

4 Methods used for health risk assessment

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4.1 Introduction

This chapter looks first at the existing legislation concerning drinking-water production, to illustrate the risks related to the use of wastewater as a drinking-water source. It then considers the main contaminants of concern in the practice of aquifer recharge and abstraction for drinking-water.

This chapter also describes in detail the two main approaches to health risk assessment. The first of these is the parameter approach, in which the estimation of the risk related to the use of water is based upon the presence of different parameters (i.e. chemicals and microorganisms). Every parameter is considered separately and its concentration is compared with a reference concentration. The parameter approach is based on either:

- water-quality standards — where the reference concentration is the one given by the drinking-water quality standards; or
- quantitative risk assessment — where toxicological data, and data on infectious doses and acceptable risk (chemical and biological) are taken as a reference.

A case study of quantitative health risk assessment in domestic water reuse is provided, to illustrate the practical application of this approach. This is followed by a detailed description of the main types of modelling technique used in risk assessment.

Finally, this chapter describes the approach of studying the effect of the water on test organisms or on the population. This ‘effects’ approach can involve using biological tests to look at how a water affects test organisms, cells or tissues, or epidemiological studies to examine the effects on a human population.

4.2 Relevant legislation

4.2.1 Legislation applying to groundwater, spring water and surface water

The main aim of legislation concerning drinking-water is to minimize the risks for water users. The production of drinking-water from groundwater or spring water is subject to very few regulations, because of the low risk to public health. The produced water must comply with the drinking-water standards, and many countries also have regulations on protection of drinking-water sources and on aquifers used for drinking-water abstraction.

Production of drinking-water from surface water is covered by more complex regulations, because of the higher health risk. Apart from the application of the drinking-water standards, regulations exist on the water source and on the minimal treatment to be applied to the surface water. An example is the European Community directive concerning production of drinking-water from surface water (Commission of the European Communities, 1975). The directive divides surface water sources suitable for drinking-water production into three classes, depending on their quality, which is assessed on the basis of 46 parameters and their concentrations. It also defines a minimal treatment, as a function of the quality. Production of drinking-water from good quality surface water (category A) requires only physical treatment (e.g. rapid filtration) and disinfection. Production from more polluted water (categories B and C) requires more sophisticated treatment

such as “break-point chlorination”, coagulation, flocculation, filtration, active carbon treatment and disinfection.

4.2.2 Legislation applying to recycled water

Very few countries intentionally produce drinking-water from recycled wastewater, and therefore, until now, no generally accepted reuse standards have been set. However, some interesting points concerning reuse legislation are described below.

It is logical that drinking-water produced from an aquifer containing recharged treated effluent should comply with the drinking-water standards. However, because effluent may contain a wide range of pollutants, an increased health risk can be expected in drinking-water from this source. To deal with the extra risk, the Californian Administrative Code, Title 22 (*Wastewater reclamation criteria*) sets standards on a list of chemicals that are not covered by the drinking-water standards, but that have an increased chance of being found in recycled water (State of California, 1978).

Drinking-water production from recycled wastewater almost always includes passage of the water through soil. The main reasons for passing the water through soil, which is considered a ‘polishing’ step, are to:

- remineralize the water
- create a lag time between infiltration and abstraction
- remove certain pollutants.

In recognition of the beneficial effects of soil passage, some regulations cover minimal soil passage conditions. An example of minimal aquifer treatment for reuse is given by the Californian legislation described above. Infiltration is allowed providing that a permit has been issued. The permit is for surface spreading at a given percolation rate (< 0.5 to < 0.8 cm/min), and specifies:

- the minimum retention time (6–12 months);
- the maximum percentage of recycled water in the extracted well water (20–50%);
- the horizontal separation between the recharge site and the abstraction well (165–330 m);
- the minimum depth to the groundwater (3.3–16.6 m)

The actual values depend on the soil type and are fixed on a case-by-case basis.

Guidelines from the United States Environmental Protection Agency (US EPA) and the Californian Title 22 both specify the minimal treatment that wastewater must undergo before aquifer infiltration. Also, both require the use of advanced wastewater treatment processes such as chemical clarification, carbon adsorption, reverse osmosis and other membrane processes, air stripping, ultrafiltration and ion exchange.

In countries where excessive groundwater extraction threatens to dry out aquifers, the use of infiltration of treated surface water is increasing. These countries have set standards on the water to be infiltrated, to protect aquifers and avoid the infiltration of hazardous components. Usually, the standards are developed for infiltration of treated surface water rather than for treated wastewater (Nederlands Staatsblad, 1993). Countries that allow infiltration of treated wastewater regulate the practice by permits, which are allocated case-by-case and granted for a limited time. Infiltration standards are set as a part of the permit.

4.2.3 Drinking-water standards relevant to aquifer recharge and abstraction

Drinking-water standards are country specific and are generally based on the WHO *Guidelines for Drinking-water Quality* (WHO, 1996). The WHO guidelines are currently being revised, with

publication expected in 2004. The European Community drinking-water directive 98/83/EC (Commission of the European Communities, 1998), based on the WHO guidelines, gives the minimal requirements with which the national standards of member states must comply.

Drinking-water standards are regularly adapted, to keep up with the latest information on negative effects of chemicals on health. Substances for which there is new evidence of harmfulness but no conclusions about the detrimental effect on human health are put on “priority pollutant” lists. These lists form the basis for research and for review of drinking-water standards. An example of a priority pollutant list is the US EPA *Drinking water Contaminant Candidate List* (CCL). A large number of the pollutants on priority lists are found in domestic wastewater.

