). The values are highly variable and show unrealistic extreme values up to 0.3 g P/L, as a result of which a reliable 95th percentile value cannot be established. The median mg P/L value for all the landfills is 0.34 mg P/L. For the Braambergen landfill the median value is 1.8 mg P/L, and for the Wieringermeer landfill it is 5.6 mg P/L. Ultimately a generic (average) value of 2.6 mg P/L was calculated on the basis of the above-mentioned median values. This corresponds to 8 mg PO4/L. The initial background concentration of PO4 in the receiving soil is 0.15 mg/L (corresponding to MPR_{eco}). The sensitivity analysis was carried out on the basis of these data.

7) Thickness of unsaturated layer

In the modelling, a standard unsaturated zone thickness of 1 metre under the landfill is assumed, corresponding to the approach used for the re-use of building materials (Verschoor et al. 2006). However, in practice this layer can vary in thickness. In order to determine the influence of a variation in thickness, a calculation was made using local thicknesses at the Braambergen landfill (0.14 metre) and at the Kragge landfill (2.7 metres). Because of the seepage situation in the Wieringermeer landfill, a decision was taken not to carry out this calculation for Wieringermeer.

8) Background concentrations

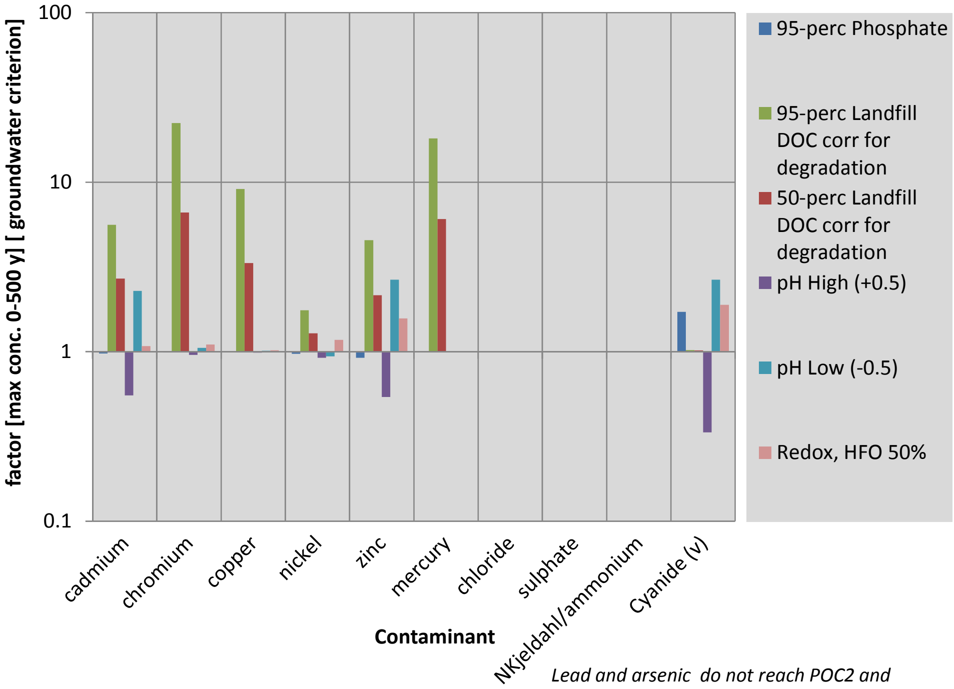
As indicated in Section 3.5, there is no uniform way of establishing the background concentration in groundwater. To establish the ETVs a decision was taken to adopt the method used for the WFD. In order to establish whether and to what extent background concentration has an influence on the ETVs, the effects of a high and a low background concentration were investigated. For the Wieringermeer landfill, no calculations were carried out.

6.2 Results of sensitivity analysis

In order to obtain an overview of the influence of the different parameters on the sensitivity analysis the results are expressed as:

Difference factor = (concentration at $POC2_{gw}$ *after 500 years)/(environmental criterion* $POC2_{gw}$*)*

The advantage of this manner of presentation is that it immediately becomes clear how large the possible effect of a parameter variation is on the ETVs. The demonstrated effect is no more than an indication, however, because in order to derive an ETV several iterations of the model are necessary. The results of the sensitivity analysis are shown for each parameter for the different landfills in Figures 6.1–6.3. Figures 6.1a, 6.2a and 6.3a show the influence of chemical parameters (DOC, redox, PO4 in the leachate) and Figures 6.1b, 6.2b and 6.3b show the influence of physical parameters and other choices (such as a decreasing concentration in the leachate, a shorter or longer time frame, and variation in background concentration and the thickness of the unsaturated zone).



Lead and arsenic do not reach POC2 and cannot be presented in this figure.

Figure 6.1a: Difference factors for alternative chemical properties of the emissions coming from the landfill and the receiving soil for the Braambergen pilot landfill.

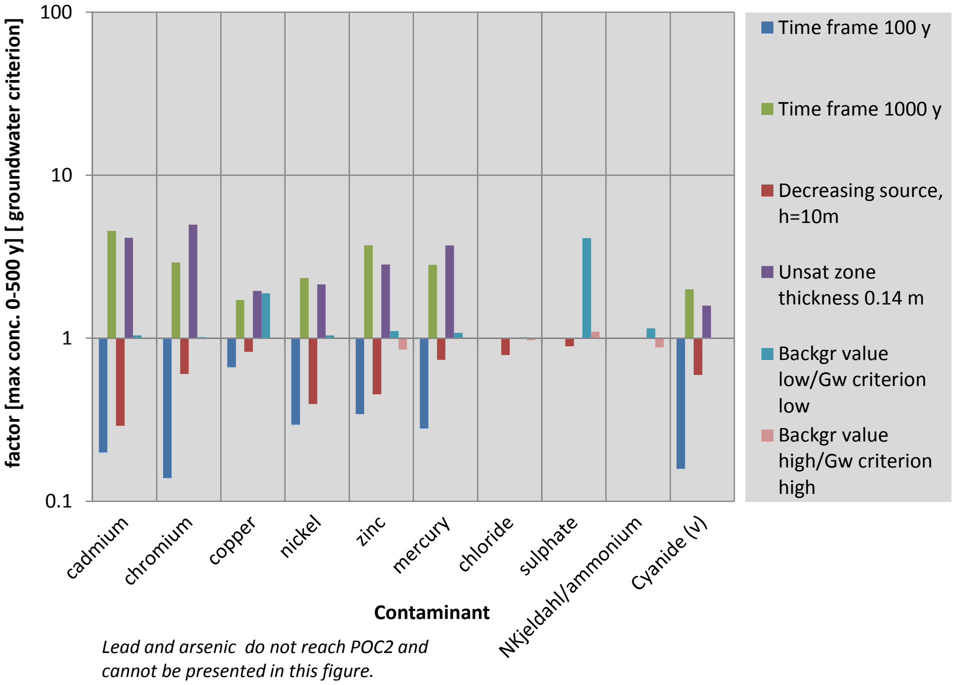


Figure 6.1b: Difference factors for alternative physical properties and other decisions for the Braambergen pilot landfill.

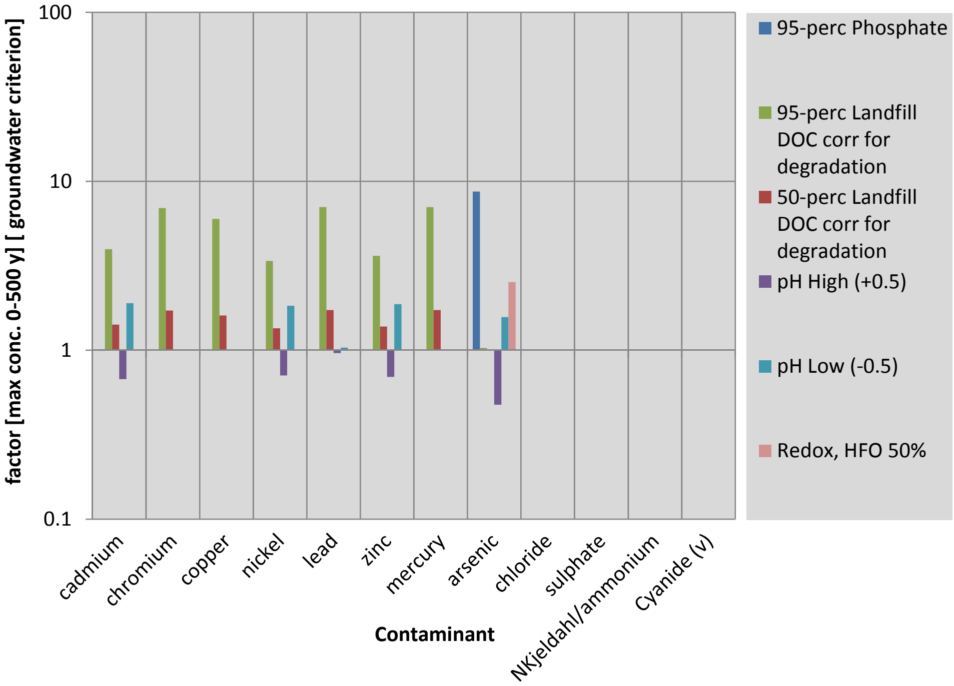


Figure 6.2a: Difference factors for alternative chemical properties of the emissions coming from the landfill and the receiving soil for the Kragge pilot landfill.

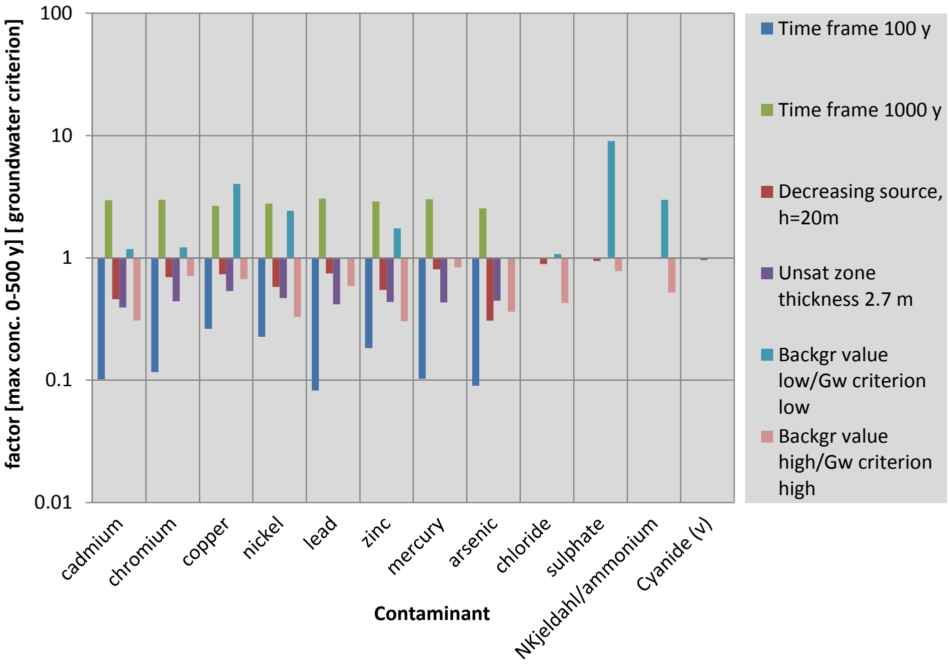
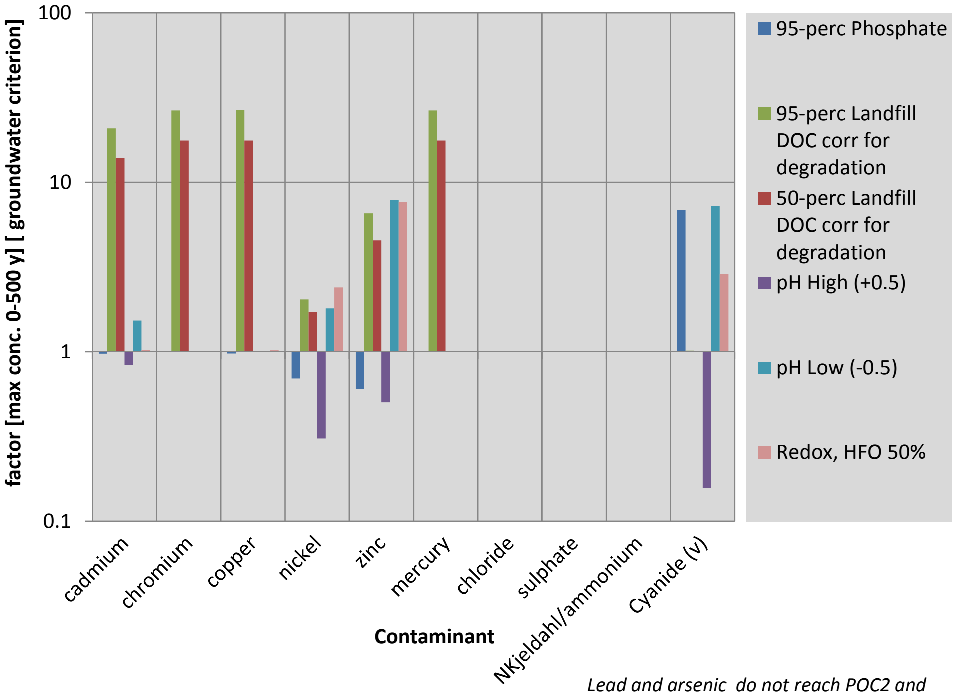
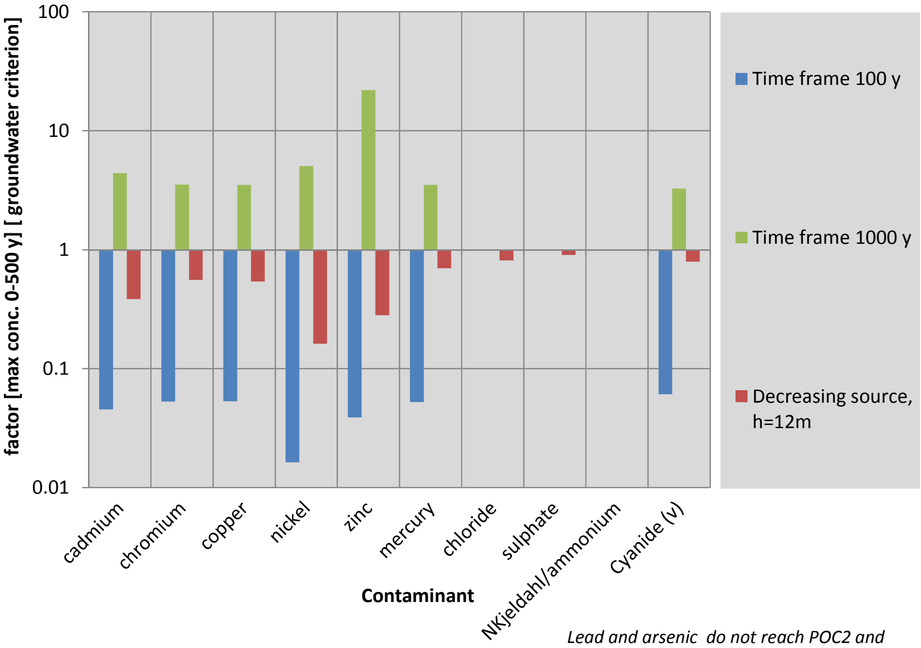


Figure 6.2b: Difference factors for alternative physical properties and other decisions for the Kragge pilot landfill.



Lead and arsenic do not reach POC2 and cannot be presented in this figure.

Figure 6.3a: Difference factors for alternative chemical properties of the emissions coming from the landfill and the receiving soil for the Wieringermeer pilot landfill.



Lead and arsenic do not reach POC2 and cannot be presented in this figure.

Figure 6.3b: Difference factors for alternative physical properties and other decisions for the Wieringermeer pilot landfill.

The following conclusions can be drawn:

- Highly soluble salts (chloride, sulphate and ammonium) are not sensitive to variations in chemical factors and physical factors such as time frame, thickness of the unsaturated layer and reducing vs. constant concentration.
- Highly soluble salts are sensitive to a variation in background concentrations.
- Variation in the time frame (from 500 years to 100 years and from 500 years to 1000 years) has the greatest influence on the concentrations at $POC2_{gw}$ for moderately mobile metals. This is in accordance with the findings of the derivation of emission limit values for building materials (with time frame = 100 years).
- Generally, by far the most influential of the chemical factors is increased DOC content, followed by pH. Redox (reduced Fe oxide content) and phosphate in the leachate are important for several anions (particularly cyanide in the Braambergen landfill and arsenic in the Wieringermeer and Kragge landfills).
- Generally, the most influential physical factors are the layer thickness and the choice of a decreasing versus a constant concentration of the leachate.

The most important effects observed on the mobility of individual substances are described below for each specific choice/parameter.

1) <u>Effect of increased DOC content</u>

A high DOC content has a strongly mobilizing effect on metals. This effect is strongest on the metals that have the highest affinity for binding to organic matter (copper, chromium and lead) and is less strong on the more weakly binding metals (zinc, cadmium and nickel). As a result of the sensitivity analysis and the comments by the TCB, the assumed DOC concentration in the emissions from the landfills was modified in the model to match the concentration that may be expected after the period of active treatment.

2) Effect of (partial) dissolution of iron oxides under reducing conditions
In addition to the effect on nickel and zinc at the Wieringermeer pilot landfill, the reduction of iron hydroxides mainly has an effect on the arsenic and cyanide anions, the sorption of which is greatly dependent on the presence of iron oxides. The effect varies per location and is partly dependent on the amount of iron oxide present in the receiving soil and the concentrations of the competitive substances that bind to this. The sensitivity analysis indicated that a reduced amount of iron oxide (50%) does not have an effect on the arrival of arsenic at $POC2_{gw}$ (at concentrations of up to 190 µg/L) within the time frame of 500 years at Braambergen. In the seepage scenario at Wieringermeer, a calculation was carried out with 50% and 25% of the amount of iron oxide (which seldom occurs in practice). This had no effect on the arrival at $POC1_{sw}$ of arsenic in that scenario. This finding is consistent with the observation that no effect could be observed on the arrival of arsenic at $POC1_{gw}$ in the infiltration scenario at Wieringermeer, either.

The conditions under the landfill are very important for the behaviour of arsenic in particular. If the conditions are aerobic (oxidative), arsenic will adsorb on iron(hydr)oxides and no large concentrations in groundwater are to be expected. However, if the situation becomes more anaerobic (reducing conditions), arsenic will become more mobile (depending on the degree of the reducing conditions and available amount of iron(hydr)oxides). The behaviour of arsenic is therefore uncertain, and the derived ETV should be interpreted with caution.

Therefore, it is advisable to monitor the conditions under the landfill regularly and to determine whether arsenic reaches POC2gw or not. For modelling purposes, a decision was taken to retain the original input.