Generally, revisions of drinking-water standards result in an increase of the number of parameters included. However, in the most recent revision of the Belgian drinking-water standards, the number of parameters decreased, although at the same time the new regulations specify that “the water supplier has to carry out analyses on the chemicals and microorganisms not covered by the standards but that can be present in the water and that can pose a risk to public health” (Flemish Government, 2002). These revised water quality standards are less useful as a reference for health risk assessment in relation to reuse.

The approach to controlling the presence of pathogens in drinking-water is changing. Classically, the microbial quality of water is assessed using microbial indicators. However, such indicators do not represent all pathogenic organisms, and faster and cheaper methods for analysis of microorganisms are becoming available, so that certain drinking-water standards now include pathogenic microorganisms. For example, the US EPA drinking-water standards contain regulations for *Cryptosporidium*, *Giardia*, *Legionella* and enteric viruses. The drinking-water standards from the Netherlands contain *Cryptosporidium*, *Giardia* and enteric viruses. The US EPA standards prescribe raw-water monitoring for these pathogens, where surface water or groundwater under the direct influence of surface water is being used. These standards also impose a minimal removal or inactivation of these organisms during the drinking-water production. The standards from the Netherlands specify that the pathogen concentration has to be measured in the raw water, and that subsequent treatment must reduce the pathogen concentration in the drinking-water to such a level that it does not present a risk of more than 1 infection per 10,000 persons per year.

4.3 Contaminants of concern

Aquifer recharge and abstraction for drinking-water can cause a health risk; careful monitoring of the presence of contaminants is therefore essential. The contaminants that are covered by drinking-water standards are well known, they are clearly described and their detection and removal during water treatment is straightforward. As such, these contaminants represent less of a risk than contaminants that are not subject to drinking-water regulation. A very large group of chemicals and microorganisms, not covered by drinking-water standards, can be present in wastewater, and it can be difficult to determine which contaminants are of most concern. Generally, the chemicals of concern are those that are found in concentrations above or close to acceptable concentrations in drinking-water. Microorganisms of concern are those with a high impact on health (because of their low infective dose or because of the fact that they cause important disease outbreaks), and the most resistant and persistent ones.

Many of the contaminants to be dealt with in drinking-water production from surface water originate from wastewater. For this reason, it is interesting to look at research carried out in preparation for the review of drinking-water standards. The CCL (US EPA, 1998), discussed above in Section 4.2.3. This list, which is planned to be updated every 5 years, lists priority contaminants for drinking-water programme activities, including research, monitoring and guidance development. The research is very broad and includes measurement and occurrence in the environment,

determination of future standard values, health effects, removal during water treatment and analytical determination methods. The decision whether or not to include a pollutant in the list is based on a number of criteria, such as health effects and expected environmental concentrations.

4.3.1 Unregulated chemicals

The US EPA's CCL includes 50 chemicals and 10 pathogens. Most of the chemicals are organic compounds, mainly pesticides. Pesticides can certainly be classified under "contaminants of concern" in the case of aquifer recharge. These chemicals may not be expected to be important at domestic level, but are apparently found on a regular base in domestic wastewater (Flemish Environment Agency, 1999, 2000). Pesticides are used in private gardens and at municipal level, and can also originate from agricultural runoff, where small watercourses are connected to the sewer.

Much effort has been put into the standardization of pesticides in drinking-water. Certain standards such as the European Drinking Water Directive 98/83/EEC (Commission of the European Communities, 1998) and the European national standards derived from this directive are based on an indicator approach and do not really reflect acceptable concentrations from a health point of view. A concentration limit of 0.1 µg/l is set for individual pesticides, and the sum of the pesticides must not exceed 0.5 µg/l. Health based standards for a large number of pesticides are found in the WHO *Guidelines for Drinking-water Quality* (WHO, 1996). Because pesticides are present on a regular base and in low concentrations, exposure to these chemicals is generally chronic. The health risk from such exposure is difficult to assess, because data on acceptable doses for chronic exposure are scarce and the low concentrations involved are difficult to monitor.

4.3.2 Pathogenic microorganisms

The pathogenic microorganisms that can be found since the most recent revision in certain drinking-water standards (*Cryptosporidium*, *Giardia*, *Legionella* and enteric viruses) are also relevant components in case of aquifer recharge. More details about the presence of these pathogens are described in Section 4.5 of this chapter.

Another group of relevant pathogens are the noroviruses (previously known as "Norwalk-like viruses"), which belong to the calicivirus family. The noroviruses form part of the microorganism category of the CCL. These viruses have recently been identified as probably the main cause of epidemics of gastroenteritis in industrialized countries. In a study of 43 epidemics in the Netherlands, the presence of Norwalk-like viruses was demonstrated in 32 out of the 43 cases (Vennema et al., 2000).

Until now, no figures have been published on the infective dose of Norwalk-like viruses, but it is believed that only a few virus particles can cause infection. Because of the low infectious dose and because of the fact that these organisms are ubiquitous in wastewater, the removal required during wastewater treatment for aquifer recharge is high. If drinking-water were produced directly out of raw wastewater, a total removal of up to 12 log (base 10) would be required for viruses, in order to reach the generally accepted infection risk level in drinking-water of 1 infection per year per 10,000 people.