3) Effect of variation in pH
The pHs of the soil at the three pilot landfills as described in the STONE database are pH 6.9–7.4 for Braambergen, pH 7.6–7.8 for Wieringermeer and pH 4.5–4.9 for Kragge. Figures 6.1–6.3 show the effects if a pH of a half-unit higher or lower is assumed. A higher pH means later arrival of metal cations in the groundwater at $POC2_{gw}$; a lower pH means earlier arrival at $POC2_{gw}$. With the low soil pH at the Kragge landfill, the arsenic anion arrives earlier, with a further reduction of the pH (as a result of protonation/charge neutralization of the arsenic anion). With the higher soil pHs of the Braambergen and Wieringermeer landfills, arsenic is insensitive to a half pH unit increase or decrease; in these cases, arsenic does not arrive in the groundwater at $POC2_{gw}$. The arsenic concentrations are, however, so low that they cannot be detected (Wieringermeer) or the effect cannot be seen due to the bigger effect of dilution with background concentrations in groundwater (Braambergen). Contrary to expectations, cyanide also arrives earlier at $POC2_{gw}$ with a higher pH (Braambergen, Wieringermeer), possibly as a result of interaction with phosphate, which binds competitively to iron oxide. As it is difficult to predict what the pH under the landfill will be after the period of active treatment and given the relatively small influence on the ETV, a decision was taken not to revise the assumption for pH.

4) Decreasing leachate concentrations
The effect of decreasing leachate concentrations coming from the landfill would generally be more tolerant ETVs. The results of the sensitivity analysis show that

the effects are largest for substances with moderate mobility, and smallest for both immobile substances (e.g. arsenic and lead) and highly mobile substances (e.g. chloride). The same effect is small or absent for immobile substances, because these substances do not reach $POC2_{gw}$ at all within the time frame of 500 years, and therefore the effect on concentrations at $POC2_{gw}$ is absent and cannot be quantified.

The effect on predicted concentrations at $POC2_{gw}$ for highly mobile substances is small because these substances reach the groundwater very fast, while their concentration is scarcely influenced by sorption processes. It should be stressed that it is the peak concentration at $POC2_{gw}$ within the time frame of 500 years that determines the value of the ETV. The effect of constant versus decreasing leachate concentrations on mobile substances is illustrated in Figure 6.4 (chloride in the Braambergen pilot). In both scenarios the starting concentrations are the same, and the dilution between $POC1_{gw}$ and $POC2_{gw}$ amounts to a factor of about 5. The peak concentration at $POC2_{gw}$ in the case of decreasing leachate concentrations (right-hand diagram) is only slightly lower (factor 0.8) than the peak concentration in the case of constant leachate concentration (left-hand diagram). This small effect is caused by dispersion.

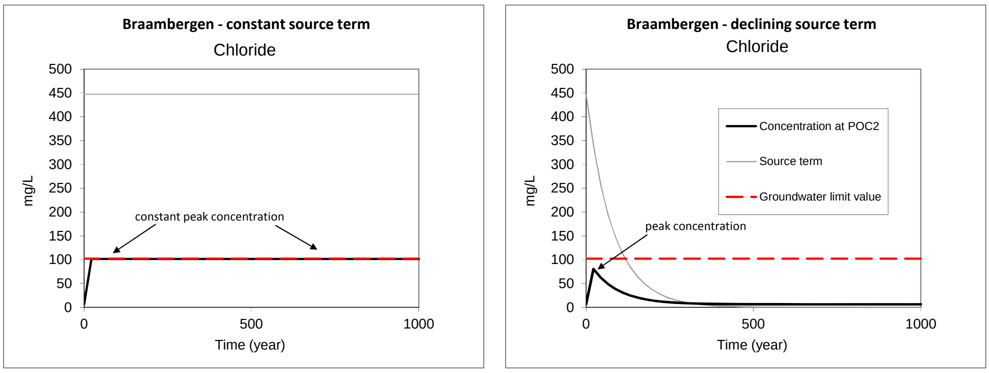


Figure 6.4: Illustration that clarifies why peak concentrations of mobile substances at $POC2_{gw}$ are not very different whether constant leachate concentrations (left diagram) or decreasing leachate concentrations (right diagram) are assumed. The peak concentrations at $POC2_{gw}$ within the period 0–500 years determine the ETV.

The conclusion is that the effects of decreasing leachate concentrations on concentrations at $POC2_{gw}$ is largest for moderately mobile substances, but small for either the immobile or highly mobile substances. The effects are, however, relatively small compared with the effects of other factors. It should be noted that an uncertainty factor in the rate at which the concentration of leachate is predicted to decline is whether the kappa values (coefficient for the decline) used are representative of landfills during the entire phase of aftercare (period after active treatment). A decision was therefore taken to retain the original input for the model.

5) Effect of the chosen time frame (100/500/1000 years)

Variation in the time frame (from 500 years to 100 years and from 500 to 1000 years) has a major influence on the concentration at $POC2_{gw}$ for the group of moderately mobile substances, including metals. At the start of the project a decision was taken to take the ‘binding’ of substances in the soil into account in the determination of the ETVs, as was done for building materials (Verschoor et al. 2006). Including this attenuation in the soil as a result of adsorption processes implies that the ETVs are also sensitive to the time frame within which concentrations at $POC2_{gw}$ are tested against the groundwater criterion. After all, the time frame has an influence on the ETVs as substances can arrive at $POC2_{gw}$ either before or after the chosen time frame. If a substance does not arrive completely at $POC2_{gw}$ within 500 years, the concentration in the model is increased in such a way that the substance does arrive after 500 years at $POC2_{gw}$ in order to derive an ETV for that substance.

With substances that reach $POC2_{gw}$ quickly (highly soluble salts such as chloride and sulphate) or that do not reach $POC2_{gw}$ at all (arsenic in the Wieringermeer pilot landfill), the choice of time frame has no influence. This can be explained by the fact that a lot of moderately mobile substances have still not reached a maximum concentration after 500 years. As a result of this, the concentrations of these substances will continue to increase after the time frame of 500 years. If the time frame becomes longer (1000 years), ETVs are lower; if the time frame becomes shorter (100 years), ETVs are higher.

The reasons for the choice of 500 years are set out in Section 3.4.4; the sensitivity analysis does not give any reason to deviate from this choice.

6) Effect of increased concentration of phosphate in the leachate

An increased concentration of phosphate in the leachate has an effect, mainly on cyanide (Braambergen and Wieringermeer) and arsenic (Kragge). In the Kragge landfill scenario, arsenic is stripped of the (iron oxide) adsorption surface by phosphate, as a result of which the arsenic becomes more mobile. Some metals are more strongly bound in the presence of phosphate; this is because of complex interactions between substances in the model.

Phosphate has no perceptible effect on arsenic (with concentrations up to 190 ug/L) in the soils of the Braambergen and Wieringermeer landfills.

Although phosphate poses no immediate health threat in groundwater, it does stimulate eutrophication in surface waterPhosphate is particularly important for the Wieringermeer landfill because part of the leachate infiltrates the nearby surface water (ditch). It is therefore desirable to derive an ETV for phosphate in surface water. However, considering the difficulties with the modelling of phosphate (see Section 4.3.5), it is impossible to do so.

Instead, an additional sensitivity analysis was carried out in the surface water scenario of the Wieringermeer landfill to the extremely high value of 150 mg/L phosphate (ten times the 95th percentile value, based on data from the landfill database), whereby both phosphate and arsenic did not arrive at $POC1_{sw}$ within 500 years (after 20 m of soil passage). Both substances arrive at $POC1_{sw}$ after the 500-year period in this scenario.

However, this does not mean that concentrations of 150 mg/L do not pose any risk and can be used as an indicative ETV, because infiltration into groundwater still takes place. The hydrological modelling by Van Someren (2013) indicated that within 750 m of the Wieringermeer landfill the leachate in groundwater will reach surface water. In this respect, a concentration of 150 mg/L is too high, because binding takes place only over 2 m in the infiltration scenario and there is no dilution in the saturated zone due to the seepage.

Based on the results of the sensitivity analysis an increased concentration of phosphate in the leachate coming from the landfill was included in the model (see Appendix 6).

7) Effect of the (site-specific) thickness of the unsaturated zone in the receiving soil

The effects of a thicker (Kragge landfill) and thinner (Braambergen landfill) unsaturated layer in the receiving soil are in accordance with expectations. At Kragge (2.7 instead of 1 m), all the reactive substances arrive later at $POC2_{gw}$, whereas at Braambergen (0.14 instead of 1 m), the opposite occurs. The retardation of the substances increases along with the thickness of the receiving unsaturated soil layer. However, the effects of variation in the thickness of the soil layer are small compared with the effects of other parameters. Therefore, the original input of 1 metre of unsaturated zone was retained.

8) Variation in natural background concentration

Variation in natural background concentration shows results only for Kragge. Especially highly soluble salts are sensitive to variations in background concentration. This is in accordance with expectations, because due to the absence of binding, high or low background concentrations directly influence the ETVs. Because of the relatively small effect of background concentration and the absence of information on this, the original input was retained.

6.3 Choices resulting from the sensitivity analysis

The outcome of the sensitivity analysis provided reasons for making alternative choices for two of the parameters in the computational model, resulting in the model as described in Chapter 4.

DOC

As a result of the sensitivity analysis and the comments made by the TCB, the DOC concentration was modified from the original assumption of ‘natural’ DOC concentration in the soil to the DOC concentrations that can be expected in the leachate after of the period of active treatment. In this case, the median DOC concentration corrected for degradation during this treatment was chosen.

Phosphate

The project team decided to include an increased concentration of phosphate as a model input. This concentration is 0.13 mg P/L, equal to 0.4 mg PO4/L. The value used initially was considered to be too ‘worst case’ (see Appendix 6).

The other factors that were studied in the sensitivity analysis did not give reasons to modify the original choices in the definitive calculations of the ETVs.

7 Reflection on model and results

7.1 Reflection on assumptions and model principles

The calculation of the transport of substances from the landfill in the groundwater derives from a model-based representation of reality. The purpose of the modelling is to derive sufficiently protective ETVs, based on a number of generic concepts and assumptions on the one hand, and available site-specific information on the other hand. Hence, the modelling approach should not be seen as an attempt to provide a detailed chemical, physical and hydrological representation of reality. It should be noted that all models are a simplification of reality. The output from a model is largely determined by the model concept, the assumptions and the model input.

For this particular project, assumptions were made on the basis of the current understanding of the pilot landfills, state-of-the-art techniques for the modelling of emissions and consistency with existing policy frameworks, such as the framework for the re-use of building materials and large soil applications. Insights and assumptions may change over time as a result of increased knowledge and changed perceptions of risks. In view of this, it is important that the influence of assumptions on the results be discussed below:

1. It is assumed that the bottom liner of the landfill fails completely and that the emission coming from the landfill remains constant in both volume and concentration. It is also assumed that no leachate is discharged via the leachate drains in the landfill. In reality, the bottom liner will still be functional after the period of active treatment. However, the life span of the bottom liner is not fully known. Research is being carried out into this by the AKS[7] working group (formerly ENS). For the period shortly after the active treatment, the assumption relating to the absence of the bottom liner will produce an overestimation of the amount of leachate coming from the landfill. In the long term, the concentration in the leachate will decrease as a result of further stabilization of the waste. Therefore, this assumption is considered to be more stringent with respect to reality.
2. In practice, it is possible that a combination bottom seal (consisting of both a mineral layer and a liner) is present at the landfills. In the model, the mineral component of the combination bottom seal is regarded as inert. In reality, depending on its composition, this layer will have a considerable decelerating effect on the transport of the substances in the soil and groundwater. Therefore, the assumption that the mineral layer is inert is considered to be more stringent with respect to reality. If, however, no mineral layer is present at the pilot landfills, this assumption corresponds with reality. Within this project it has not been investigated whether or not a mineral layer is present at the pilot landfills.
3. In the modelling, a top cover is not taken into account. In reality, a top cover of approximately 1 m of clean soil will probably be applied. This top cover has an influence on the extent of infiltration of precipitation into the landfill. If the top cover has a high permeability, the decision not to take a top cover into account will have minimum consequences with respect to

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[7] The advisory board on landfill policy (Advieskamer stortbesluit), formerly called Expertise Network landfill policy (ENS), focuses on technical aspects of landfills and handles questions from the government and from the landfill sector.

reality. However, if the top cover has a low permeability (and therefore little infiltration takes place into the landfill), the decision not to take the top cover into account will turn out to be a conservative one in respect to reality.

4. The geochemical transport model does not take into account the natural attenuation of organic substances under the landfill. There are two reasons for this. First, it is expected that the degradation of organic substances will take place largely in the landfill during the period of active treatment. After the completion of this treatment no high concentrations of organic substances are expected in the leachate. During the period of active treatment it should become clear whether this expectation is correct. Second, specific knowledge about the conditions under the landfill is required for assessing the attenuation of organic substances. This knowledge is not available. Not including natural attenuation in the model is most likely more stringent with respect to reality, because a certain amount of degradation will take place in practice. It is, however, difficult to predict the contribution of this degradation.
5. Only dilution of the substances in the groundwater between $POC1_{gw}$ and $POC2_{gw}$ was taken into account. In reality, the substances will bind to soil particles, just as they do between POC0 and $POC1_{gw.}$ Organic substances can possibly degrade in other, either more or less hazardous degradation products. In reality, the transport of substances in the direction of $POC2_{gw}$ will be slower because of the binding. In order to be able to take this into account, specific knowledge of the soil structure in the saturated zone is necessary. This knowledge and information are lacking, even in the STONE database. The dilution in the saturated zone is assumed to take place across the whole thickness of the aquifer under the landfill. Assuming a relatively shallow aquifer (<10 metres deep) and a time frame of 500 years, this assumption can be defended. It is likely that substances from the landfill can mix in 500 years across the entire thickness of the aquifer. However, for thicker aquifers (>10 m) this assumption needs to be reconsidered. Therefore, the assumption that dilution takes place over the total thickness of the aquifer is less stringent with respect reality in the short term, but realistic in the long term.
6. The geochemical transport model does not take into account the presence of reducing conditions in the saturated zone. The current models are still insufficiently validated to be able to quantitatively predict the effect of sulphide precipitation in particular. For the unsaturated zone of the receiving soil, especially after the period of active treatment, it is plausible to assume aerobic conditions. If reducing conditions occur, several substances will behave differently. Under strongly reducing conditions sulphides can precipitate and the transport of metals that precipitate with these sulphides can be delayed. Whether this occurs depends on different factors, including the availability of sufficient sulphate in the immediate environment of the landfill or in the leachate. Another effect of reducing conditions is the dissolution of iron oxide surfaces, to which some metals and anions such as arsenic can bind. As a result of this, the transport of these metals is accelerated. Oxidizing or reducing conditions also have an effect on the degradation of organic substances. Depending on local conditions (for example, the formation of sulphides), these processes can result in the acceleration or deceleration of the dispersion of specific substances in groundwater. So, the choice of not taking into account of reducing conditions is either more tolerable or more stringent with respect to reality depending on the substance considerd.

7. Density flow is not explicitly taken into account in the modelling. Density flow can accelerate the dispersion of substances in reality. Based on the regulations relating to the landfilling of materials in the Decree on Landfill and Soil Protection, the concentrations coming from the landfill after the period of active treatment are not expected to be so high that this process will play a role (for example, organic substances and metals will be flushed/bound or degraded). It is recommended that, during monitoring, substances for which density flows are relevant are taken into account, such as mineral oil. For chloride, the concentrations are too low to allow density flow to occur. According to Bot (2011), differences in density of chloride in groundwater become significant in the flow modelling if the chloride content rises above 5000 mg/L. This is not the case at any of the pilot landfills. As salts are transported unimpeded in the soil and arrive at $POC2_{gw}$ well within the chosen time frame of 500 years, the inclusion of density flow would have no influence on the calculated ETVs of these substances. Therefore, the decision not to take into account density flows is neutral in respect to reality.
8. The spatial discretization (i.e. subdivision into one layer of 1 metre) of the unsaturated zone is different from the discretization used for the derivation of emission limit values for the re-use of building materials and large soil applications in the Soil Quality Decree (Verschoor et al. 2006). In the latter work, the unsaturated zone was subdivided into ten layers, each of 0.1 metre thickness, as in the STONE database. The reasons for deviating from this approach and choosing a homogeneous layer of 1 metre thickness originate from uncertainty and sensitivity analyses that were performed following advice from the TCB in 2006, and are reported in Comans et al. (2014). The TCB pointed out in its advice of 2006 that the model approach with a fine grid (i.e. layers of 0.1 metre) used in Verschoor et al. (2006) did not sufficiently reflect the potential effects of preferential flow. Also, the TCB recommended further development of the model such that a solution would be found that enabled the derivation of the emission limit values that are less sensitive to the chosen time frame (in the Soil Quality Decree, the time frame was set at 100 years; see chapter 2).
   An approach based on a homogeneous layer of 1 metre thickness results in larger numerical dispersion of substances (mixing), which is more conservative with respect to the likely occurrence of preferential flow paths, their change over time, and the mixing that occurs due to bioturbation over long time scales (Comans et al. 2014). Because of the homogeneous layer, some substances arrive earlier at POC1 in groundwater, while still the complete reactivity of the unsaturated zone is taken into account. Coarser spatial discretization has a relatively small effect on concentration levels (see the comparison of different model approaches in Comans et al. 2014).
9. In addition, assuming a homogenous layer of 1 metre thick in the unsaturated zone makes the results less dependent on the chosen time frame of 500 years. Instead, when the unsaturated zone is subdivided into ten layers of 0.1 metre, as in the approach used for building materials (Verschoor et al. 2006), concentration fronts become very sharp, and the model results become extremely sensitive for the exact moment of arrival at $POC2_{gw}$ (see discussion in Verschoor et al. 2006, and in Comans et al. 2014). Due to the larger compartiments, the transport of substances is described more diffuse dispersely (spread out) and results become less sensitive for to the time frame.
10. With respect to the upper metre of the saturated zone, it should be noted that concentrations were also mixed in the derivation of emission limit values for building materials using the PEARL and ORCHESTRA models (Verschoor et al. 2006). In Verschoor et al. (2006) concentrations in the

lower ten layers in the model were explicitly mixed. The difference between explicit mixing of these layers and a homogeneous layer of 1 m thickness has been shown to be very small (Comans et al. 2014).

11. Diffusion as a transport mechanism is not taken into account in the transport model, because its contribution to the overall (fixed) transport velocity (i.e. a pore water velocity of approximately 1 metre per year, due to advection of substances) is low. Also, the contribution to the concentration front spreading is negligible, considering the already substantial numerical dispersion due to the homogenization of the unsaturated zone.
12. The fixed downward infiltration of 300 mm/y and the saturated pore space in the unsaturated zone is a simplification of reality. However, in Verschoor et al.(2006) a direct comparison was (similar to the scenario used in this work) between calculated transport of linear sorbing substances under unsaturated, non-stationary conditions in a sandy soil (with the model PEARL-SWAP) and a scenario using stationary saturated conditions (with ORCHESTRA), using the same fine grid (layers of 0.1 m) in both models and for a period of 100 years. Under those conditions, the differences appeared very small (see figure 4.6 in Verschoor et al. 2006). A similar comparison was made with ORCHESTRA coupled to SWAP for unsaturated, non-stationary transport and a scenario using stationary saturated conditions, both in combination with non-linear sorption processes (Comans et al. 2014). The differences were again small, suggesting that adding complexity to the transport approach is not necessary. For clay soils, the fixed downward infiltration rate of 300 mm/y may be conservative, as, in reality, water is flows around semi-permeable layers.
13. The model does not take into account a 'first flush' effect when the bottom liner actually fails. The result of such failure could be that locally a high flux of leachate escapes to the groundwater. The moment at which the bottom liner starts to fail is not known. However, for the position of the concentration fronts of the substances after, e.g., a period of 100 years, there is in principle no difference in concentration of the ETVs if the water that accumulates during those 100 years infiltrates gradually.

Table 7.1: Qualification of the assumptions made to derive the ETVs.

Assumption	**Assumed qualification**
Constant concentration in leachate compared with complete failure of bottom liner and no influence of leachate drains on amount of leachate	More stringent with respect to reality
Mineral layer of bottom liner is inert	Neutral/more stringent with respect to reality
Absence of top cover	Neutral*/more stringent with respect to reality
No natural attenuation of organic substances	More stringent with respect to reality
Increased concentration of DOC in the leachate	Neutral with respect to reality
Dilution within the total thickness of the aquifer	More tolerant with respect to reality in the short term; realistic with respect to reality in the long term
Binding in unsaturated zone (POC0–POC1$_{gw}$) taken into account Binding in the saturated zone (POC1$_{gw}$–POC2$_{gw}$) not taken into account	Neutral with respect to reality More stringent with respect to reality
No influence of reducing conditions (dissolving of iron oxides)	No generic qualification can be given because this is substance- and location-dependent and can be either more tolerant or more stringent with respect to reality
Density flow not taken into account	Neutral with respect to reality
Heterogeneous flow of water and mixing of the unsaturated zone over long time scales implicitly taken into account (homogeneous unsaturated zone versus discrete layers in the reactive transport model)	Realistic with respect to reality in the long term
Fixed downward infiltration of 300 mm/y instead of non-stationary unsaturated flow	Neutral with respect to reality; for clay soils possibly more stringent
Diffusion not taken into account in combination with fixed advective flow and large numerical dispersion	Neutral with respect to reality
A first flush effect (local failure of liner) not taken into account	Neutral with respect to reality

** Neutral means no effect on results*

7.2 Reflection on the results

7.2.1 High emission testing values in the leachate

The ETVs are expressed in terms of concentration (μg/L) in the leachate (in contrast to the emission limit values for building materials, which are expressed in mg/kg dry substance at L/S 10).

A consequence of expressing the ETVs in terms of concentrations in the leachate is that there may be a tendency to compare these directly with protection criteria from other policy frameworks (e.g. protection criteria in (ground)water) that are expressed in the same units. When doing so it may seem that the ETVs derived in this report are less strict than the protection criteria for groundwater. It is important to realize that in this project, as in the normative guidelines for the re-use of building materials and large-scale soil applications, a reduction in concentrations coming from the landfills as a result of binding and/or dispersion in the receiving soil is taken into account. Within the set time frame at $POC2_{gw}$ the environmental criterion in groundwater is complied with. In other words, at POC0 the concentrations are relatively high, whereas the concentrations at $POC2_{gw}$ comply with the environmental criteria for groundwater set in this report.

As a comparison, the emission limit values (POC0) for building materials also often go far above the environmental protection criteria for groundwater if expressed in mg/L. But these too comply with the environmental requirements at the set POC1 within the set time frame.

Finally, it should be noted that the environmental protection criteria for groundwater were established for a completely different purpose from that of the ETVs in this report. The environmental protection criteria for groundwater are used to determine the need to remediate, whilst in this project the ETVs are used to determine what emissions from landfills are acceptable in terms of protection of the groundwater.

7.2.2 Lead and arsenic

For both the Braambergen and Wieringermeer pilot landfills (in the case of both infiltration in groundwater and leaching to surface water) the derived ETVs for lead are high, with concentrations of 60,000 µg/L and 25,000 µg/L for groundwater for the Braambergen and Wieringermeer landfills, respectively, and 11,000 µg/L for surface water at the Wieringermeer landfill. Due to strong binding in soil, lead did not arrive at the designated POC when using the environmental criterion for groundwater or surface water. The use of the environmental criterion MPA_{eco} for soil was investigated as an alternative, which was also done within the framework of the re-use of building materials. This resulted in the concentrations mentioned above. Such high ETVs are not desirable and unnecessary for the success of the introduction of sustainable landfill management at the pilot landfills. The ministry therefore decided to lower these values. At the request of the ministry, alternatives were discussed in the working group and it was decided that for both landfills the same ETVs would be used as for the Kragge landfill (130 µg/l). This value was chosen because the ETV for the Kragge pilot landfill was based on a criterion for groundwater and concentrations that did reach the designated $POC2_{gw}$.

To a lesser extent the same situation applies to the ETV for arsenic at Braambergen and Wieringermeer. When considering the site-specific background concentrations of 1 (being an LOQ due to insufficient data) and 17.0 µg/L, respectively, an ETV of 190 µg/L can be considered high. Still, concentrations of 190 µg/L in groundwater can occur under natural conditions in The Netherlands.

As discussed in Section 6.2, the conditions under the landfill are very important for arsenic. In the default modelling, arsenic would not arrive at $POC2_{gw}$ within 500 years. So the ETV was derived from the environmental criterion for soil as an alternative. This resulted in the ETV of 190 µg/L. However, if in the future the conditions become anaerobic, arsenic will become more mobile and the ETV of 190 µg/L will be insufficient and under-protective. At the moment it is difficult to predict how the conditions under the landfill will develop.

The sensitivity analysis showed that arsenic would not arrive at POC2_{gw} within 500 years under worst-case conditions (a maximum of 50% of the expected binding places for arsenic). Therefore, the calculated ETVs for the Braambergen and Wieringermeer pilots are considered to be protective for the time being. However, it is advisable to determine the conditions in the leachate after the period of active treatment. If it turns out that the situation is more anaerobic than assumed in the current modelling, a revision of the ETV for arsenic will be necessary to comply with the environmental criterion at POC2_{gw}.

7.2.3 Emission testing values below the LOQ

Several of the calculated ETVs for the organic substances are well below the LOQ for waste water as reported by three randomly chosen analytical laboratories. The relevant substances are:

- Mineral oil fractions both aliphatic and aromatic
- Vinylchloride
- Dichloromethane
- 1,1 dichloroethane (only for the Wieringermeer landfill)
- 1,1 and 1,2 dichloroethene
- 1,1,1 and 1,1,2 trichloroethane
- Tetrachloromethane
- Tetrachloroethene
- Naphthalene (only Kragge and Wieringermeer)
- Anthracene (only Braambergen and Wieringermeer)
- Phenanthrene (only Wieringermeer)
- Fluoranthene (only Wieringermeer)
- Chrysene (only Wieringermeer)
- Beno(a)anthracene
- Benzo(b)pyrene
- Benzo(k)fluoranthene
- Benzo(ghi)perylene
- Indeno(1,2,3cd)-pyrene.

In practice, a policy decision has to be made to overcome this problem. A temporary solution (also used in the policy on groundwater and soil) is to use the current LOQ of the laboratories. If in the future the LOQs of the laboratories are lowered as a result of technical improvements, it might be possible to enforce the calculated ETVs. This decision will have to be made at the end of the period of active treatment. If, however, the LOQs from the laboratories are still not sufficiently low, it is recommended to use the best LOQs available at that time as temporary ETVs. This report includes the current LOQs for the sake of completeness (see Table 7.2). How these LOQs should be used after the period of active treatment is a policy decision.

Table 7.2: Overview of the current LOQs for wastewater based on information given by three randomly selected analytical laboratories.

Substance	**LOQ waste water (µg/L)**
Mineral oil fractions aliphatic	
EC5-EC6	<10
EC6-EC8	<10
EC8-EC10	<10
EC10-EC12	<10
EC12-EC16	<10
EC16-EC21	<10
Mineral oil fractions aromatic	
EC5-EC7	<10
EC7-EC8	<10
EC8-EC10	<10
EC10-EC12	<10
EC12-EC16	<10
EC16-EC21	<10
EC21-EC35	<10
VOX	
Vinylchloride	<0.2
Dichloromethane	<0.2
1,1 dichloroethane	<0.1
1,1 dichloroethene	<0.1
1,2 dichloroethene (cis,trans)	<0.1
1,1,1 trichloroethane	<0.1
1,1,2 trichloroethane	<0.1
Tetrachloromethane (tetra)	<0.1
Tetrachloroethene (per)	<0.1
PAH	
Naftalene	<0.05
Phenantrene	<0.01
Antracene	<0.01
Fluoranthene	<0.01
Chrysene	<0.01
Benzo(a)antracene	<0.01
Benzo(a)pyrene	<0.01
Benzo(k)-fluoranthene	<0.01
Indeno(1,2,3cd)-pyrene	<0.01
Benzo(ghi)perylene	<0.01

8 Conclusion & recommendations

8.1 Conclusion

The modelling described in this report resulted in the ETVs presented in Table 8.1. These values can be used after the period of active treatment at the landfills to determine whether emission reduction was successful. Substances with an * require special attention, which is explained in the corresponding footnotes.

Several of the calculated ETVs for the organic substances are well below the LOQs for wastewater. A policy decision has to be made to overcome this problem. A temporary solution (also used in the policy on groundwater and soil) is to use the LOQs of the laboratories. If in the future the LOQs of the laboratories is lowered as a result of technical improvements, it might be possible to enforce the calculated ETVs.

Table 8.1: Proposed ETVs for Braambergen, Kragge and Wieringermeer pilot landfills.

Substance	Braambergen pilot landfill	Kragge pilot landfill	Wieringermeer pilot landfill
Inorganic substances (µg/L)			
Arsenic	190	100	190
Cadmium	6.4	3.6	1.3
Chromium	210	140	37
Copper	50	64	19
Mercury	5.8	4.1	1
Lead	60,000*	130	11,000*
Nickel	21	47	21
Zinc	160	120	39
Free cyanides	61	6.8	35
Macroparameters (mg/L)			
Chloride (mg/L)	450	160	2400
N-Kjeldahl/ ammonium (mg/L)	1.8**	1.1**	50
Sulphaate (mg/L)	700	200	1400
Phosphate (mg/L)	n.a.	n.a.	***
Organic substances(µg/L)			
Mineral oil aliphatic (µg/L)			
EC5-EC6	0.8	0.17	0.17
EC6-EC8	0.37	0.039	0.039
EC8-EC10	0.047	0.005	0.01
EC10-EC12	0.00127	0.00127	0.0025
EC12-EC16	0.00071	0.00071	0.0014
EC16-EC21	-	-	-
Mineral oil aromatic (µg/L)			
EC5-EC7	4.7	1.4	1.2
EC7-EC8	3.9	2.3	0.83
EC8-EC10	2.6	1.5	0.55
EC10-EC12	1.5	0.87	0.32
EC12-EC16	1.3	0.38	0.28
EC16-EC21	0.36	0.21	0.076

Substance	Braambergen pilot landfill	Kragge pilot landfill	Wieringermeer pilot landfill
EC21-EC35	0.06	0.035	0.0064
Mineral oil sum EC10-EC40	470	270	100
VOX (µg/L)			
Vinylchloride	0.047	0.014	0.01
Dichloromethane	0.047	0.014	0.01
1,1 dichloroethane	4.7	1.4	1
1,2 dichloroethane	14	4.1	3
1,1 dichloroethene	0.047	0.014	0.01
1,2 dichloroethene (cis,trans)	0.047	0.014	0.01
Dichloropropane (1,2)	3.8	1.1	0.8
Dichloropropane (1,3)	3.8	1.1	0.8
Trichloromethane (chloroform)	4.7	1.4	1
1,1,1 trichloroethane	0.047	0.014	0.01
1,1,2 trichloroethane	0.047	0.014	0.01
Trichloroethene (tri)	47	14	10
Tetrachloromethane (tetra)	0.047	0.014	0.01
Tetrachloroethene (per)	0.047	0.014	0.01
PAH (µg/L)			
Naftalene	0.047	0.014	0.01
Phenantrene	0.028	0.016	0.006
Antracene	0.0066	0.0038	0.0014
Fluoranthene	0.056	0.033	0.006
Chrysene	0.056	0.033	0.006
Benzo(a)antracene	0.0019	0.0011	0.0002
Benzo(a)pyrene	0.0094	0.0054	0.001
Benzo(k)-fluoranthene	0.0075	0.0044	0.0008
Indeno(1,2,3cd)-pyrene	0.0075	0.0044	0.0008
Benzo(ghi)perylene	0.0056	0.0033	0.0006
PAH (sum10)	1.9	1.1	0.2
BTEX (µg/L)			
Benzene	0.94	0.27	0.2
Xylene	0.94	0.27	0.2
Toluene	4.7	1.4	1
Ethylbenzene	4.7	1.4	1
Other (µg/L)			
Phenols	0.94	0.27	0.2

n.a. = not applicable

** = By policy decision, this value is lowered to 130 µg/L. In Section 7.2.2 a further explanation on this topic is given.*

*** = If there is reason to expect that the specific pilot landfill will not be able to meet the calculated ETV. It can be argued to allow a higher emission of up to 50 mg/L for ammonium if account is taken of the terms and conditions described in Appendix 1. This is a policy decision which is not taken in this report.*

**** = For phosphate no reliable ETVs can be calculated (see Section 4.3.5). From the sensitivity analysis it can be concluded that as long as the concentrations of phosphate in the leachate remain below 150µg/L, phosphate will probably not reach the surface water. This value should, however, not be interpreted as an ETV of any kind.*

To derive the ETVs several modelling principles were used and several assumptions were made (see chapter 3 and 4). Although attempts were made to use site-specific information as much as possible, some assumptions made were generic. Where important information was missing generally, worst-case assumptions were made to prevent overestimation of the ETVs, such as the decision to use LOQs as background concentrations in groundwater. Uncertainties in the assumptions were analysed in the sensitivity analysis in a quantitative way (see chapter 6). This report describes in detail the assumptions made and the modelling principles used to derive the ETVs.

However, it should be noted that these assumptions were made and these principles were adopted on the basis of the current understanding of the pilot landfills, state-of-the-art techniques for the modelling of emissions and the available time and capacity. Insights and assumptions may change over time as a result of increasing knowledge and changed perceptions of risks. If, after the period of active treatment of approximately ten years, the conditions at the pilot landfills vary greatly from the assumptions and principles described in this report, the current ETVs – which determine whether or not the treatment and therefore the experiment was successful – may need to be updated. The conclusion drawn, therefore, is that, shortly before the final decision is made as to whether or not the pilot landfills comply with the ETVs, the situation at the landfills should be compared with the assumptions and principles in this report. Special attention in this respect is to be given to the composition of the leachate, such as the DOC content and the presence of aerobic or anaerobic conditions under the landfills.

8.2 Recommendations

8.2.1 Representative monitoring

For various substances, information on concentrations in groundwater and leachate for one or more pilot landfills was scarce. The LOQs are often reported in the monitoring reports and these LOQs are also high relative to what is technically feasible. This has resulted more than once in unreliable outcomes when trying to derive, for example, a site-specific background concentration in groundwater or dilution factors in the surface water of the Wieringermeer pilot landfill. To overcome this problem a conservative approach was often selected as an alternative, because no robust scientific evidence could be found to determine a realistic case. This has resulted in background concentrations equal to the LOQs, which (in most cases) are lower than the expected concentrations in groundwater.

It is therefore advisable that, during the period of active treatment of the pilot landfills, representative monitoring of the concentrations in groundwater (upstream of the pilot landfills) and in the leachate of the relevant landfill compartments be undertaken.

This recommendation also applies to the other landfills that will be selected for active treatment in the future once the experiment at the three pilot landfills proves to be successful.

With representative monitoring, sufficiently low LOQ and a all of substances as described in this report should be pursued. In groundwater, special attention must be given to the metals (cadmium, copper, lead, zinc and mercury) and the macroparameters (sulphate and phosphate; but also calcium, sodium, carbonate). With regard to the leachate, attention should also be paid to the organic substances (individual PAHs, VOX and mineral oils). Once this information has been accounted for, a more realistic and site-specific derivation of ETVs can be aimed for and worst-case assumptions can be turned into realistic scenarios.

Furthermore, it is recommended that a benchmark study be carried out at the beginning of the period of active treatment to determine which other substances (other than the ones described in this report) are present at the pilot landfills.

A selection of substances that were deemed most relevant (see Sections 3.3.2 and 3.3.3) was made. This selection was based on the substances that have to be measured as part of the regulations for landfills in general. Site-specific substances were added to the list for each pilot landfill.

It was clear in advance that these lists were not exhaustive with regard to, for example, the legislation regarding soil and groundwater quality. However, it was also unclear whether or not additional substances such as barium, antimony, tin, vanadium and selenium were relevant substances at the pilot landfills. If it turns out that other substances of concern are present in relevant quantities, the derivation of additional ETVs for such substances should be considered.

8.2.2 *Arsenic*

As discussed in Section 6.2, the conditions under the landfill are very important for arsenic. In the default modelling, arsenic would not arrive at POC2gw within 500 years. So the ETV was derived from the environmental criterion MPA_{eco} for soil as an alternative. This resulted in the ETV of 190 µg/L. However, if in the future the conditions become anaerobic, arsenic will become more mobile and the ETV of 190 µg/L may be insufficient and under-protective. At the moment it is difficult to predict how the conditions under the landfill will develop.

The sensitivity analysis showed that arsenic would not arrive at $POC2_{gw}$ within 500 years under worst-case conditions (a maximum of 50% of the expected binding places for arsenic). Therefore, the calculated ETVs for the Braambergen and Wieringermeer pilots are considered to be sufficiently protective. However, it is advisable to determine the conditions in the leachate after the period of active treatment. If it turns out that the situation is more anaerobic than assumed in the current modelling, a revision of the ETVs for arsenic may be necessary to comply with the environmental criterion at $POC2_{gw}$.

8.2.3 *Breakdown of ammonium*

Ammonium proved to be an important substance for all three of the pilot landfills because of the low ETV with respect to the concentrations of ammonium in leachate to be expected after the period of active treatment. Although the present concentrations of ammonium are expected to be reduced (significantly) during the period of active treatment, it is expected by the pilot landfill operators that the calculated ETVs will not be met.

During the project, limited research (initiated by the pilot landfill operators) was performed to determine whether ammonium would break down further outside the landfill (see Appendix 4 for a summary of the results). The results of this study indicated that a breakdown of ammonium could have occurred at old landfills. However, the likelihood of reduction processes at the pilot landfills and the expected level of ammonium breakdown remain unclear.