Other organisms of concern in relation to reuse of wastewater are the opportunistic pathogens. These organisms are not pathogenic for healthy individuals, but can easily infect individuals with decreased immunity, such as the elderly and infants. Examples of opportunistic pathogens found in the contaminant candidate list are microsporidia, echoviruses, coxsackieviruses and *Mycobacterium avium* complex (MAC — i.e. *M. avium* and *M. intracellulare*) (Lechevalier, 1999a). WHO is currently reviewing the public health issues concerning selected water-related emerging pathogens, including MAC.

4.3.3 Priority pollutants

Worldwide, there are numerous lists of hazardous substances or priority pollutants, such as those from the European Water Framework Directive 2000/60/EEC (Commission of the European Communities, 2000), the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention) (OSPAR, 2002) and the list of persistent organic pollutants (POPs) drafted by the Stockholm Convention (UNEP, 2002). The pollutants on these lists have been selected because of their high toxicity to the environment and to aquatic organisms. The selection of the most dangerous pollutants out of the hundreds or thousands that can be found has been carried out using prioritization models based on expected environmental concentrations, toxicity, persistence or bioaccumulation.

The different lists have similar purposes, they aim to reduce or stop the emission of certain substances into the environment, and to reduce or stop the production of certain compounds. At first sight, the pollutants covered by these lists are not highly relevant to aquifer recharge, because their selection was based on their toxicity to the environment and aquatic organisms rather than to humans. Doses for acceptable intake are not necessarily the same for humans as for aquatic organisms. However, a reduction in the emission of hazardous and priority pollutants will be beneficial for public health in general by reducing the probability that these pollutants will end up in drinking-water sources.

4.3.4 Pharmaceuticals

Several pharmaceuticals and their metabolites have been found in raw wastewater, in surface water and even in drinking-water (Derksen & De Poorter, 1997; Ternes, 1998). These products originate mainly from human excretion. The concentrations measured up to now are well below those relevant from a health risk point of view, although some exceptions may occur as discussed earlier. In the case of direct reuse of domestic wastewater, specific pharmaceuticals should be removed, in order to avoid risks caused by these substances.

4.3.5 Natural hormones

Natural hormones also give cause for concern. The most powerful estrogen excreted by vertebrates is 17 β -oestradiol, a metabolite of estrogen, which is a female hormone that is excreted by both males and females (Blok & Wösten, 2000). This hormone has an effect on aquatic organisms at concentrations as low as 10^{-4} $\mu\text{g/l}$ (Ghijsen & Hoogenboezum, 2000). Natural hormones are assumed to cause an effect in humans only at higher levels; however, to date, no threshold value for an acceptable daily intake (ADI) for humans has been fixed. Natural hormones are very important in the context of aquifer recharge and potable reuse. A treatment aimed at removing them from water to be used for aquifer recharge is therefore strongly recommended.

4.3.6 Endocrine disrupting chemicals

Endocrine disrupting chemicals, also called hormonally active agents, can influence the endocrine systems of certain organisms. A clear relationship has been found between the presence of endocrine disrupting compounds and developmental changes in a number of animal species; for example, in seals in the North Sea and in sea slugs in the Scheldt estuary (Ghijsen & Hoogenboezum, 2000). The biotests described in Section 4.9.1 for identifying endocrine disrupting compounds compare the activity of substance under test with the activity of 17 β -oestradiol.

At present, there is no conclusive evidence about negative effects of endocrine disrupting chemicals on humans. However, it seems likely that eventually a connection will be found between substances with endocrine effects and undesirable effects on humans (Ghijsen & Hoogenboezum, 2000). Because of the lack of evidence that these compounds affect humans and the lack of

knowledge about reaction mechanisms, threshold values and acceptable risk values, it may be premature to include them in the group of contaminants of concern. Nevertheless, the latest findings and research results should be closely followed and should be taken into account in the design of aquifer recharge treatment schemes.

4.3.7 Personal care products

Very few toxicological data are available for personal care products. Products used for personal care, such as soap, shampoo, cosmetics and shaving foam must be tested by the manufacturer for their toxicity (Hutzinger, 1992; Talmage, 1994). These tests are limited to what is considered “normal cosmetic use”, such as skin contact or accidental swallowing. Therefore, little or no data are available on chronic exposure through the digestive system.

Synthetic perfumes or musks give cause for concern because they are resistant to degradation and are fat soluble; therefore, they are regarded as persistent environmental contaminants. These chemicals have been detected in wastewater treatment plant effluent in the microgram per litre range (Heberer, Gramer & Stan, 1999). What this means in relation to human health is not clear, because no data are available on acceptable doses.

4.3.8 Surfactants

Ecotoxicological data for surfactants are readily available. The concentrations found in raw wastewater greatly exceed the no-effect concentrations for the aquatic environment. As is the case for personal care products, toxicological data on chronic exposure to surfactants or detergents by drinking-water are limited. The toxicity of surfactants has been tested on small mammals, and no carcinogenic, mutagenic, teratogenic effects or effects on the reproductive system have been observed (Hutzinger, 1992). Nonylphenol, a biodegradation product of the detergent nonylphenol ethoxylate is persistent and has been found to have an endocrine disrupting effect on fish, although the effects on humans have yet to be determined (National Research Council, 1998, 1999). Therefore, it can not be said with certainty whether or not surfactants should be included in the contaminants of concern. As is the case for personal care products and endocrine disrupting chemicals, research results concerning these compounds should be followed closely.