If there is a desire to take into account the breakdown of ammonium under the landfill, it is recommended that during the period of active treatment further research on the breakdown of ammonium be performed at the pilot landfills. Based on the results of the preliminary research performed by the landfill operators (see Appendix 4), special attention should be paid to the presence of the Anammox process at the landfills and the expected quantitative contribution of this breakdown.

Furthermore, it is recommended that further research be performed on how the breakdown of ammonium could be accounted for in the modelling of the ETV.

8.2.4 Hydrological modelling

Because of the complexity of the hydrological situation at the Wieringermeer pilot landfill, a hydrological modelling of this situation was performed by order of the landfill operator (see Van Someren, 2013, for details). This proved to be a very informative exercise that allowed for an even more site-specific approach to the landfill. During the project it became clear that some questions also remain with regard to the hydrological situation at the Kragge pilot landfill.

A modelling of the hydrological situation could also be considered for the Kragge and Braambergen landfills. Although it should be noted that each modelling of the hydrological situation comprises some uncertainties, the additional information that could be retrieved from this exercise might result in a more realistic approach to the derivation of the ETVs.

This recommendation also applies to the other landfills that are selected for active treatment in the future once the experiment at the current pilot landfills proves to be successful.

8.2.5 Measuring mineral oil fractions (TPH)

To date, the measurement of TPH fractions divided into aliphatic and aromatic fractions is not a routine job for the analytical laboratories. There is currently a discussion about measuring TPH fractions in the future within the policy framework regarding contaminated soils. In anticipation of this decision, ETVs have been derived for the THP fractions divided into aromatic and aliphatic fractions.

Although the final decision to enforce these ETVs is to be taken by the competent authority after the period of active treatment, it is recommended that during the period of active treatment oil fractions are reported by the laboratories as summed EC10-12, EC12-16, EC16-21 and EC21-35 fractions. This will provide insight into the distribution of the fractions in the landfills, but will not add to the costs of analysis. A division into aromatic and aliphatic fractions is not necessary in this case.

8.2.6 *Emission testing value for phosphate*

No ETVs were calculated for phosphate because validation by measurements indicates that phosphate model predictions are still inadequate (Dijkstra et al., 2009). At the same time, phosphate strongly influences the behaviour of other anions, such as arsenic and sulphate. Currently, phosphate is predicted to strongly sorb in the soil, which would result in an artificially high concentration of phosphate in the assumed emissions, which in turn would result in extremely low calculated ETVs for the substances that are influenced by phosphate. Phosphate is, however, a substance that is frequently measured at landfills in order to comply with the conditions of the landfill permit. Although phosphate is not an immediate risk to groundwater, it does have a eutrophying effect when the groundwater comes in contact with surface water. Therefore, the leaching of phosphate from the pilot landfills still requires monitoring after the period of active treatment. If the concentrations become too high and effects in surface water are expected, action should be taken to prevent the leaching of phosphate from the landfill.

The monitoring of phosphate is especially relevant for the Wieringermeer pilot landfill, because part of the leachate infiltrates the nearby surface water (ditch) surrounding the landfill. The sensitivity analysis showed that the arrival of phosphate at $POC2_{gw}$ is expected only at concentrations as high as 150 mg/L. Although these concentrations are not expected at the pilot landfills (current concentrations for Wieringermeer are around 5–6 mg/L), there should be monitoring of any undesired phosphate effects in the ditch as well as the magnitude of the phosphate concentrations occurring in the leachate. If necessary, action should be taken to prevent any negative effects.

Furthermore, it should be noted that the value of 150 mg/L is not suitable as an indicative value, reference value or ETV of any kind, because the hydrological modelling by Van Someren (2013) indicated that within 750 m of the Wieringermeer pilot landfill the leachate in groundwater would reach surface water and negative effects of eutrophication are to be expected at these concentrations.

8.2.7 *Vulnerable receptor at POC3 and revision of the ETVs*

In the current model the groundwater or surface water at POC2 next to the landfill is designated as a receptor that needs protecting. To this end current receptor-specific environmental criteria are selected (e.g. limits for groundwater or surface water). It is, however, possible that a vulnerable receptor (such as a nature area) is present near the landfill – at a not yet defined POC3 – requiring special attention.

This receptor can be more sensitive than the environmental protection criterion at $POC2_{gw.}$ Also, a combination toxicology between substances can occur increasing the effects on this receptor. The effects are higher than the effect of the individual substances alone. In other words, substances increase each other’s effect on a receptor. In the current study neither a vulnerable receptor at POC3 nor combination toxicology has been taken into account.

It is recommended that in the final evaluation after the period of active treatment the possible presence of a vulnerable receptor be determined. Furthermore, it should be ensured that the environmental criterion at POC2 also provides sufficient protection at POC3. This is probably the case for substances to which the NR_{eco} applies. However, for substances for which the MPR is the environmental criterion (inorganic substances) the possibility of effects at POC3 should be considered if a vulnerable receptor is present. If so, additional measures to prevent negative effects should be considered.

The competent authorities could consider revising the ETVs to protect this vulnerable receptor. This would mean that the ETVs for the period after active treatment are more stringent than the ETVs derived in this report. This would prevent increased emissions coming from the landfill after the period of active treatment.

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List of abbreviations

BC	Background concentration
Bbk	Soil Quality Decree
Bssa	Decree on Landfills and Landfill Bans
COD	Chemical oxygen demand
DHI	Danish Hydraulic Institute
DOC	Dissolved organic carbon
ECN	Energy Research Centre of The Netherlands
ETV	Emission testing values
GWD	Groundwater Directive
HC	Hazardous concentration
HHNK	Hoogheemraadschap Hollands Noorderkwartier
IBC	Isolation, management and control
IDS	Introduction of Sustainable Landfill Management
Ministry of I&M	Ministry of infrastructure and the Environment
JG-MKN	Yearly average environmental quality standard
Kd	Distribution coefficient
LOD	Limit of detection
LOQ	Limit of quantification
MPA	Maximum permissible addition
MPR	Maximum permissible risk
NAVOS	NAzorg VOormalige Stortplaatsen; aftercare of former landfills
NR	Negligible risk, sometimes also referred to as target value
POC	Point of compliance; the addition of 1, 2 or 3 stands for the location of the POC in the conceptual model, and the addition gw or sw stands for groundwater or surface water, respectively.
P95	95th percentile
PAH	Polycyclic aromatic hydrocarbon
Rbk	Regulation soil quality
RIVM	National Institute for Public Health and the Environment

SOC	Solid organic carbon
SRC	Serious risk concentration
TCB	Soil Protection Technical Committee
TDS	Total dissolved substances
TOC	Total organic carbon
VOX	Volatile organochlorine compounds
WFD	Water Framework Directive

Appendix 1: Advice of the Technical Committee on Soil Protection (TCB, reference A082(2013))

A1.1 Advice of the Technical Committee on Soil Protection (TCB)

During the process of deriving ETVs, the TCB was asked by the Ministry of I&M to advise on the starting points relating to the derivation of the ETVs proposed in this report. The TCB's advice included several discussion points and recommendations. The TCB's full advice is included in a TCB document (reference A082(2013)), which can be downloaded from the TCB website. The main discussion points and recommendations are discussed in the sections below. An indication is also given as to whether, and if so how, these points were taken into consideration during the derivation of the ETVs.

A1.1.1 Choice of location of POC

Depending on the hydrological situation, the local conditions present should determine the choice of POC. TCB supports the decision to locate the POC for dispersion via groundwater in a horizontal direction at 20 metres from the boundary of the landfill.

For the Braambergen and Kragge pilot landfills, the location of $POC2_{gw}$ at 20 m downstream of the landfill corresponds to the hydrological situation, as at both landfills there is an infiltration situation with dispersion via the groundwater in a horizontal direction.

The Wieringermeer pilot landfill deviates from this hydrological situation, as there is seepage around the landfill. At the site of the landfill the leachate is expected partly to infiltrate the groundwater and partly to run off to nearby surface water (ditch) (Van Someren, 2013). The groundwater is expected to surface at a relatively short distance from the landfill in other surface waters. The hydrologic modelling carried out by Van Someren (2013) refers to a maximum distance of approximately 750 m. Based on the hydrological situation, two proposals for ETVs that are in keeping with the hydrological situation were taken into account: namely, a standard infiltration scenario, locating $POC2_{gw}$ at 20 m from the landfill in the groundwater, and an additional scenario where $POC2_{sw}$ is located in the channel Westfriesche Vaart as this is where the discharge point is for the water from the ditch. This discharge point is the testing point designated by the competent authority, HHNK. The stricter scenario (infiltration or seepage) will determine the definitive list of ETVs for the Wieringermeer pilot landfill.

By taking into account the specific hydrological situation per landfill, the TCB recommendation is complied with.

A1.1.2 *No increasing trend in concentrations*

The TCB can accept the chosen protection levels for the groundwater at POC2, as long as the additional criterion is adopted that there is no increasing trend in concentrations.

During the period of active treatment, the processes in the landfill are stimulated causing increasing trends in the leachate concentrations, to simulate the degradation and flushing of substances. However, increasing concentrations in the leachate are an inherent part of the period of active treatment.

It is conceivable that after the period of active treatment there may still be an increasing trend in concentrations at POC0. However, all the landfill compartments are closed to further landfill activities and no new waste will be added to the landfill compartments in question. The emission of substances will thus be limited to the loads already present in the landfill.

Increasing trends in the leachate may be a reason to extend the period of active treatment by some time so that the landfill can further stabilize, as long as there is still a functioning bottom liner and the leachate drains are still functioning so that the leachate can be monitored. This choice can be made by the competent authority and the Ministry of I&M after the initial treatment period of approximately ten years.

Ultimately, it is a policy decision whether or not to include an additional criterion about increasing trends in leachate concentrations. From a model point of view this cannot be fleshed out in the derivation of the ETVs.

A1.1.3 *Fleshing-out of protection levels for surface water*

The TCB believes that further protection levels should be worked out if the POC is located in surface water.

The protection levels for surface water (relevant only to the Wieringermeer pilot landfill) are in line with the existing policy for surface water (the Quality Requirements and Water Monitoring Decree (2009) and the Water Framework Monitoring Decree (2010)). Local standards, as drawn up by the water authority HHNK, are also taken into account. To this end, contact was made with the HHNK during the project.

This complies with the TCB recommendation that further protection levels for surface water be worked out.

A1.1.4 *The policy-based increased ETV for ammonium*

The TCB believes that the chosen protection level for ammonium (a policy-based increase to 50 mg/L) for groundwater is acceptable under strict conditions. The strict conditions should prevent any negative influence on surface water due to eutrophication.

In this report the local background concentration is chosen as a protection level for ammonium in accordance with the starting points discussed in Section 3.4. The local background concentration in groundwater is often higher than the environmental protection target in groundwater. Assessment against the environmental protection target in groundwater would be strict in this situation. The landfill sector has indicated that an ETV of between 50 and 100 mg/L can be achieved after the period of active treatment. This is a lot higher than the background concentration adopted for deriving the ETVs.

The Ministry of I&M commissioned a study with the aim of drafting strict conditions under which an increased emission for ammonium from the pilot landfills is acceptable. A maximum concentration of 50 mg/L was adopted.

An ETV of up to 50 mg/L at POC0 in groundwater appears to be acceptable in the specific situation of the landfills that undergo a period of active treatment for the following reasons:

- It relates to a naturally occurring substance that is not very toxic to organisms in groundwater.
- Under the right circumstances, ammonium can be quickly converted into nitrogen, as a result of which concentrations decrease.
- Ammonium is not a persistent substance and does not accumulate in the food chain.
- As only three landfills are looked at within this project, it is possible to identify vulnerable objects near the landfills and to adopt a suitable protection limit for them.
- The greatest objection to higher concentrations of ammonium is the eutrophying effect if ammonium gets into the surface water. If no contact with surface water occurs, effects are expected to be minimal and local. In the IDS project the landfill Wieringermeer has direct contact with surface water. An increased emission of ammonium is in this case discouraged.

An increased emission for ammonium of a maximum of 50 mg/L in groundwater can be considered if the following strict conditions are met:

1. An assessment needs to be carried out with the landfill operator for each landfill individually to ascertain whether an ETV of 50 mg/L is necessary. The advice is not to allow increased emissions generically for ammonium for the three pilot landfills (and all the other landfills in the Netherlands). The landfill operators will have to supply any requested information so that the Ministry of I&M can make a judgement about the need for an increased emission. The landfill operators will also have to indicate why the ETV cannot be achieved for their specific landfills.
2. The advice is to only allow an increased emission of 50 mg/L for the three landfills that are part of the experiment to test active treatment and only if there is a reason to do this on the basis of the hypothesis about the progress of ammonium, as included in the ‘Action Plans’ for each landfill. Both during and after the period of active treatment, an assessment will need to be carried out, in accordance with the ‘Monitoring Plans’, with regard to the development of ammonium concentrations during the treatment period and with regard to the end result and the developments still to be expected after that period. Then the results of the treatment period will also be assessed for all the other substances. On the basis of these assessments, a definitive decision can be made on the allowed emission for ammonium for the pilot landfills. The overall environmental advantages achieved by the active treatment of the landfills can be assessed vis-à-vis the environmental disadvantages of the restricted authorization of higher emissions for ammonium.
3. The advice is to allow an increased emission only if it is clear in the Action Plans that during the active treatment period focused measures are taken to keep ammonium emissions from the landfills to a minimum. After the active

treatment period, accountability should be taken for the results in the monitoring reports and the verification study. It is recommended that measures taken be positioned in relation to the costs of these efforts. Part of these efforts could, for example, be a study into the effect of a longer period of active treatment on the concentration of ammonium.

4. If, during the period of active treatment, the concentrations in the leachate deviate greatly from the forecasts, adjustments should be made and/or a study should be carried out into whether the degradation process can be stimulated, for example through intensification of the measures.
5. If the concentrations in the leachate still show a clear downward trend after completion of the active treatment, and the groundwater monitoring (in accordance with the Landfill and Soil Protection Decree) does not show a failure of the bottom liner, consideration can be given, in consultation with the Ministry of I&M, to continuing with the treatment and/or at least with the monitoring (of the leachate) for a few more years.
6. Vulnerable receptors should be fully protected at all times. These are the designated areas to be protected in the catchment area management plans (implementation of the WFD) as well as the groundwater functions to be protected such as drinking water extraction wells (implementation of the Groundwater Directive). Specifically, the following vulnerable objects are to be protected:
    a. capture zone of the groundwater extractions designated as part of the WFD intended for human consumption;
    b. industrial groundwater extraction points;
    c. soil volumes, surface water, and soil or banks of surface water that come within or form part of: shellfish waters, water for salmon-like and carp-like fish, bathing water and Natura2000 areas;
    d. certain other natural areas (such as designated natural monuments);
    e. certain private water extractions;
    f. areas which are appointed strategic provisions for public drinking water extraction.

    The advice is given to ensure that the vulnerable objects will not be reached geohydrologically in the next 500 years. The standard for ammonium of 0.2 mg/L on the basis of the Drinking Water Act remains fully in force for drinking water extraction wells. The standard for ammonium of 0.3 mg/L in surface water remains fully in force for surface water.

The assessment of ammonium against a concentration of 50 mg/L in surface water is strongly discouraged because of the direct effects of ammonium on the ecosystem and its eutrophying effect on the surface water. Furthermore, if groundwater is in direct contact with surface water at a short distance from the landfill, an allowed increase of the ammonium emission to 50 mg/L is discouraged.

The above conditions provide a fleshing-out of the TCB recommendation. It is a policy decision whether to allow an increased emission of ammonium from the landfills and, if so, under what conditions.

A1.2 Dilution of substances in the groundwater and the occurrence of density flows

The TCB does not find the assumption of dilution over the whole thickness of the aquifer defensible and it cannot accept the way in which the dilution process was physically described. The TCB expects a relatively thin leachate plume coming from the landfill, which remains thin, with dilution occurring over a thickness of *only several centimetres or decimetres. The TCB recommends that the dilution aspect be included in a sensitivity analysis. The TCB also states that density flows should be taken into account, as high chloride content or other anions may make the water from the landfill heavier, as a result of which it infiltrates faster.*

The TCB states that the leachate plume that will occur under the landfill is and will remain thin. In the short term, this is a conceivable scenario as the bottom liner will not fail instantaneously over the entire area of the landfill. At several critical places there will be leaks, which may give rise in the first instance to point sources. However, a time frame of 500 years has been adopted as a starting point and between 50 and 100 years after the fitting of the bottom liner this will increasingly fail until it may be regarded as being absent. Research into the actual life span of the bottom liner is still being carried out as part of the AKS (Advisory Committee on the Landfill Decree).

When the bottom liner fails, infiltration into the soil will take place over the whole area of the landfill, which will make the leachate plume wide. The image of a thin leachate plume is therefore not regarded as realistic for the entire time frame of 500 years and will also not be in keeping with the assumption of a fully failing bottom liner. The thickness of the aquifers of the landfills in question is a maximum of 10 metres (Braambergen pilot landfill). In view of the time frame of 500 years and the relatively shallow groundwater, mixing over the entire thickness cannot be excluded. It may well be necessary to reconsider this assumption for the other landfills (PDS locations) where there is an aquifer of >10 metres.

Finally, in practice, widely fanned-out contaminant plumes, both horizontal and vertical, in the groundwater are found in cases of soil contamination where there is dispersion from a point source (or, for example, from a landfill without a bottom liner and top cover, such as the NAVOS landfills).

The occurrence of density flows is not expected. A starting point for sustainable landfill management is that the concentrations of metals and organic substances are so low that density flows will not occur. According to Bot (2011), differences in the density of chloride in groundwater are significant in flow modelling if chloride concentrations are above 5000 mg/L. This is not the case with the pilot landfills. So the occurrence of density flows is not taken into account.

A1.3 Influence of dissolved organic matter

The TCB believes that the expectation that the concentration of dissolved organic matter (DOC) is greatly decreased at the end of the period of active treatment is very uncertain. The TCB expects that after approximately ten years there will still be a substantial amount of DOC present in the leachate, which will have an influence on the mobility of other substances.

The starting point for DOC was that no increased concentrations of DOC were expected after the treatment period, as these would have been degraded/bound during the treatment period. Further research and the results of the sensitivity analysis prompted a review of this assumption. The sensitivity analysis showed that the influence of DOC was so great that not including an increased DOC concentration in the expected emission from the landfill would be unrealistic. In the derivation of the ultimate ETVs an increased DOC concentration from the landfill was therefore taken into account. This complies with the TCB recommendation.

In view of the influence of the concentration of DOC on the ETVs, it is recommended that after the treatment period of approximately ten years, the DOC concentrations present be monitored and compared with the assumption in this report. If the concentrations measured deviate considerably from the assumption in this report , it is to be recommended that the ETVs be re-established.

A1.4 Testing value for dissolved organic matter

The TCB puts forward for consideration that a testing value for dissolved organic matter in the leachate, comparable with an ETV, be considered. On the basis of this value and the trend found in the concentration of dissolved organic matter, a decision can possibly be taken to extend the treatment of the landfill.

It is not possible to derive a testing value for DOC in accordance with the method that was used for inorganic and organic substances. First of all, a judgement should be made about what an acceptable DOC concentration in groundwater is, as there is no environmental criterion in this respect. Second, it is not possible to predict the behaviour of DOC in the zone between POC0 and $POC2_{gw}$ with the current models. So it is not possible to derive a DOC concentration at POC0. Finally, no testing value for DOC can be derived, as DOC itself has an exceptionally great influence on the ETVs of metals. So it is not possible to simultaneously find an ETV for both DOC and the substances that bind to it.

A1.5 Influence of reducing conditions

The TCB believes that the increased mobility of substances (such as arsenic) as a result of the release of iron oxides under reducing conditions should be considered in the sensitivity analysis. It is proposed that regional, sediment-geochemical data be used.

It is difficult to predict what the chemical composition of the soil under the landfill will be in the future. Yet it is useful to know the influence of reducing conditions on the ETVs. So the influence of reducing conditions was included in the sensitivity analysis. The starting point was the dissolving of 50% of the assumed amount of iron oxides (HFO) in the receiving soil. Full dissolving of iron oxides is highly improbable (only in the case of extremely low pH and extremely low redox potential). The sensitivity analysis showed that the influence of reducing conditions is relatively low compared with, for example, the influence of increased DOC concentrations.

In view of the limited influence of reducing conditions on the ETVs and the fact that it is difficult to predict the chemical composition of the soil under the landfill, reducing conditions were not taken into account in the derivation of the ETVs.

A1.6 New substances

The TCB believes that so-called ‘new substances’ are not included in the current list of substances. The TCB therefore recommends that a benchmark study with a wide screening focused on organic micropollutants is performed. An assessment could be carried out against the signal value (0.1 µg/l) for ‘other substances’ from the Decree on Drinking Water. The TCB further recommends extending the benchmark study with biological effect measurements (bioassays) so that there is insight into the toxicity of the whole mixture of substances coming from the landfill.

The selection of the relevant substances for each landfill took into account the substances that are regularly found, or the substances that have to be included in the monitoring as part of the conditions of the permit of the pilot landfills in question. It was clear in advance that this is a limited list of substances in comparison with the standard monitoring list for groundwater. In addition to the organic substances, the new substances, such as tin, antimony and barium, should be included in the benchmark study. On the basis of the benchmark study, it can be concluded if additional ETVs are necessary for these new substances For new substances for which no environmental criterion is available, an initial ETV could be derived on the basis of the signal value referred to by the TCB, as described in the Decree on Drinking Water (Ministry of I&M, 2011b).

Before considering bioassays it sould be clear what is the goal of such a study. This advice concerns above all the monitoring of the landfill, or the start-up of the period of active treatment. For the establishment of the current set of ETVs the use of bioassays has no further consequences for the time being.

A1.7 Overview of actions resulting from the TCB recommendation

Table A1.1 gives an overview of the actions resulting from the TCB recommendations that have been included in the model design for the derivation of the ETVs.

Table A1.1: Overview and explanation of why TCB advice was adopted in the model structure or not.

	Explanation
Advice adopted in the model structure	
Determine the position of the POC on the basis of the hydrological situation	In accordance with advice
Base protection levels on surface water if necessary	In accordance with advice
Include increased DOC concentration coming from the landfill	In accordance with advice
Advice not adopted in the model structure	
Dilution over entire thickness of aquifer	Based on the time frame of 500 years and the relatively shallow aquifers, it cannot be ruled out that dilution will occur over the entire thickness of the aquifer
Take into account density flow	Density flow is not expected after the period of active treatment at the landfills
Derive a testing value for organic matter	It is technically not possible to do so
Include increased mobility of substances due to the release of iron oxides	Variation in mobility is examined in the sensitivity analysis but because of limited influence is not incorporated into the model structure
Advice which has no influence on the model structure	
Include a criterion to prevent increasing concentrations in the leachate	This is a policy decision, which will not be made in this report
Allow increased emissions of ammonium	This is a policy decision, which will not be made in this report
Perform a benchmark study at the start of the period of active treatment	This is a policy decision, which will not be made in this report

Appendix 2: Environmental criterion at POC2

Tables A2.1 and A2.2 present the relevant substances and environmental quality criteria for the pilot landfills. In Table A2.1 columns 2–4 present the NR$_{eco}$ (=MPR$_{eco}$/100), MPR$_{eco}$ for groundwater and the drinking water standards. As a reference, the intervention values for groundwater are also presented in column 5. The Wieringermeer pilot landfill is additionally compared to environmental criteria for surface water – see Table A2.2. In columns 2, 3 and 4 of this table, the yearly average environmental quality standard for surface water (JG-MKN), the MPR$_{eco}$ for surface water and locally determined values of the water board are presented.

Table A.2.1: Relevant environmental protection criterion for groundwater at POC2$_{gw}$ for the Braambergen, Kragge and Wieringermeer (groundwater) pilot landfills.

Substance	**NR$_{eco}$ groundwater (µg/L)**	**MPR$_{eco}$ (ground)water (µg/L)**	**Drinking water standard (µg/L)**	**Intervention values groundwater (µg/L)**
In organic substances (metals)	(For metals equal to target values deep groundwater including generic BC)	(MPA for metals)		
Arsenic	7.2[1]	24[2]	10[3]	60[1]
Cadmium	0.06[1]	0.34[2]	5[3]	6[1]
Chrome	2.5[1]	8.7[2]	50[3]	30[1]
Copper	1.3[1]	1.1[2]	2,000[3]	75[1]
Mercury	0.01[1]	0.23[2]	1[3]	0.3[1]
Lead	1.7[1]	11[2]	10[3]	75[1]
Nickel	2.1[1]	1.9[2]	20[3]	75[1]
Zinc	24[1]	7.3[2]	3,000[3]	800[1]
Mineral oil				
Aliphatic EC5-EC6	0.17[4]	17[4]	1[3]	-
Aliphatic EC6-EC8	0.039[4]	3.9[4]	1[3]	-
Aliphatic EC8-EC10	0.005[4]	0.5[4]	1[3]	-
Aliphatic EC10-EC12	0.00127[4]	0.127[4]	1[3]	-
Aliphatic EC12-EC16	0.00071[4]	0.071[4]	1[3]	-
Aromatic EC5-EC7	1.23[4]	123[4]	1[3]	-
Aromatic EC7-EC8	0.83[4]	83[4]	1[3]	-
Aromatic EC8-EC10	0.55[4]	55[4]	1[3]	-
Aromatic EC10-EC12	0.32[4]	32[4]	1[3]	-
Aromatic EC12-EC16	0.14[4]	14[4]	1[3]	-
Aromatic EC16-EC21	0.038[4]	3.8[4]	1[3]	-
Aromatic EC21-EC35	0.0032[4]	0.32[4]	1[3]	-
SUM mineral oil	50[1]	-	-	600[1]
VOX				
Vinylchloride	0.01[1]	1[6]	0.1[3]	5[1]
Dichloromethane	0.01[1]	1[6]	1[3]	1,000[1]

Substance	**NR$_{eco}$ groundwater (µg/L)**	**MPR$_{eco}$ (ground)water (µg/L)**	**Drinking water standard (µg/L)**	**Intervention values groundwater (µg/L)**
1,1 dichloroethane	7[1]	700[6]	1[3]	900[1]
1,2 dichloroethane	7[1]	700[7]	3[3]	400[1]
1,1 dichloroethene	0.01[1]	1[6]	1[3]	10[1]
1,2 dichloroethene (cis,trans)	0.01[1]	1[6]	1[3]	20[1]
Dichloropropane (1,2)	0.8 (sum) [1]	806	1[3]	80[1] (sum)
Dichloropropane (1,3)	0.8 (sum) [1]	76[7]	1[3]	80[1] (sum)
Trichloromethane (chloroform)	6[1]	600[7]	1[3]	400[1]
1,1,1 trichloroethane	0.01[1]	1[6]	1[3]	300[1]
1,1,2 trichloroethane	0.01[1]	1[6]	1[3]	130[1]
Trichloroethene (tri)	24[1]	2,400[7]	10 (sum with per) [3]	500[1]
Tetrachloromethane (tetra)	0.01[1]	1[6]	1[3]	10[1]
Tetrachloroethene (per)	0.01[1]	1[6]	10 (sum with tri) [3]	40[1]
PAH				
Naftalene	0.01[1]	1.2[8]	-	70[1]
Phenantrene	0.003[1]	0.3[8]	0.1 (sum) [3]	5[1]
Antracene	0.0007[1]	0.08[8]	0.1 (sum) [3]	5[1]
Fluoranthene	0.003[1]	0.3[8]	0.1 (sum) [3]	1[1]
Chrysene	0.003[1]	0.9[8]	0.1 (sum) [3]	0.2[1]
Benzo(a)antracene	0.0001[1]	0.03[8]	0.1 (sum) [3]	0.5[1]
Benzo(a)pyrene	0.0005[1]	0.05[8]	0.01[3]	0.05[1]
Benzo(k)fluoranthene	0.0004[1]	0.04[8]	0.1 (sum) [3]	0.05[1]
Indeno(1,2,3cd)-pyrene	0.0004[1]	0.04[8]	0.1 (sum) [3]	0.05[1]
Benzo(ghi)perylene	0.0003[1]	0.03[8]	0.1 (sum) [3]	0.05[1]
BTEX				
Benzene	0.2[1]	20[6]	1[3]	30[1]
Xylene	0.2[1]	20[6]	1[3]	70[1]
Toluene	7[1]	700[6]	1[3]	1000[1]
Ethylbenzene	4[1]	370[7]	1[3]	150[1]
Cyanide	5 (free) [1] 10(complex) [1]	-	50[3]	1500[1]
Phenols	0.2 (sum) [1]	-	1[3]	2000[1]
Macroparameters				-
Chloride	100 (mg/L) [1]	94 (mg/L)[5]	150 (mg/L) [3]	-
Sulphate	-	-	150 (mg/L) [3]	-
Ammonium/nitrogen	-	0.304[9]/2.2[9] (mg/L)	0.2 (mg/L) [3]	-

1 *Ministry of I&E (2013). Circular on soil remediation. Gazette no. 16675.*

2 *Verbruggen E.M.J., Posthumus R., Van Wezel A.P. (2001). Ecotoxicological serious risk concentrations for soil, sediment and (ground)water. Updated proposals for first series of compounds. National Institute for Public Health and the Environment, Bilthoven, Netherlands. RIVM report no.711701 020.*

3 *Ministry of I&E (2011b). Decree on Drinking Water. Gazette no. 293.*

4 *Verbruggen E.M.J., Beek, M., Pijnenburg, J., Traas, T.P. (2008a). Ecotoxicological environmental risk limits for total petroleum hydrocarbons on the basis of internal lipid concentrations. Environmental Toxicology and Chemistry, vol. 27, no. 12, pp. 2436–2448.*

5 *Verbruggen E.M.J., Moermond C.T.A., Janus J.A., Lijzen J.P.A. (2009). Afleiding van milieurisicogrenzen voor chloride in oppervlaktewater, grondwater, bodem en waterbodem (in Dutch). RIVM, Bilthoven, The Netherlands. RIVM report no. 711701 075.*

6 *Derived from the target value x 100.*

7 *Van de Plassche E.J., Bockting G.J.M. (1993). Towards integrated environmental quality objectives for several volatile compounds. RIVM, Bilthoven, The Netherlands. RIVM report no. 679101 011.*

8 *Kalf D.F., Crommentuijn G.H., Posthumus R., Van de PLassche E.J. (1995). Integrated environmental quality objectives for polycyclic aromatic hydrocarbons (PAHs). RIVM, Bilthoven, The Netherlands. RIVM report no. 679101 018.*

9 *Risk from compounds (Database RIVM, http://www.rivm.nl/rvs/).*

Table A.2.2: Relevant environmental protectioncriteria at POC2$_{sw}$ for surface water for the Wieringermeer pilot landfill (surface water).

Substance	**JG-MKN surface water (µg/L)**	**MPR surface water (µg/L)**	**Locally determined values of the water board HHNK (mg/L)**	**Source**
Inorganic substances (metals)				
Arsenic	-	32	-	1
Cadmium	0.15	-	-	2
Chrome	3.4	-	-	1
Copper	-	3.8	-	1
Mercury	7.2	-	-	2
Lead	20	-	-	2
Nickel	0.05	-	-	2
Zinc	7.8	-	-	1
Mineral oil				
Aliphatic EC5-EC6	-	17	-	3
Aliphatic EC6-EC8	-	3.9	-	3
Aliphatic EC8-EC10	-	0.5	-	3
Aliphatic EC10-EC12	-	0.127	-	3
Aliphatic EC12-EC16	-	0.071	-	3
Aromatic EC5-EC7	-	123	-	3
Aromatic EC7-EC8	-	83	-	3
Aromatic EC8-EC10	-	55	-	3
Aromatic EC10-EC12	-	32	-	3
Aromatic EC12-EC16	-	14	-	3
Aromatic EC16-EC21	-	3.8	-	3
Aromatic EC21-35	-	0.32	-	3
SUM mineral oil	50	-	-	2
VOX				
Vinylchloride	0.09	-	-	1
Dichloromethane	20	-	-	1
1,1 dichloroethane	-	700	-	1
1,2 dichloroethane	10	-	-	2
1,1 dichloroethene	9	-	-	1
1,2 dichloroethene (cis,trans)	6.8	-	-	1
Dichloropropane (1,2)	280	-	-	1
Dichloropropane (1,3)	-	76	-	4
Trichloromethane (chloroform)	2.5	-	-	2
1,1,1 trichloroethane	21	-	-	1
1,1,2 trichloroethane	22	-	-	1
Trichloroethene (tri)	10	-	-	1
Tetrachloromethane (tetra)	12	-	-	2
Tetrachloroethene (per)	10	-	-	2
PAH				
Naftalene	2.4		-	2
Fenantrene		0.3	-	1
Antracene	0.1	-	-	2
Fluoranthene	0.1	-	-	2
Chrysene	-	0.9	-	1
Benzo(a)antracene	-	0.03	-	1

Substance	**JG-MKN surface water (µg/L)**	**MPR surface water (µg/L)**	**Locally determined values of the water board HHNK (mg/L)**	**Source**
Benzo(a)pyrene	0.05	-	-	2
Benzo(k)fluoranthene	0.03	-	-	2
Indeno(1,2,3cd)-pyrene (Sum with benzo(ghi)pyrene)	0.002	-	-	2
Benzo(ghi)perylene (Sum with Indeno(1,2,3cd)-pyrene	0.002	-	-	2
BTEX				
Benzeen	10	-	-	2
Xyleen	2.44 (SuM)	-	-	5
Tolueen	74		-	1
Ethylbenzeen	-	370	-	1
Cyanide	-	0.23 (vrij) 0.13 (complex)	-	5
Fenolen	-	100	-	5
Macroparameters				
Chloride	-	94 (mg/L)[7]	460 (mg/L)	6
Sulphate	-	100 (mg/L)[5]	382 (mg/L)	6
Ammonium/nitrogen	-	0.304[5]/2.2[5] (mg/L)	0.65 (mg/L)	6
Phosphate	-	0.15 (mg/L)[5]	1.75 (mg/L)	6

1 *Ministry of I&M (2010) Regulation on monitoring the Water Framework Directive. Gazette no. 5615.*

2 *Ministry of Infrastructure and Environment (20010). Decree on quality standards and monitoring water. Gazette no. 15 (2010).*

3 *Verbruggen, E.M.J., Beek, M., Pijnenburg, J., Traas, T.P. (2008a). Ecotoxicological environmental risk limits for total petroleum hydrocarbons on the basis of internal lipid concentrations. Environmental Toxicology and Chemistry, vol. 27, no. 12, pp. 2436–2448.*

4 *Van de Plassche E.J., Bockting G.J.M. (1993). Towards integrated environmental quality objectives for several volatile compounds. RIVM, Bilthoven, The Netherlands. RIVM report no. 679101 011.*

5 *Risk from compounds (Database RIVM, http://www.rivm.nl/rvs/).*

6 *Locally established interim quality standards by the water board (HHNK).*

7 *Verbruggen E.M.J., Beek M., Pijnenburg J., Traas T.P. (2008). Ecotoxicological environmental risk limits for total petroleum hydrocarbons on the basis of internal lipid concentrations. Environmental Toxicology and Chemistry, vol. 27, no. 12, pp. 2436–2448.*

Appendix 3: Options considered for the environmental criterion for chloride

Some of the pilot landfills and the so-called PDS locations are in groundwater areas with high chloride concentrations (brackish groundwater). For these landfills, assessment against the MPR_{eco} for chloride at $POC2_{gw}$ has no value, as the concentration of chloride in the surrounding groundwater is higher than the MPR_{eco} that was derived for fresh groundwater areas.

As it is not technically possible to derive an MPR_{eco} for brackish groundwater, an alternative test method needs to be found. It is generally accepted that the exposure of organisms to chloride in brackish groundwater systems may be higher than the background concentrations naturally present because of the adaptation of the organisms to increased chloride concentrations. It cannot be ascertained on the basis of scientific knowledge how great this additional load might be. So within the IDS project various options for testing chloride were considered. A balanced consideration was made between protecting the ecosystem, carrying out the active treatment and respecting current environmental policy.

The following sections describe the four options that were considered.

A3.1 Option 1: Always assess against the local background concentration + MPR_{eco} even if the local background concentration is <MPR_{eco}

In this option the local background concentration is always added to the MPR_{eco}, even if there is a fresh groundwater situation (AW <MPR_{eco}). With this option the local concentrations in groundwater are taken into account. The local background concentration has no influence on the extent of the permitted influence of the groundwater quality at $POC2_{gw}$. This is always equal to the MPR_{eco}.

In this option, for practical reasons, the MPR_{eco} was chosen as an addition to the local background concentration. As indicated above, there is no scientific method for determining how large the additional load in brackish groundwater is. The MPR_{eco} has no value for brackish groundwater in a toxicological sense.

A3.2 Option 2: If the local background concentration is <MPR_{eco}, assess against MPR_{eco}; if the background concentration is >MPR_{eco}, assess against local background concentration + MPR_{eco}

If the local background concentration is <MPR_{eco}, there is a fresh groundwater situation and assessment is carried out against the MPR_{eco} for fresh groundwater in accordance with the generic framework. If the local background concentration >MPR_{eco}, assessment against MPR_{eco} + local background concentration provides additional space for higher chloride loads at locations with brackish groundwater.

For landfills where the local background concentration is <MPR_{eco}, the local background concentration has an influence on the extent of the allowed additional load in groundwater. The influence of the groundwater quality reduces increasingly as the BC becomes closer to the MPR_{eco} (see Table A3.1). For example, if the local background concentration is 10 mg/L, the additional load of the groundwater can be 84 mg/L, and if the local background concentration is 90 mg/L, the additional load of the groundwater can be 4 mg/L.