4.4 Use of water quality standards for health risk assessment

Comparing water-quality with existing standards is one of the two parameter-based approaches to assessing the health risk of recycled water. In this approach, the concentrations of chemicals and microorganisms prescribed by risk-based water-quality standards are compared with the concentrations in the recycled water. The advantages of using water-quality standards to estimate risk are that:

- complex analyses are not required;
- drinking-water regulations are easy to obtain;
- drinking-water regulations differ little from one country to another.

The approach is based on the assumption that water complying with the standards is safe, because drinking-water standards are made to protect public health. However, the reality is more complex, and it is important to bear in mind that drinking-water quality guidelines were not developed with wastewater reuse as the main application.

4.4.1 Microbial aspects of the water-quality standard approach

Generally, the presence of pathogens in drinking-water is regulated by testing for microbial indicators, to ensure that they are absent. Microbial indicators are used to search for contamination

of the water by faecal matter, indicating the possible (or highly probable) presence of pathogens. Indicators are used because it is easier to search for indicators of faecal pollution than to attempt to test for a wide range of pathogens in the water. A good indicator of faecal pollution should fulfil the following requirements (WHO, 1996):

- be present universally and in large numbers in the faeces of humans and warm-blooded animals;
- be readily detectable by simple methods
- not grow in natural waters;
- have similar properties to pathogens in terms of its persistence in water and its removal by water treatment.

The ideal indicator, meeting all these conditions, does not exist. Also, a major shortcoming in the context of health risks is that some pathogens are more resistant to disinfection than the indicator organisms, and thus may be present even though no indicator organisms are found. For example, water that has been disinfected will not necessarily be free of enteroviruses and the cysts of some parasites (e.g. *Cryptosporidium* and *Giardia*). This means that it is possible for water to comply with water-quality standards, yet contain pathogens and be unsafe. This issue is important in the context of reuse of domestic wastewater, knowing that this water is a major source of faecal pathogens (coliforms).

4.4.2 Chemical aspects of the water-quality standard approach

In the process of setting drinking-water standards for chemicals, different aspects are taken into account. There are certain hazardous chemicals for which the analyses are costly and time consuming; therefore, drinking-water standards are a compromise between the scientific health risk and the feasibility of water analysis. Standards are generally based on the assumption that natural water is used as the water source.

Domestic wastewater contains a large number of the priority pollutants mentioned in Section 4.3.3. It also contains many chemicals, such as those with endocrine disrupting activity, for which research into potential adverse effects on human health has yet to come to a conclusion. These chemicals are not covered by drinking-water standards.

Thus, although drinking-water standards can be used as a reference to estimate the health risk related to water use, it is important to be aware that such an approach may overlook certain risks.

4.5 Quantitative health risk assessment

As with the water quality standard approach, a quantitative health risk assessment is a parameter-based approach that involves studying each component in the water separately. It is based on:

- the presence of harmful substances and microorganisms in the water;
- acceptable and infective doses;
- estimations of the exposure of the water users.

Using these data, the health risk can be calculated and compared with the risk that is agreed to be acceptable. This method allows comparison of different treatment scenarios for reuse; it also means that risk calculations can be used to design the installation required to obtain a certain treatment level. Modelling techniques based on quantitative risk assessment allow quite accurate estimations of exposure and risk, provided the necessary input data are available.

The way in which people are exposed to contaminants in water will depend on how the water is used. Chemicals and microorganisms can be ingested orally, through skin contact or by inhalation of aerosols. Direct ingestion is the most documented and studied route, whereas research on the effects of exposure through skin or inhalation is restricted to specific chemicals (e.g. those present in personal care and hygiene products) or microorganisms (e.g. *Legionella pneumophila*).

A quantitative assessment of the overall health risk, taking into account the different exposure routes, is not feasible. First, the data relating to exposure by inhalation or through the skin are very scarce or often not available. Second, with different exposures occurring simultaneously, it is difficult or even impossible to find out which route gives cause to which proportion of the risk. Because of these difficulties, a quantitative health risk assessment is instead based on the most “risky” exposure, which is assumed to be oral ingestion. Thus, the assessment is based on data related to drinking-water, and makes the assumption that the water is only in contact with the body via the digestive system. The amount of water taken up is based on estimations. In the case of drinking-water, this amount is assumed to be 2 litres per person per day. In the case of reuse for industrial, irrigation or other uses (e.g. toilet flushing), the volume of water accidentally taken up is estimated.

Table 4.1 describes the different steps in a quantitative health risk assessment. The procedure was initially set up for evaluating the health risk of specific chemicals, but it can also be used for microbial contaminants. The discussion here distinguishes between chemical and microbial risk assessment, because of the fundamental differences between both groups.

Table 4.1 Steps in the risk assessment procedure proposed by the National Research Council (1998)

(a) Hazard identification, involving definition of the human health effects associated with any particular hazard.
(b) Dose–response assessment, involving characterization of the relationship between the dose administered and the incidence of the health effect.
(c) Exposure assessment, involving determination of the size and nature of the population exposed and the route, amount and duration of the exposure.
(d) Risk characterization or integration of steps (a)–(c), to estimate the magnitude of the public health problem.

4.5.1 Quantitative microbial health risk assessment

Implementing a quantitative microbial risk assessment requires the analysis of each of the steps shown in Table 4.1. For every step, data need to be collected, and problems and assumptions clarified.