A3.3 Option 3: If the local background concentration is <MPR_{eco}, assess against MPR_{eco}; if the local background concentration is >MPR_{eco}, assess against local background concentration + a fixed % of the local background concentration

This option is a variant on Option 2. However, here a percentage (to be further defined) of the local background concentration is chosen if the local background concentration is >MPR_{eco}. Here, too, the size of the percentage cannot be scientifically established; 30% of the local background concentration was an initial proposal. This option has the same limitations as Option 2.

A3.4 Option 4: If the local background concentration is <MPR_{eco}, assess against MPR_{eco}; if the local background concentration is >MPR_{eco}, assess against local background concentration

This option is often used as a pragmatic choice within the Soil and Groundwater Quality Framework. As this framework is often concerned with concentrations of chloride that are already present, this option is adequate within this framework. However, within the IDS project there is the intention to permit a minimum residual emission from the landfills within the policy-related and accepted frameworks.

In the case of landfills where the local background concentration is >MPR_{eco}, the concentration coming from the landfill should be equal to the local background concentration, however it is precisely these areas that can better tolerate the additional load because of the adaptation of organisms.

A3.5 Influence of the environmental criterion

To clarify the influence of the above options, Table A3.1 gives an overview of the standards for chloride per option and the additional load of the groundwater quality at POC2_{gw}. The local background concentrations used are fictitious but real concentrations.

Table A3.1: Influence of the discussed options as environmental criterion for chloride (mg/L).

	Environmental criterion groundwater at POC2_{gw} and space with respect to local background concentration							
Local BC	**Option 1 POC2_{gw}**	**Option 1 space to BC**	**Option 2 POC2_{gw}**	**Option 2 space to BC**	**Option 3 POC2_{gw}**	**Option 3 space to BC**	**Option 4 POC2_{gw}**	**Option 4 space to BC**
10	104	94	94	84	94	84	94	84
30	124	94	94	64	94	64	94	64
50	144	94	94	44	94	44	94	44
90	184	94	94	4	94	4	94	4
100	194	94	194	94	130	30	100	0
200	294	94	294	94	260	60	200	0
500	594	94	594	94	650	150	500	0
1000	1094	94	1094	94	1300	300	1000	0

Taking the above into account a decision was made always to test against the MPR_{eco} + a local background concentration (Option 1), even if there is a fresh groundwater situation. In contrast with generic groundwater policy, this option permits an additional chloride load on top of the MPR_{eco} for organisms in fresh groundwater. Any effects on freshwater organisms are expected to be mainly local and to be minimal.

The reason for choosing Option 1 is that it is probable that organisms in brackish groundwater can handle the extra emission equal to the MPR_{eco} on top of the local BC. Although the MPR_{eco} does not have any significance for toxic effects in brackish groundwater, this was chosen for pragmatic reasons. Option 1 is also in line with the starting points of the WFD. In the WFD, generic testing against the MPR_{eco} is carried out for surface water. However, the local background concentration can also be taken into account (EC, 2000) if there is reason to do so. Within the Groundwater Directive, too, there is the opportunity to take local background concentration into account (EC, 2006).

Within the IDS project there is a landfill-specific approach where local conditions are taken into account, so Option 1 does not deviate from the WFD and the GWD. The assessment against the MPR_{eco} + local background concentration for chloride is therefore also regarded within this project as a supplement to the generic assessment against the MPR_{eco}.

Finally, it should be noted that generic (ground)water policy is based on an ‘old’ MPR_{eco} for chloride of 200 mg/L and in this report the new, scientifically established MPR_{eco} of 94 mg/L is used as a basis (Verbruggen et al. 2008b).

Appendix 4: Research on the degradation and binding of ammonium under landfills

A4.1 Introduction

During the derivation of the ETVs the landfill operators indicated that an ETV for nitrogen and in particular ammonium that is based on the current environmental criteria in groundwater will probably not be achievable after the period of active treatment. According to the landfill operators, current concentrations of ammonium in the leachate for several pilot landfills are approximately 1500 mg/L. The following protection levels (located 20 m downstream of a landfill in the aquifer ($POC2_{gw}$)) were discussed as the basis for the calculation of the ETV:

- Ammonium: 0.3 mg/L (based on surface water);
- Ammonium in drinking water: 0.2 mg/L;
- Total nitrogen compounds: 2.2 mg/L (based on surface water).

At the request of the Ministry of I&M the possibility of taking degradation and/or binding of ammonium in soil into account in the modelling was explored.
The following aspects were investigated:

- the model-based inclusion of the retardation of ammonium between $POC1_{gw}$ and $POC2_{gw}$ (in addition to the hydrological dilution already used);
- degradation of ammonium using, for example, the Anammox process. To this end, Royal Haskoning/DHV was commissioned by the landfill operators to carry out an explorative investigation into the degradation of ammonium under landfills in the NAVOS project. The results of this study could result in a factor for generic degradation under landfills. Refer to the study by Royal Van Meeteren & Van Vliet (2012) for details. Within this study there was also consultation with Mr Van Loosdrecht (Delft Technical University). Mr Van Loosdrecht carries out research on the Anammox process.

A4.2 Retardation of ammonium through exchange

The ECN carried out a study into the retardation of ammonium by binding to clay particles. The question that was central to this study was:

Does the binding (through exchange) of ammonium to clay particles in the soil produce sufficient retardation in the soil such that the ETV is influenced by it?

In order to answer this question, ammonium was explicitly added to the ORCHESTRA model calculation files. Exchange takes place in the Donnan layer of the clay particles (positively charged ammonium versus permanently negatively charged clay surfaces). A study was set up for the Braambergen pilot landfill, where the clay content is relatively high (20–25%) and the effect of the addition was expected to be most visible. Calculations were done for a 2 metre soil passage and an exchange simulation was done over a distance of 20 metres (corresponding with the whole distance between POC0 and $POC2_{gw}$). Figures A4.1a and b and A4.2 show the arrival of ammonium at $POC2_{gw}$ (breakthrough curves).

This shows that the effect on the retardation of ammonium by binding of ammonium to clay particles is too small to influence the ETVs. The arrival of ammonium takes place virtually at the same time as that of a mobile substance such as chloride (Figures A4.1a and A4.1b). If a simulation is done for exchange across the entire distance POC0–$POC2_{gw}$, a small retardation effect is visible

(Figure A4.2), but ammonium still arrives well within 500 years at $POC2_{gw}$. So the binding of ammonium in the soil does not influence the ETVs derived.

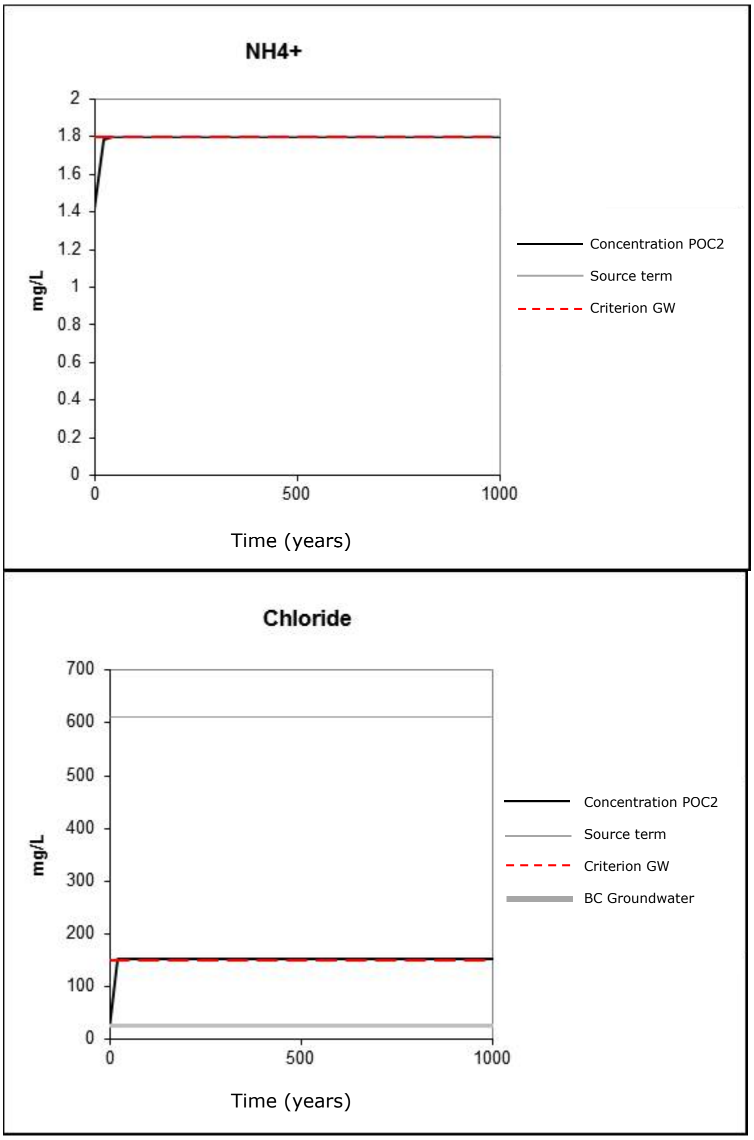

Figure A4.1a: Time of arrival of ammonium and chloride (A4.1b) at $POC2_{gw}$ *if retardation of ammonium by the binding on clay particles is taken into account between 0 and 2 metres depth.*

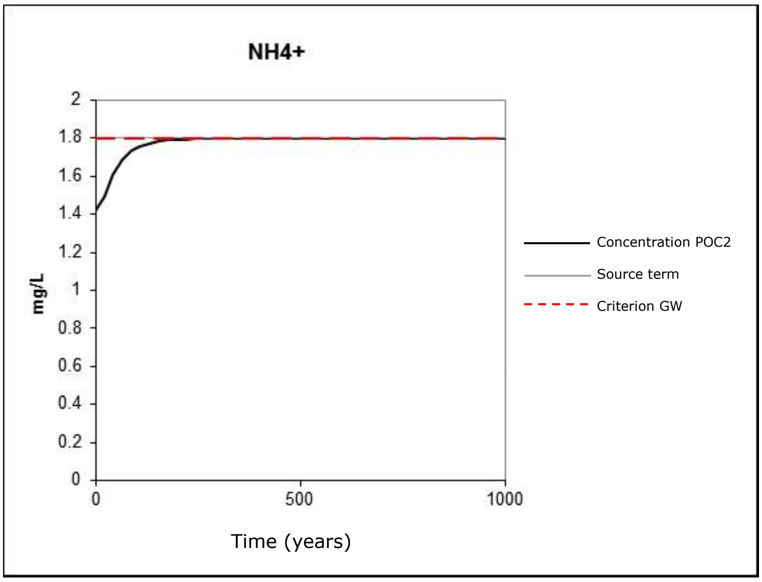

Figure A4.2: Time of arrival of ammonium at $POC2_{gw}$ if retardation of ammonium by binding to clay particles over the total distance between POC0 and $POC2_{gw}$ is taken into account.

A4.3 Degradation of ammonium by the Anammox process

At the request of the landfill operators, Royal Haskoning/DHV carried out an exploratory study into the degradation of ammonium by the Anammox process or any other degradation processes at the NAVOS landfills. For details of this study refer to the report 'Analysis of NA-measurements for NAVOS landfills: Sub-project as part of the Introduction of Sustainable Landfill Management' by Van de Sande and Van Vossen (2012).

The study by Royal Haskoning/DHV demonstrates that there may be removal of ammonium under the landfills of the NAVOS project. However, there is still insufficient insight as to whether this removal is caused by degradation, binding or hydrological dilution.

In addition, it is still unclear whether the degradation of ammonium can actually be modelled for the three pilot landfills and whether the degradation can take place at the selected pilot landfills. Specific knowledge is lacking about the presence of critical success factors. For example, knowledge is lacking about:
1. the presence of the Anammox bacterium;
2. the occurrence of sufficient mixing of the groundwater and the leachate under the pilot landfills to allow the Anammox process to take place;
3. the presence of an electron acceptor (e.g. oxygen or nitrite) in the groundwater upstream of and under the pilot landfills;
4. the presence of nitrate in the groundwater upstream of and under the pilot landfills;
5. the presence of organic matter from the pilot landfills (ammonium is not oxidized in the presence of organic matter);
6. the isotopic distribution of the residual nitrogen (the distribution indicates which degradation process has taken place under the pilot landfills).

Without knowledge of these factors it is not possible to make a judgement about the presence or extent of the occurrence of the Anammox process.
Finally, the model-based inclusion of a generic reduction factor, such as can be derived from the report by Royal Haskoning/DHV, is not possible for ammonium in the short term.

In view of the points listed above, the degradation or binding of ammonium under the landfills is not included in the modelling of the ETVs.

Appendix 5: Soil data from the STONE database

For information about the soil profile of the subsoil under the pilot landfills, data from the STONE database were used. The STONE database was developed by the DLO (Foundation of Agricultural Research), the RIVM and the former RIZA (Rijksinstituut voor Integraal Zoetwaterbeheer en Afvalwaterbehandeling (Institute for Inland Water Management and Waste Water Treatment), now called Rijkswaterstaat Water Service) for scenario studies into the flushing of nitrogen and sulphate in The Netherlands. This database shows the subsoil of The Netherlands for which the most important soil and hydrological characteristics are known (Kroes et al. 2001).

For the three pilot landfills, the nearest STONE plot was selected in order to gain insight into the soil profile. Tables A5.1, A5.2 and A5.3 show the selected data from the STONE database for the Braambergen, Kragge and Wieringermeer pilot landfills, respectively. The average distance of the specific STONE plots from the pilot landfills was approximately 2 km.

Table A5.1: Soil profile in selected STONE plot for the Braambergen pilot landfill. The selected STONE plot has the following features: coordinates X: 148000 Y: 483000, no. 2250, BCE 61333.

Top layer (cm-mv.)	**Humus (% solid phase)**	**% clay (% mineral parts)**	**% loam (% mineral parts)**	**pH-KCl**	**FeAl_ox (mmol/kg)**	**Fe_Ox (mmol/kg)**	**Al_Ox (mmol/kg)**	**CEC**	**$CaCO_3$ (% solid phase)**	**Density (Kg/dm^3)**
0	5.1	22.3	54.3	6.1	130	64	65	180	1.5	1.291
5	5.2	23.5	58.2	6.1	127	61	66	169	1.5	1.279
15	4.3	24.2	59	6.2	130	64	66	170	1.5	1.31
20	3.8	24.8	61	6.2	124	63	60	167	1.5	1.324
25	2.9	25.6	58.9	6.3	123	62	60	174	3	1.349
35	2.1	26.2	60.5	6.3	116	60	55	184	3	1.371
50	1.7	25.3	54.2	6.5	110	63	46	189	3	1.391
60	1.5	21.9	46	6.6	109	53	56	172	5	1.425
75	1.4	18.1	35.7	6.7	97	55	41	138	4	1.46
100	1.5	12.3	24	6.9	70	51	18	160	4	1.507

Table A5.2: Soil profile in selected STONE plot for the Kragge pilot landfill. The selected STONE plot has the following features: coordinates X: *82306 Y: 391357, no. 5945, BCE 51121.*

Top layer (cm-mv.)	Humus (% solid phase)	% clay (% mineral parts)	% loam (% mineral parts)	pH-KCl	FeAl_ox (mmol/kg)	Fe_Ox (mmol/kg)	Al_Ox (mmol/kg)	CEC	$CaCO_3$ (% solid phase)	Density (Kg/dm^3)
0	9.7	6.2	15.5	4.6	94	49	45	78.7	0	1.197
5	9.7	6.2	15.5	4.6	95	49	46	101	0	1.197
15	14.7	7.2	12.4	4.4	99	50	49	101	0	1.082
20	23.6	4.5	13.3	4.2	131	55	75	72	0	0.603
25	14.5	3	13.4	4.3	90	45	44	0	0	1.124
35	12.7	2.5	12.4	4.3	75	40	35	100.5	0	1.169
50	12.2	2.7	11.6	4.1	49	19	30	62.5	0	1.183
60	11.7	2.7	12.5	4.2	42	18	23	62.5	0	1.195
75	14	2.5	11.1	4.3	32	22	9	32.8	0	1.139
100	12.5	2.2	12	4.7	34	24	9	33.3	0	1.175

Table A5.3: Soil profile in selected STONE plot for the Wieringermeer pilot landfill. The selected STONE plot has the following features: coordinates X: 133500 Y: 531500, no. 2079, BCE 163333.

Top layer (cm-mv.)	Humus (% solid phase)	% clay (% mineral parts)	% loam (% mineral parts)	pH-KCl	FeAl_ox (mmol/kg)	Fe_Ox (mmol/kg)	Al_Ox (mmol/kg)	CEC	$CaCO_3$ (% solid phase)	Density (kg/dm^3)
0	3.5	25.4	66.2	7	119	68	50	209	7	1.33
5	3.6	25.7	67.2	7	119	67	52	214	7	1.325
15	3.4	26.3	69.1	7	117	67	49	212	7	1.328
20	3.3	26.7	69.2	7	115	67	47	216	7	1.328
25	2.8	25.3	65.2	7	114	67	47	219	8	1.355
35	2.1	23.9	59.5	7	116	60	55	192	8	1.388
50	1.8	22.2	54.8	7.1	110	63	46	171	8	1.412
60	2	20.4	50.4	7.2	116	66	49	156	8	1.419
75	1.9	17.2	41.7	7.2	108	67	40	123	8	1.449
100	2.3	12.5	25.5	7.2	77	57	20	106	8	1.474

Appendix 6: Phosphate concentration in the leachate

Following the sensitivity analysis, attempts were made to establish a realistic concentration for phosphate in the emission from the landfill based on data from the landfill database. This concentration was then used to model the ETVs.

A phosphate concentration of 0.05 mg PO4/L was chosen as the initial concentration value in the pore water of the receiving soil and as the background value in the groundwater. This value is based on one soil from the dataset established by Dijkstra et al. (2009). In this dataset, in addition to the concentration of P (phosphorous) between pH 4 and 8, PO4 was measured. The measured PO4 concentration was ten times lower than that of P in the dataset. However, the data are very limited. Total P concentrations are higher than PO4 concentrations but probably partly organic. The MPR_{eco} for phosphate in groundwater is 0.15 mg/L.