Hazard identification for microbial contaminants

In a microbial risk assessment, hazard identification involves identifying pathogenic organisms that can be transmitted by treated recycled wastewater. The list of potential waterborne pathogens contains dozens of bacteria, viruses and protozoa. These organisms can be harmful directly (by causing infection) or indirectly (e.g. by releasing toxins). Because of the large number of potential pathogens, certain organisms must be selected for inclusion in the risk assessment. This selection is not straightforward. From a health point of view, it is best to calculate health risk based on the pathogens with the highest impact on public health, such as those known to cause epidemics or with a very low infective dose. From a technological viewpoint, it is best to select organisms with high persistence (survival outside the host, in the environment), and with the highest resistance to destruction or inactivation.

For a quantitative microbial risk assessment to be feasible, there is a minimum amount of data that must be available on the selected organisms, to allow the risk to be calculated. The data required are: infective dose, concentrations in raw wastewater and the percentage of the organism removed by different water treatment techniques. A lot of pathogens — mainly newly discovered ones — cannot be included in this type of assessment because the required data are lacking. This is the case for the 10 organisms included in the US EPA's CCL⁶ and the so-called “emerging pathogens”, such as the noroviruses (previously known as Norwalk-like viruses or caliciviruses). It is believed that only a few virus particles can cause infection, but no data on infective dose have been published up to now. Therefore, the noroviruses cannot be included in the assessment. Other organisms of concern in relation to reuse are the opportunistic pathogens. These organisms are not pathogenic for healthy individuals, but they can easily infect individuals with impaired immunity, such as the elderly or infants.

Dose–response assessment for microbial contaminants

The dose–response assessment step of a quantitative microbial risk assessment is aimed at determining the relationship between the ingested dose and the effect on health. In the case of pathogens, this relationship is characterised by the infectivity. A threshold concentration for pathogens, under which no infection occurs, does not exist. Infection is a phenomenon that needs to be expressed as a risk and the infective dose always has to be seen as a range, because of the variation in sensitivity of the population. In a large population group, the possibility exists that the ingestion of a single pathogenic organism can infect certain individuals. There are three ways to characterise infectivity:

- the ID₅₀ or the infective dose is the dose that causes infection in 50% of the persons exposed to the pathogen;
- the P_{inf}(1.0) is the probability of infection following the exposure to a single organism;
- the most complete form for characterising infectivity is with dose/response curves, giving the probability of infection as a function of the dose.

The values for infectivity are determined by exposing human volunteers to different doses of the examined pathogen, observing the effect and recording the infections. The criteria used for infectivity differ from one study to another. The following criteria have been used to determine that a person is infected:

- certain symptoms of illness are observed, with the symptoms to be defined case-by-case;
- antibodies are found in the blood or an increase in antibodies is observed;
- the pathogens are found in the stools, under a form to be defined, (e.g. cysts, oocysts or eggs).

It is clear that the infectivity measured depends on the way the infection is defined. A person can have, for example, antibodies in the blood without showing clear symptoms of infection. The magnitude of infective doses, expressed as the ID₅₀, is in the range of 1–200 oocysts for *Cryptosporidium*, 10–100 cysts for *Giardia*, 1–10 plaque-forming units for rotavirus and 10⁴–10⁷ colony-forming units for *Salmonella typhi*. Generally, viruses are the most infective agents. The large variation (up to a factor of 1000) found on the value of the infective dose for an organism is also an illustration of the fact that the determination of infectivity is not straightforward.

⁶ *Acanthamoeba*, adenoviruses, *Aeromonas hydrophila*, caliciviruses, coxsackieviruses, cyanobacteria (blue-green algae), other freshwater algae and their toxins, echoviruses, *Helicobacter pylori*, microsporidia (*Enterocytozoon* and *Septata*) and *Mycobacterium avium* complex (MAC).

Exposure assessment for microbial contaminants

For microbial hazards, exposure is assessed by estimating the amount and the duration of exposure to the pathogens. Pathogenic organisms are found in wastewater when disease carriers are present in the community. The duration of the exposure can therefore be assumed to be in the order of days to weeks. An individual's exposure is calculated from the concentration of the pathogens in the water and the volume of water consumed by the individual. Concentrations of pathogens are variable, but are at their highest during disease outbreaks. The volume of water consumed is generally estimated as being 2 litres per person per day. For microbial risk assessment, some sources only take the amount of unboiled water into account, estimated as being 0.25 litres per person per day (WHO, 1996). In the case of reuse for applications other than drinking-water, such as household water (e.g. for toilet flushing, garden watering and cleaning), the microbial risk is calculated in the same way as for drinking-water. The difference is in the water volume, which in this case is the water ingested accidentally. This volume is based on estimations.

Risk characterization for microbial contaminants

Data on the amount of pathogens to which individuals are exposed and the infective dose are used to calculate the risk of infection, which is then compared with the 'acceptable' risk of infection. Generally, a yearly infection risk of 10^{-4} (i.e. 1 person in 10,000) is considered acceptable.

Microbial risk calculations are carried out on a daily basis. Because of the inherent variations, published values for pathogen concentrations in wastewater cover a wide range. Risk calculations should be carried out with the highest reliable concentrations found, even if these concentrations are temporary.