Considerations with respect to phosphate are:

- There are no measurement data for phosphate at any of the three pilot landfills.
- There are (incomplete) measurement data for total phosphorous (P) at the Braambergen and Wieringermeer landfills; there are none for the Kragge landfill. However, it is questionable whether total phosphorous concentrations can be used because a considerable share of this concentration is expected to be organically bound phosphorous and not phosphate.
- High assumed concentrations of phosphate in the model input will strongly influence the results of the model. The concentration used in the sensitivity analysis was probably on the high side with respect to reality.
- For the landfill database as a whole (all landfills), phosphate measurements are limited, and those that are available are incomplete and not reliable (they contain extreme values and it is usually not clear whether it is P phosphorous or PO4).
- The concentrations of phosphorous given in the landfill database (including all the other landfills) show a strongly linear connection between total phosphorous and DOC. On the basis of the Redfield composition of organic material (including the ratio of carbon (C): phosphorous (P)) it can be expected that most of the concentrations of phosphorous consist of organic phosphorous (part of DOC) and that the concentrations of free phosphate are very low (see Table A6.1).
- In the sensitivity analysis, a concentration of 2.6 mg P/L is used as phosphate, corresponding to 8 mg PO4/L ($8.4e^{-5}$ M PO4). This value was derived from the average of the median values of total phosphorous in the Braambergen and Wieringermeer pilots and the overall median of total phosphorous, whereby total phosphorous was converted to phosphate.
- The Redfield composition based on measured and estimated DOC overestimates the amount of phosphorous. This is not a result of an overestimation of DOC concentrations from COD (chemical oxygen demand) (see Figure A6.2); DOC is considerably well estimated.
- A generic concentration for phosphate should therefore be estimated on the basis of (ortho)phosphate measurements. These are not sufficiently available

for the pilot landfills. So the median value of the measurements from the landfill database was used (see Table A6.1).

Conclusion

- For phosphate a generic concentration can be chosen that is equal to the median measured concentrations of phosphate in all landfills. This is 0.13 mg P/L, equal to 0.4 mg PO4/L, equal to 4e^{-6} M PO4.
- The advantage of choosing the median value is that this is not influenced by the relatively small number of (possibly unreliable) high concentrations among the available phosphate measurements, which we cannot be sure are total phosphorous measurements.
- The original concentration of phosphate in the modelling sessions was 0.05 mg PO4/L (~ 5e^{-7} M). The new concentration is thus eight times higher than the original concentration.
- The new concentration of 0.4 mg PO4/l is 20 times lower than the concentration from the sensitivity analysis (8 mg PO4/L). As a result of this, considerably smaller effects on the other substances such as arsenic and cyanide are expected than would be expected on the basis of the sensitivity analysis.

Table A6.1: Data on total phosphorous and phosphate from the landfill database.

	P measured (µg/L)	**PO4 measured (µg P/L)**
n	957.0	263.0
min.	10.0	0.1
max.	300,000.0	10,768.4
average	1995.4	1226.2
95%	6300.0	5514.7
50%	342.0	128.9

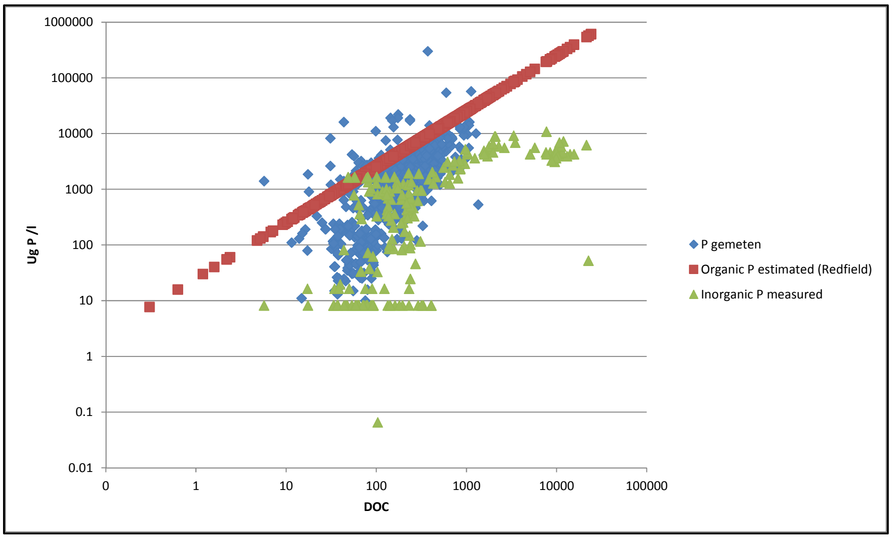

Figure A6.1: Available samples of total phosphorous and phosphate ('inorganic phosphorous'), both expressed as µg P/L. The concentration of organic P is also presented in accordance with the Redfield ratio (amount of phosphorous in organic matter). Based on the figure it is expected that total P mainly consists of organic phosphorous. For the measurements of phosphate it is not always clear if the sample is really phosphate or if it is total phosphorous (especially at high concentrations).

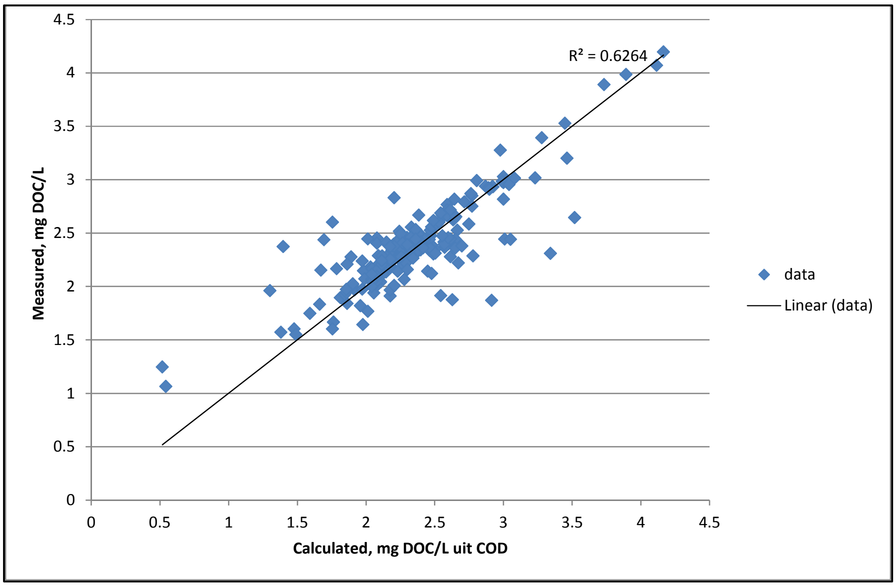

Figure A6.2: Measured DOC versus calculated DOC concentrations (from COD) for all landfills on the database.

Appendix 7: Derivation of landfill-specific dilution factors

A7.1 Site-dependent dilution factors

For a complete modelling of the dilution factors in groundwater, a lot of detailed data are required. Obtaining these data is time-consuming and expensive. The estimation of a dilution factor is therefore limited to a rough per pilot landfill.

The following assumptions were made:
- net precipitation 300 mm/year;
- complete failure of bottom liner (worst case);
- that dilution takes place over the entire thickness of the aquifer (landfill-specific);
- that there is a stationary state and full mixing of the leachate over the full height of the aquifer at 20 metres downstream of the landfill ($POC2_{gw}$).

The following literature was consulted to determine the dilution factors: Gronert (2010) for the Braambergen pilot landfill, Anonymous (2008) for the Kragge pilot landfill and Van Someren (2013) for the Wieringermeer pilot landfill.

A7.1.1 Braambergen pilot landfill in Almere

The Braambergen pilot landfill is located in a forested area on Pleistocene sedimentations east of Almere harbour (Gronert, 2010). The contour image in the report of Gornert (2010) shows that there is a more or less semi-circular run-off of the groundwater in a south-westerly to south-easterly direction from the first aquifer (medium deep). In this report it is assumed that any substance will disperse mainly in this aquifer.

Magnitude of leachate from the landfill to $POC1_{gw}$:
The landfill compartments 3, 4 and 5 are excluded from active treatment and these landfill compartments are therefore also excluded in the derivation of a dilution factor.
Area of landfill compartments 6–10: 89,074 m^2
Area of landfill compartments 11 and 12: 92,044 m^2
Total area : 186,474 m^2
Net precipitation: 300 mm/y

Total volume of leachate from the landfill compartments (the interface of the saturated and unsaturated zones under the landfill) (J1):
$J1 = 300/1000 \times 186{,}474/365 = 153\ m^3/day$

Magnitude of flow at POC2$_{gw}$:
The total length of the flowpath towards POC2$_{gw}$ (20 metres downstream of the landfill) at 20 m from the outside of the landfill along the SW, SE and NE edges can be estimated from the map in appendix 1.3 (Gronert, 2010) as approximately 1800 m.

The gradient in the aquifer was determined on the basis of the contour image added to the report, approximately in the middle of the SE edge between the -4.3 and -4.5 contours. The contours run approximately semi-circularly in accordance with the SW, SE and NE edges of the landfill.

Thickness of aquifer:	10 m
kD value:	200 m^2/day
k value:	20 m/day

Gradient: i = 0.2/(0.01 x 10000) = 0.002 m/m = 2 m/km

Specific flow rate in the aquifer:
Q = k x i = 20 x 0.002 = 0.04 m/d

Where:
Q = flow rate (m^3/m^2/day)
k = k value (m/day)
i = gradient (m/km)

Total flow at POC2$_{gw}$ (J2):
J2 = 1800 x 10 x 0.04 = 720 m^3/day

Calculation of dilution factor taking the background concentration into account:
Assumptions are full mixing over the total thickness of the saturated zone of the aquifer, and a stationary state. It follows from the law of conservation of mass for a dissolved substance:

Mass flow POC2$_{gw}$ = Mass flow POC1$_{gw}$ (from landfill) + Mass flow POC1$_{gw}$ (from laterally flowing water)

The mass flow per substance and location (POC1$_{gw}$ of POC2$_{gw}$) can be calculated as:

Mass flow of substance = (volume flow rate of water) x (concentration of substance in water)

It follows from this that:

J2 x C2 = J1 x C1 + (J2 - J1) x AW

Where:
C1= concentration of substance at POC1$_{gw}$ from the flux from the landfill (µg/L)
C2= concentration of substance at POC2$_{gw}$ (µg/L)
J1 = total magnitude of the outflow from the landfill compartments (m^3/day)
J2 = total flow at POC2$_{gw}$ (m^3/day)
AW= background concentration of laterally flowing water at POC1$_{gw}$ (µg/L)

With w (dilution factor) = (J2/J1) = (720/153) = 4.7 we can write this as:

<u>Calculation of dilution factor (C1/C2):</u>

(C1/C2) = w – (w-1) (AW/C2)

where (C1/C2) is the effective dilution factor at AW ≥0.

With a background concentration of zero (AW = 0) the dilution factor is (C1/C2) = w = 4.7.

The maximum applicable dilution factor for the Braambergen pilot landfill is (C1/C2) = 4.7.

A7.1.2 Kragge pilot landfill in Bergen op Zoom

The Kragge pilot landfill at Bergen op Zoom is located, according to the data from the geohydrological model Noord-Brabant in REGIS, in an agricultural area with a covering layer of Aeolian and fluvial sands 5–10 metres thick, known as the Formatie van Boxtel z2. The contours in the document titled 'Drawing of extrection filters old numbering' (source Attero) show that the groundwater flow in this phreatic package takes place mainly in a north-easterly direction. It is assumed that any substance will disperse mainly in the top phreatic aquifer.

<u>Volume of leachate from the landfill to $POC1_{gw}$</u>

The leachate from the landfill is:
Area of landfill compartments 1 and 2: 46,040 m^2
Area of landfill compartments 3–5: 110,000 m^2
Net precipitation: 300 mm/year
Total volume of the leachate to $POC1_{gw}$ (J1) Q_{POC1} = 300/365 x 0.001 x 110,000 = 90.4 m^3/day

<u>Magnitude of flow at $POC2_{gw}$:</u>
The rise height difference and the distance over which this difference occurs were also determined from this document.

Total length of the flow path towards $POC2_{gw}$ L_{POC2gw} = 435 m
Rise height difference dh = 1.5 m
Distance over which rise height difference occurs ds = 475 m
Gradient i = dh/ds = 1.5/475 = 0.003 m/m

The transmissivity (T) and the thickness (D) of the aquifer in the Kragge surroundings were determined on the basis of Regis.
T = 50–100 m^2/day
D = 5–10 m
k = T/D = 10 m/day

The specific flow rate in the aquifer: Q = k x i = 10 x 0.03 = 0.03 m^3/m^2/day

Total flow at $POC2_{gw}$ (J2):
J2 = L_{POC2gw}x H x Q = 435 x 2.5 x 0.03 = 32.6 m^3/day

Where:
H = thickness of the aquifer layer, 2.5 m (on the basis of Fugro measurements)

Calculation of dilution factor (C1/C2):

(C1/C2) = (J1 + J2)/J1 = (90.4 + 32.6)/90.4 = 1.36.

With a background concentration of zero (AW = 0) the dilution factor (C1/C2) is = 1.36.

The maximum applicable dilution factor for the Kragge pilot landfill is (C1/C2) = 1.36.

A7.1.3 *Points of special attention regarding the calculation method*

The concentration C1 is the concentration of the leachate that flows into the groundwater at $POC1_{gw}$, the interface of the saturated and unsaturated zones under the landfill. This is not equal to the concentration of the leachate from the landfill. When samples are taken, it is assumed that the concentrations are measured against a filtered sample and that in the soil, too, the particles dispersed in the leachate are largely (naturally) filtered out and thus do not contribute to the transport of substances far outside the landfill. The dilution factor therefore applies only to the dissolved metals. For metals that are bound to the particles dispersed in the leachate there are additional effects that limit the transport.

The leachate concentrations are in practice not constant over time (influence of wet and dry periods) or place (per landfill compartment and as a result of the formation of preferential channels and leaks in the bottom liner). Average values are used as a basis here.

A7.1.4 *Wieringermeer pilot landfill in Middenmeer: dilution factors and immission test*

Dilution in groundwater

The Wieringermeer pilot landfill is located in a situation with seepage. The dilution factor in groundwater was made equal to 1, becausethe leachate will infiltrate the groundwater as a result of pressure from the landfill, but at a relatively short distance from the landfill (maximum 750 m) it will surface again; this is shown by the study by Van Someren (2013) (see Figure A7.1). Over a distance of 20 m, mixing with the flowing groundwater will be virtually nil because of the seepage pressure from below, and the leachate will flow out virtually undiluted. These assumptions deviate from the scenarios for infiltration of groundwater used in the Braambergen and Kragge pilots and are determined by the specific situation around Wieringermeer.

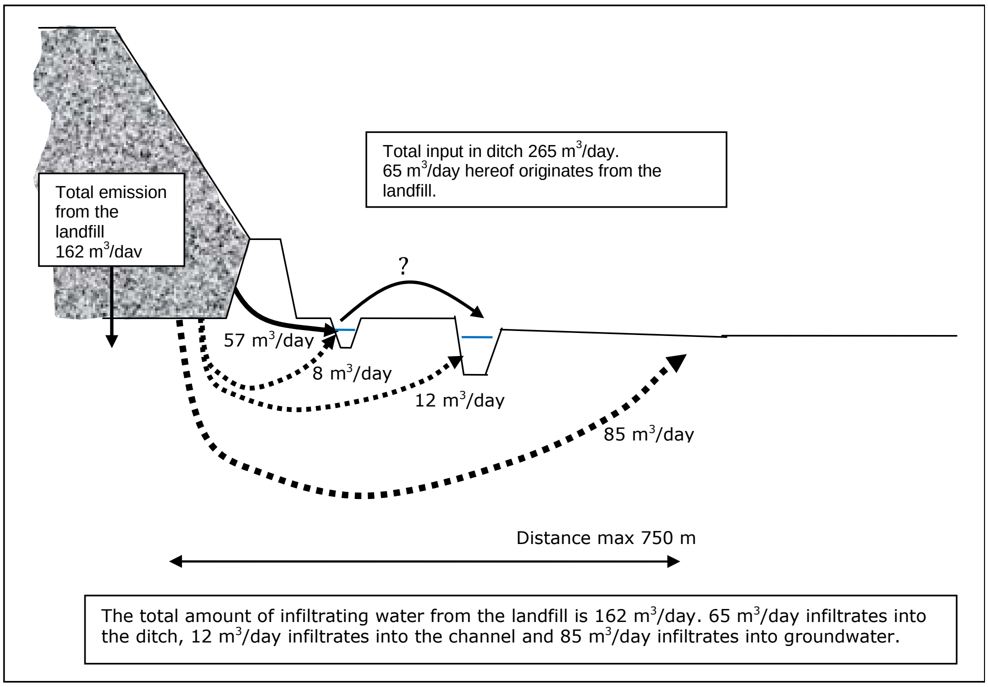

Figure A7.1: Schematic representation of the hydrological situation at the Wieringermeer pilot landfill, based on the research by Van Someren (2013). The diagram is not to scale.

Dilution in surface water

Dilution between ditch and channel

The leachate disperses via shallow flow (unsaturated zone) through the dike and ends up in the ditch surrounding the Wieringermeer pilot landfill. From the ditch it is discharged into the channel Westfriesche Vaart. Only a small amount of leachate will surface in the Westfriesche Vaart and this is considered to be negligible.

In accordance with the WFD’s emission-immission test (Ministry of I&M, 2011a; Kleissen, 2012), it is assumed that the ditch is part of the construction of the Wieringermeer pilot landfill and that the effects on the surface water are tested in the channel Westfriesche Vaart (POC2$_{sw}$). Figure A7.1 shows the steps in the assessment of the effects on the surface water. The allowed concentrations in the ditch are calculated using the immission-emission test. Table A7.2 shows the results of this in the ditch.

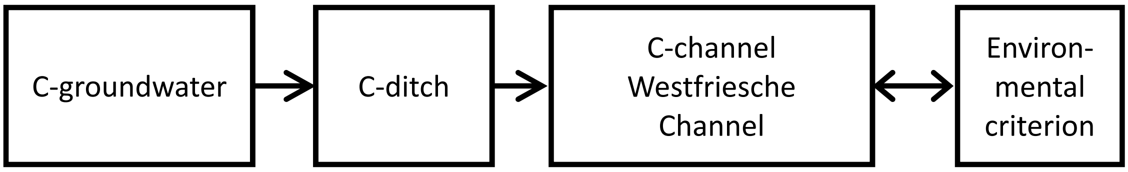

Figure A7.2: Schematic overview of the assessment of effects on surface water C stands for concentration.

Table A7.1: Allowed concentrations in ditch according to the immission-emission test of the WFD.

Substance	Allowed concentration in ditch (µg/L)
Metals	
Arsenic	44.9
Cadmium	0.38
Chrome	7.78
Copper	9.7
Mercury	0.089
Lead	10.8
Nickel	30.8
Zinc	20.6
Macroparameters	
Chloride (mg/L)	1053
Sulphate mg/L)	860
Ammonium (mg/L)	1.4
Phosphate (mg/L)	3.95
VOX	
Vinylchloride	0.17
Dichloromethane	27.2
1,1 dichloroethane	910
1,2 dichloroethane	13.6
1,1 dichloroethene	12.24
1,2 dichloroethene (cis,trans)	9.25
1,2 dichloropropane	364
1,3 dichloropropane	98.8
Trichloromethane (chloroform)	4.85
1,1,1 trichloroethane	28.56
1,1,2 trichloroethane	29.92
Trichloroethene (tri)	13.6
Tetrachloromethane (tetra)	16.32
Tetrachloroethene (per)	13.6
PAH	
Naftalene	4.66
Phenantrene	0.58
Antracene	0.19
Fluoranthene	0.19
Chrysene	1.75
Benzo(a)antracene	0.057
Benzo(a)pyrene	0.095
Benzo(k)fluoranthene	0.057
Indeno(1,2,3cd)-pyrene	0.0038
Benzo(ghi)perylene	0.0038
BTEX	
Benzene	13.6
Xylene	4.7
Toluene	96.2
Ethylbenzene	481
Cyanide	0.45

Substance	**Allowed concentration in ditch (µg/L)**
Phenols	130

Dilution between landfill and ditch

To determine the dilution of the leachate in the ditch, the data from Table A7.2 were used.