4.5.2 Quantitative chemical risk assessment

Hazard identification for chemical contaminants

In the case of a chemical risk assessment for wastewater reuse, the hazard identification consists of finding the components in the water that are hazardous and are present in sufficiently high concentrations to adversely affect health. Much research is being carried out into the harmful effects of chemicals on humans and on the environment. In the past ago, new chemicals were released into the environment without knowledge of their consequences for men and nature. Today, manufacturers must examine the toxicity of newly developed chemicals before commercial release. In addition, international scientific organizations have been created to identify and screen potentially dangerous chemicals, in order to minimize their harmful effects. As a result, there are now numerous lists of toxic chemicals and priority pollutants. A case study describing examination of the literature on chemicals and priority pollutants present in domestic wastewater is given below in Section 4.6.

Hazardous chemicals can be divided according to their mechanism of action and their effect. Apart from some small groups of chemicals with very specific effects, chemicals can be divided into three main groups:

- toxic chemicals — these have a wide range of different effects, which vary from one chemical to another;
- carcinogens — substances that induce mutations, possibly followed by cancer, they are classified as either genotoxic or nongenotoxic, depending on their mode of action;
- endocrine disrupters or endocrine disrupting compounds — substances believed to interfere with the functioning of the endocrine system, the system that regulates the development, growth, reproduction and behaviour of human beings and wildlife.

To date, little is known about the effects of endocrine disrupting chemicals on humans, although they raise serious concerns. The data necessary to include these chemicals in a quantitative chemical risk assessment are not yet available.

Dose–response assessment for chemical contaminants

Data on the detrimental effect of chemicals are mainly acquired through research on test organisms. Studies of accidental spills of chemicals and of accidents leading to exposure of populations can also contribute to the understanding of the toxicology of chemicals. Such studies, combined with toxicological data on test organisms, are used to determine the acceptable dose for humans. To ensure that the risk is acceptable, a large uncertainty factor is introduced when determining the acceptable dose in this way. The uncertainty factor takes into account any interspecies and intraspecies variation, the nature and severity of effect and the adequacy of the studies (WHO, 1996). The total uncertainty factor is the product of each of these uncertainties, and it should not exceed 10 000. If the uncertainty factor is above this figure, the threshold value below which no risk exists cannot be fixed. Acceptable doses are calculated on a daily basis and are generally expressed as acceptable daily intake (ADI), tolerable daily intake (TDI) or reference dose (RfD)⁷.

Chemicals classified as “toxic” are not harmful under a certain threshold value; the same applies to nongenotoxic carcinogens. However, genotoxic carcinogens are believed to have no threshold value, which means that there is a probability of harm at any level of exposure. The harmful effect of genotoxic carcinogens is expressed as a risk of producing tumours. Determining whether a chemical is a carcinogen or not is a long process involving a lot of research, and it is not always possible to conclude with certitude that a component is a carcinogen for humans. Because of this uncertainty, the International Agency for Research on Cancers (IARC) has classified carcinogens according to their potential carcinogenic risk to humans. The classification divides substances into four groups, based on the certainty and uncertainty of being carcinogenic (IARC, 1987).

Exposure assessment for chemical contaminants

In a quantitative chemical risk assessment, the elements that are important in assessing exposure are the amount, the duration and the frequency of exposure. In relation to reuse of wastewater as a drinking-water source, a distinction is made between acute exposure (high concentration and short time) and chronic exposure (low concentration and long time). In the exposure assessment for domestic wastewater, acute exposure is not taken into account because such exposure through wastewater is mainly due to accidents and therefore cannot be predicted. In relation to reuse of wastewater, chronic exposure to household chemicals that are present permanently in wastewater in low concentrations probably represents the highest risk. Unfortunately, data on chronic-exposure toxicity are the most difficult to obtain, because of the low concentrations and long test periods required. Chronic or subchronic toxicity tests can last several years (Sekizawa et al., 2000). ADI values for chronic exposure are extrapolated from the highest dose or concentration that causes no detectable adverse health effect (no-observed-adverse-effect level, NOAEL) on test organisms. If no NOAEL is available, data on the lowest observed dose or concentration at which there is a detectable adverse health effect (lowest-observed-adverse-effect level, LOAEL) can be used instead (WHO, 1996). ADI values for chronic exposure are only available for a small number of chemicals. The exposure to hazardous chemicals is calculated from their concentration in the water and is based on a water consumption of 2 litres per day.

⁷ The reference dose is the estimate of the daily exposure to the human population that is likely to be without appreciable risk of deleterious effects over a lifetime (Dictionary of terms, 2002).

Risk assessment for chemical contaminants

The risk assessment is carried out by comparing the daily exposure with the acceptable daily intake. For genotoxic carcinogens, the exposure is compared with the dose that corresponds with a lifetime cancer risk of 10^{-5} , which is set as the “acceptable risk”.

Apart from water, two other important sources of exposure to pollution are food and air. The drinking-water standards are fixed in such a way that the sum of the three sources does not exceed the ADI. If the contribution from water is unknown, 10% of the ADI is allocated to water, with the other 90% being accredited to food and air. By allocating such a low percentage to water as a source of pollution, the standard value for water will be low, which adds extra safety to the standard (WHO, 1996).

4.6 Case study of quantitative risk assessment applied to household wastewater

This section describes a quantitative health risk assessment applied to the reuse of wastewater from domestic origin for production of potable water. For reasons explained below, such an assessment does not reflect the absolute risk; rather, it gives a theoretical indication of the health risk. The concentrations of hazardous substances and organisms still need to be checked in the produced water. Nevertheless, a quantitative risk assessment makes it possible to identify weak points in the treatment or components likely to cause the highest health risk; it also allows comparison of treatment processes and drinking-water sources.