Table A7.2: Contribution of different surfaces to the total water flow into the ditch.

Different surfaces at Wieringermeer pilot landfill	**Surface area (ha)**	**Precipitation (mm/year)**	**Contribution precipitation to dilution in ditch**	**Flow (m^3/year)**
Seepage	75.4	36.5	100%	27,521
Arable land	17.9	300	100%	44,700
Business park run-off (built)	21.7	650	67%	94,504
Business park infiltration		300	33%	21,483
Wieringermeer East (built)	20	300	40%	27,600
Wieringermeer West (built)	10.8	300	100%	32,400
Wieringermeer West (not built)	5	300	100%	15,000
Total water flow into ditch	**75.4**			**263,208**

The dilution factor was then established on the basis of the expected leachate concentration coming from the eastern part of the landfill. The following formulas were used for this:

$$c_{R,j} = \frac{\sum_i c_{i,j} * Q_i}{\sum_i Q_i}$$

where:
$c_{R,j}$ = resulting concentration in the ditch (µg/L)
$c_{i,j}$ = concentration of substance *j* in the source *i* (or the various sections) (µg/L)
Q_i = flow rate of the various sources (or sections) (m^3).

The dilution factor was defined as the quotient of the substance concentration in the leachate *i* and the resulting substance concentration in the reservoir (or the water from the ditch).

$$D_j = \frac{c_{i,j}}{c_{R,j}}$$

where:
$c_{i,j}$ = concentration of substance *j* in the source *i* (µg/L)
$c_{R,j}$ = resulting concentration of substance *j* in the reservoir (or ditch) (µg/L).

The dilution factors for ammonium (10.3) and chloride (1.9) were derived from this. For the other substances it is currently impossible to establish a dilution factor as there is either no information or only limited information available (often organic and substances and macros parameters were not monitored), or detection limits were reported. This is the case for cadmium, copper, mercury, lead and zinc.

Appendix 8: Model output

This appendix presents the model output (for metals and macroparameters). In Sections A8.1–A8.3 the curves represent the time of arrival of the substance in groundwater at $POC2_{gw}$ at the Braambergen, Kragge and Wieringermeer pilot landfills. In Section A8.4 the curves represent the time of arrival in surface water at $POC1_{sw}$ at the Wieringermeer pilot landfill.

A8.1 The Braambergen pilot landfill

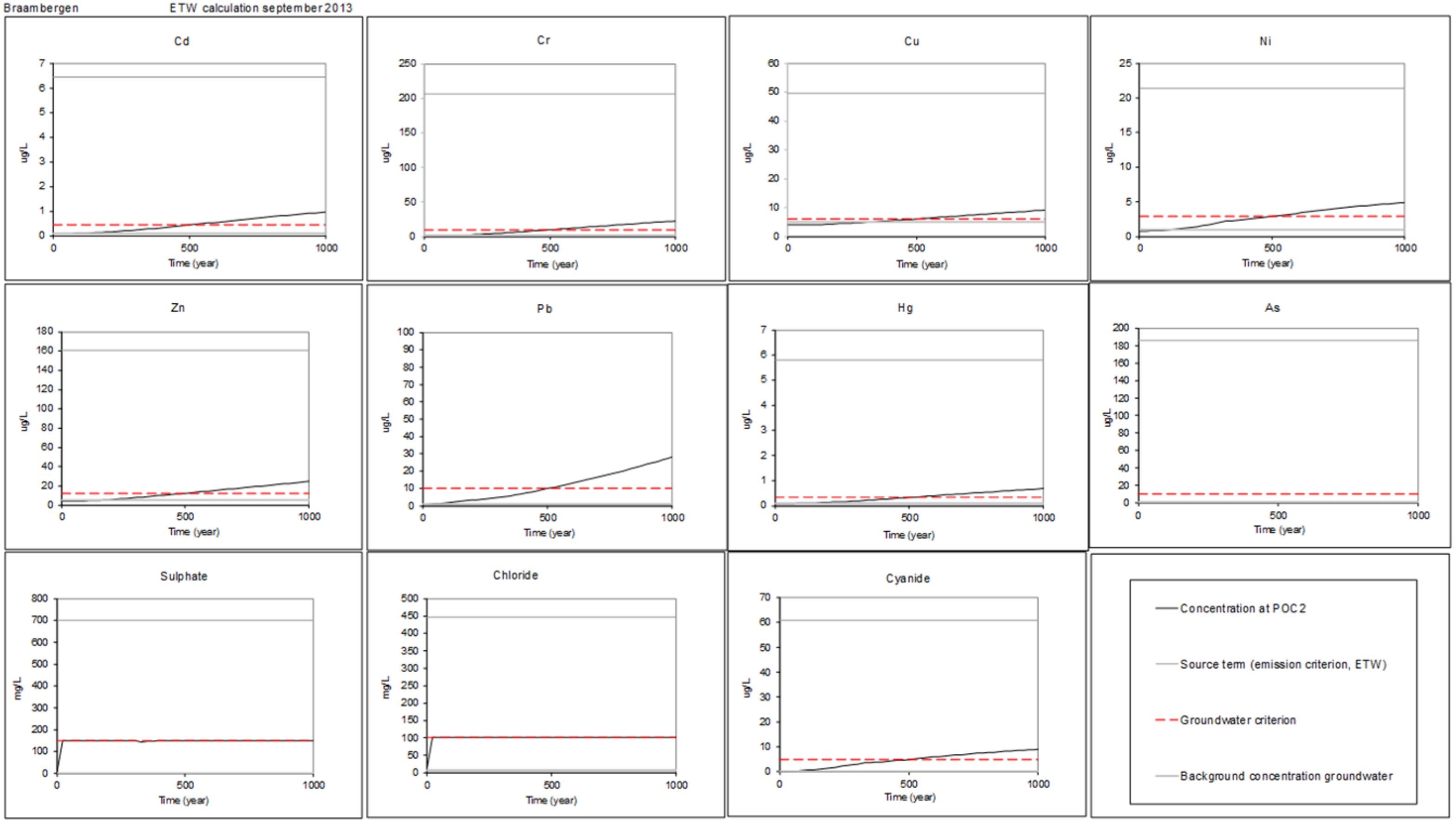

Figure A8.1: Time of arrival at POC2$_{gw}$ for metals and macroparameters at the Braambergen pilot landfill.

A8.2 The Kragge pilot landfill

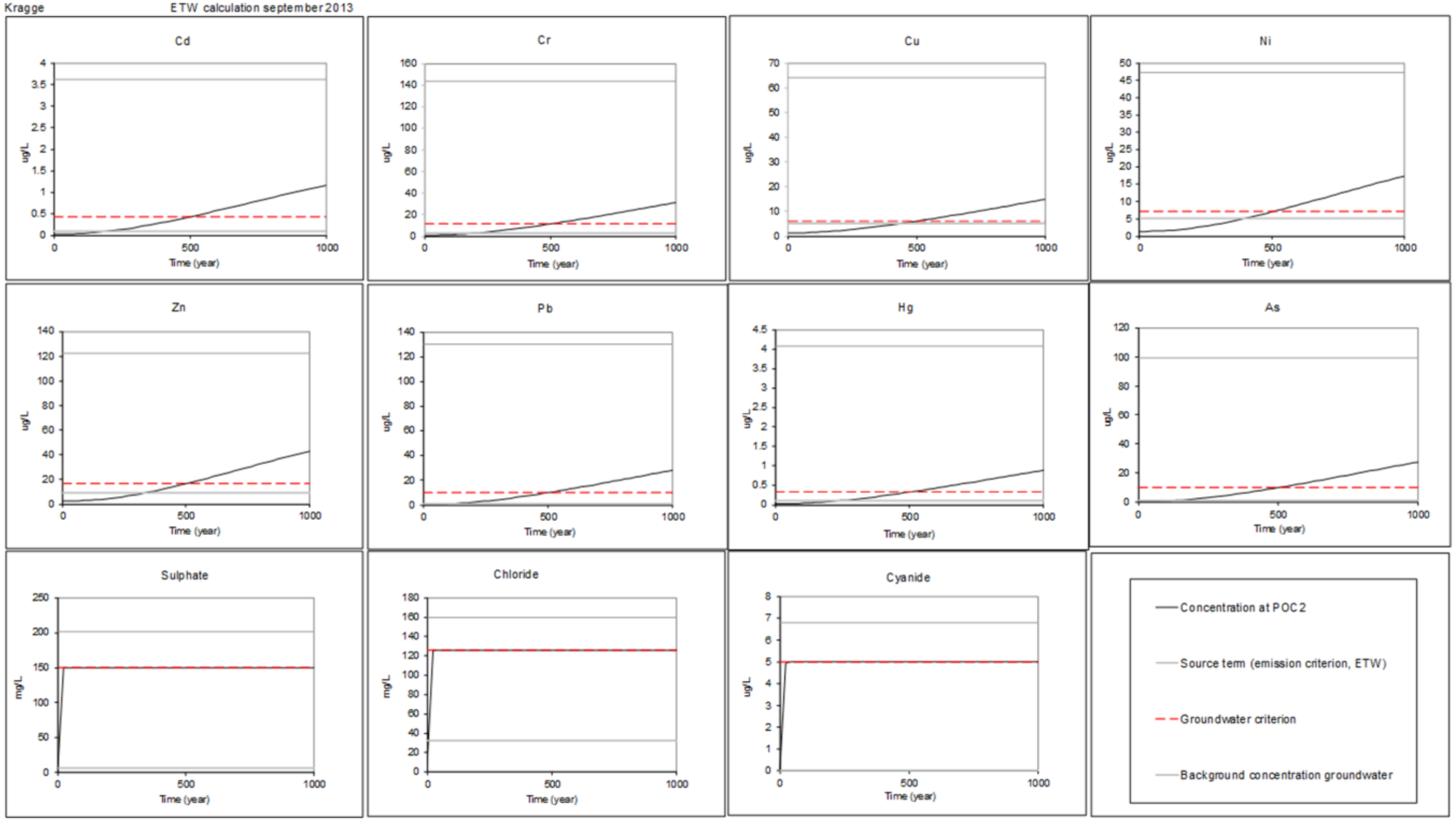

Figure A8.2: Time of arrival at $POC2_{gw}$ for metals and macroparameters at the Kragge pilot landfill.

A8.3 Wieringermeer pilot landfill – infiltration scenario

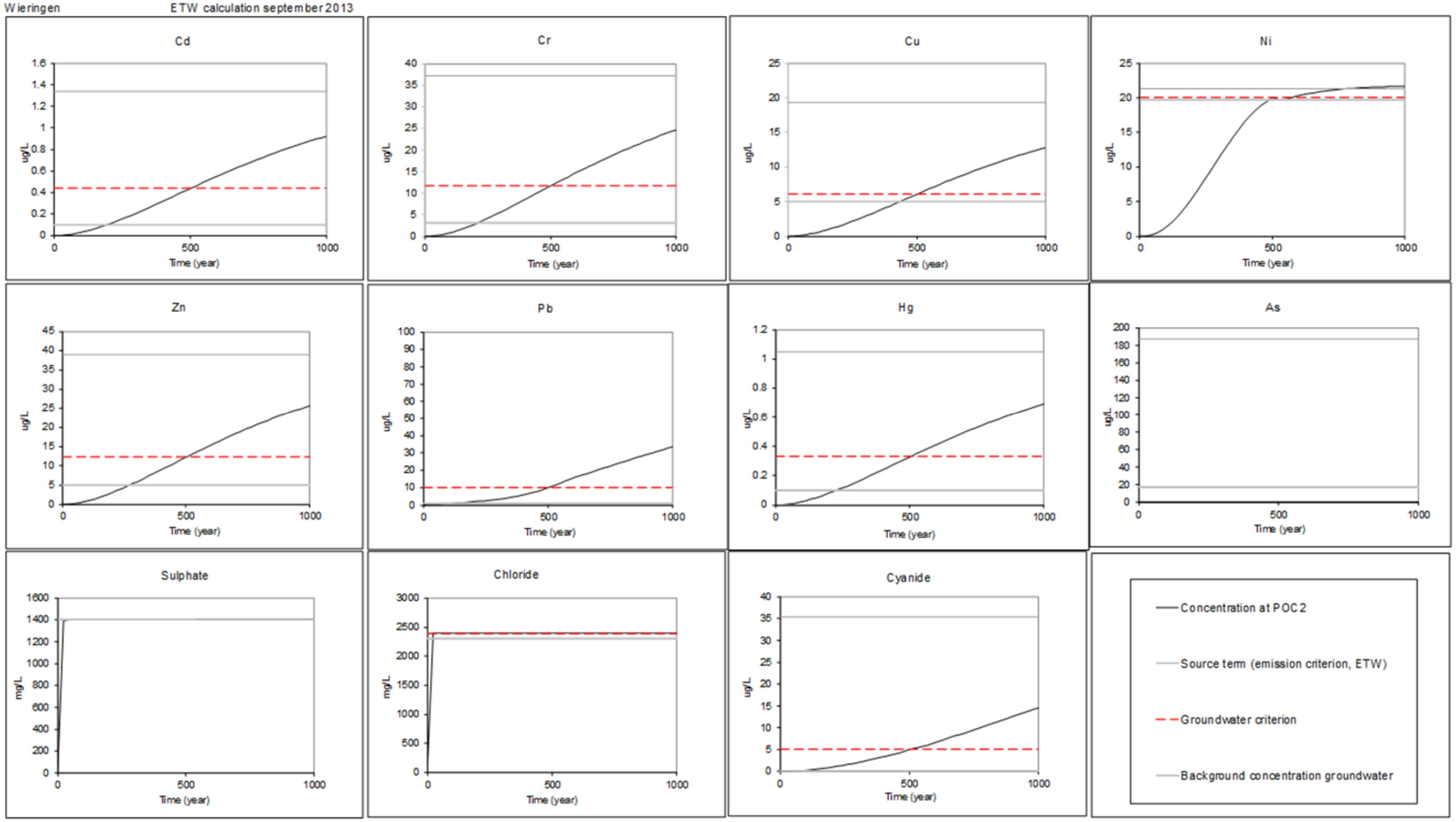

Figure A8.3: Time of arrival at POC2$_{gw}$ for metals and macroparameters at the Wieringermeer pilot landfill, infiltration scenario.

A8.4 Wieringermeer pilot landfill – seepage scenario

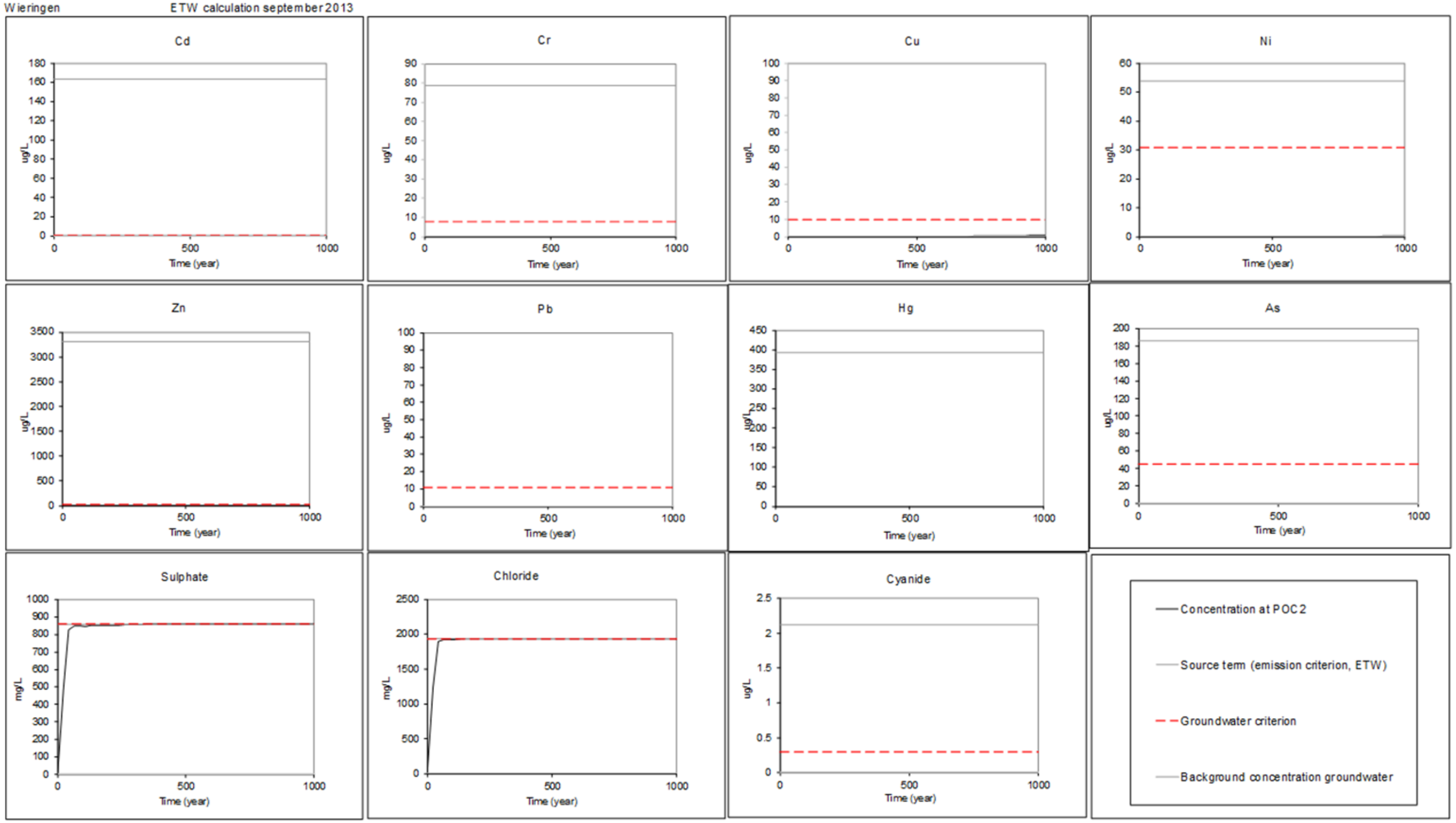

Figure A8.4: Time of arrival at POC1$_{sw}$ for metals and macroparameters at the Wieringermeer pilot landfill, seepage scenario.

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