The selection of the microorganisms for the assessment was based on the following criteria:

- impact on health (low infective dose, occurrence of epidemics);
- high persistence and resistance;
- availability of sufficient data on infective dose;
- concentrations in raw wastewater;
- removal during treatment.

The selection had to contain at least one representative of each main group of pathogens. The following pathogens were selected: enteroviruses, *Salmonella typhi*, *Cryptosporidium parvum* and *Giardia intestinalis*.

The selection of the chemicals to be included was based on the following criteria:

- the chemicals must be specific for domestic wastewater;
- data on the harmfulness of the chemicals must be available in the form of ADI, TDI or Rfd;
- the concentration of the chemicals in domestic wastewater must be known, and must be high enough to represent a risk to human health.

Data from measurements on domestic wastewater show that the metals most likely to cause a problem are generally lead, arsenic, nickel and cadmium. Domestic wastewater is also likely to contain fairly high concentrations of copper and zinc, due to leaching from piping and fittings; however, because the ADI values for copper and zinc are relatively high, these metals do not represent a risk for humans in average wastewater, although they may affect organisms in the environment.

Nitrogen and its components are found in domestic wastewater in concentrations above those considered acceptable in drinking-water. This is especially the case for nitrite ions and, to a lesser extent, for nitrate ions. Nitrogen components originate from proteins and urea, which are ubiquitously present in domestic wastewater. Nitrite is included in the quantitative risk assessment.

The presence of chemicals in grey water⁸ is the subject of extensive research, mostly in relation to reuse. Grey water can contain hundreds of compounds, and the literature suggests that few of them meet the criteria given above for including chemicals in a quantitative risk assessment. The main problem is that the concentrations are low and are therefore difficult to measure, and that data on toxicity are scarce. Only three groups of chemicals in grey water meet the criteria set: pharmaceuticals, pesticides and dioxins.

A study on the presence of pharmaceuticals in surface water, based on sales figures, showed that nine pharmaceuticals⁹ might be present in wastewater in such a concentration that they could conceivably present a health hazard. For the quantitative risk assessment, the pharmaceutical acetylsalicylic acid was selected, because data on ADI, concentrations and removal were available for this chemical.

Measurements on pesticides in effluent from wastewater treatment plants reveal quite high pesticide concentrations (Flemish Environment Agency, 1999, 2000). The sum value of atrazine, simazin and diuron exceeds the acceptable concentration for “total pesticides” in drinking-water given in the European Drinking Water Directive 98/83/EC (Commission of the European Communities, 1998). Therefore, the parameter “total pesticides” was selected for the risk assessment.

Dioxins have been detected by in domestic wastewater and urban runoff (Horstman & McLachlan, 1995) at the picogram toxicity equivalent (TEQ¹⁰) level per litre. These dioxins are believed to originate from textiles (Horstmann & McLachlan, 1995). With the TDI value of 1–4 picogram per kilogram body weight (WHO, 1996), and the assumption that 10% of the total intake is allocated to drinking-water, the concentration in wastewater exceeds the acceptable concentration for drinking-water. Thus, dioxins meet all the criteria set for chemicals and were therefore selected for the quantitative health risk assessment.

Table 4.2 summarizes the selected organisms and chemicals, showing their maximum concentrations in domestic wastewater, the concentrations allowed in drinking-water and the required removal, expressed as the logarithmic reduction ($\log_{10}(C_0/C)$). The concentration allowed in drinking-water is taken from drinking-water standards (if available), or otherwise is calculated using ADI or ID₅₀ values, assuming a water consumption of 2 litres per person per day for the chemicals and 0.25 litres per person per day for the pathogens.

Among the natural estrogens, surfactants, solvents, musks and other endocrine disrupting chemicals, no compounds could be found that satisfied all the criteria for inclusion in the assessment, mainly due to the absence of data on chronic toxicity for humans. However, some of these compounds, such as natural estrogens (e.g. 17 β -estradiol) and biodegradation products of surfactants (e.g. nonylphenol), have a proven harmful effect on test organisms.

Table 4.2 gives the required removal for each component. Usually, one treatment step will be insufficient to reach the concentration acceptable for drinking-water; rather, an array of treatments will be needed, each one removing a certain percentage of the pollutant or pathogen. The total removal by the array of treatments is equal to the sum of the removal efficiencies — expressed as \log_{10} values — of the different treatment steps. This method of combining log values of removal efficiencies to obtain the total removal of the treatment process has been followed in the quantitative health risk assessment described here.

⁸ Grey water is the wastewater from showers, baths, hand basins, laundry tubs, washing machines, dishwashers and kitchen sinks, it does not include water from toilets.

⁹ Progesteron, paracetamol, chloorhexidine, povidon iodine, erythromycin, doxycyclin, acetylsalicylic acid, clavulaan acid and sulfamethoxazol.

¹⁰ Toxicity equivalent in comparison with 2,3,7,8-trichloro-dibenzo-p-dioxine.

Table 4.2 Summary of organisms and chemicals selected for quantitative health risk assessment

Component selected	Concentration			Log ₁₀ removal		
	Raw wastewater (maximum)	Allowed drinking-water	In WWTP effluent	Total required	By WWTP	Required after WWTP
Lead	0.045 mg/l	0.01 mg/l ^e	0.007 mg/l	0.65	0.80	0
Nitrite	na ^a	0.2 mg/l (chronic exposure) ^d	3.9 mg/l	na	na	1.3
Acetylsalicylic acid	314.5 µg/l	17.5 µg/l ^f	59.8 µg/l	1.25	0.75	0.5
Dioxins	14 pg/l	3 pg/l	1.4 pg/l	0.67	1.0	0
Total pesticides	na ^b	0.5 µg/l ⁵	1.95 µg/l	na	na ^c	0.6
<i>Cryptosporidium parvum</i>	6 x 10 ⁴ oocysts/l	2.6 x 10 ⁻⁵ oocysts/l ^h	5.4 x 10 ⁴ oocysts/l	9.35	1.1	8.3
<i>Giardia intestinalis</i>	1.5 x 10 ⁴ cysts/l	5.5 x 10 ⁻⁶ cysts/l ^g	0.6 x 10 ⁴ cysts/l	9.4	0.4	9.0
Enteroviruses	3 x 10 ⁵ PFU/l	1.8 x 10 ⁻⁷ PFU/l ^g	1.56 x 10 ⁵ PFU/l	12.2	0.3	11.9
<i>Salmonella typhi</i>	8 x 10 ⁴ CFU/l	0.19 CFU/l ^g	8 x 10 ⁴ CFU/l	5.6	1.0	4.6

CFU = colony forming units; na = not applicable; PFU = plaque forming units; WWTP = wastewater treatment plant

^a The nitrite concentration in the influent has not been taken into account, since the concentration can increase during active sludge treatment

^b Data on pesticide concentrations in raw wastewater are not available

^c Unknown

^d WHO (1996)

^e Directive 98/83/EC (Commission of the European Communities, 1998)

^f Claeys & Van Hoof (2001)

^g Versteegh, Evers & Havelaar (1997)

^h Schijven & Hassanizadeh (2000).

Treatment technologies relevant to reuse that were studied in the quantitative risk assessment were:

- coagulation/flocculation/sedimentation
- rapid filtration
- soil passage
- active carbon filtration
- reverse osmosis
- microfiltration
- ultrafiltration
- disinfection (chlorination, ultraviolet or ozonation)
- slow sand filtration
- water collection basins
- constructed wetland
- lagoons
- ion exchange

- electro dialysis.

The literature was screened to find the removal efficiencies of these treatment technologies. In general, data on this subject are very scarce, and imprecise; for example, data on removal percentage found in the literature usually have a margin of 20–30%. Also, little information is available on variables influencing the removal, such as influent concentration or temperature.

The total \log_{10} removal for each parameter was calculated for some existing reuse installations. The calculation was divided into two parts: a worst-case scenario and an optimal scenario (based on minimal and maximal removal efficiency, respectively). In several of the reuse installations, the removal of pathogens in the worst case was found to be insufficient to achieve an acceptable level of risk (the highest risk of infection is caused by the enteroviruses). The real situation lays somewhere between the worst case and the optimal situation. Modelling techniques make it possible to simulate the real situation more closely.

Because the calculation showed that the highest removal is required for pathogens, it gives the impression that the highest risk is caused by pathogens rather than chemicals. However, the risks that are compared are very different, as described above in the sections on microbial and chemical health risk assessment (sections 4.5.1 and 4.5.2). The calculation also indicated that the removal required for chemicals is relatively small. Again, this conclusion needs to be looked at carefully because many chemicals had to be omitted from the assessment because insufficient data were available.

4.7 Model approach in quantitative risk assessment

The risk characterisation using worst-case default values described in Section 4.5, above, indicated that in several reuse installations the removal of pathogens may be insufficient to reduce the risk to acceptable levels. However, this conclusion resulted from deliberately calculating a high-end risk estimate. Overreliance on worst-case values can lead to the risk being so grossly overestimated that it is totally unrealistic. The high-end estimates result from the multiplication of high-end values for input parameters and variables (e.g. in the example in Section 4.6, maximum concentration of the hazardous agent in raw wastewater was multiplied by minimum removal efficiency at the reuse installation for each treatment step). The more variables for which high-end values are selected, the higher the likelihood of deviating from the actual worst-case situation.

The point of risk analysis is to determine risk levels as realistically as possible, and the number of variables involved is generally large. Therefore, the question arises: How best to integrate quantitative risk characterization into a model-based decision-making process?

This section sets out a systematic probabilistic approach that can add credibility to scientific interpretation and can help risk assessors and managers to achieve effective decisions and communication in situations of uncertainty where the stakes are high. It describes:

- an overview of the systematic approach
- an analytical approach for incorporating variability and uncertainty
- strengths and advantages of the approaches
- weaknesses and disadvantages of the approaches.

4.7.1 Systematic approach

As summarised in Table 4.1, quantitative health risk assessment involves hazard identification, exposure assessment, dose–response analysis and risk characterization. A useful way to efficiently

4 Methods used for health risk assessment

perform these complex tasks is to proceed in a stepwise fashion. A minimum of four phases should be included:

- screening
- desk research
- field research
- risk management scenario analysis.

These phases are described in detail below. At the end of each phase, risk–cost–benefit should be analysed; the results obtained may lead to reiteration of one or more preceding steps. The stepwise model approach is illustrated in Figure 4.1.

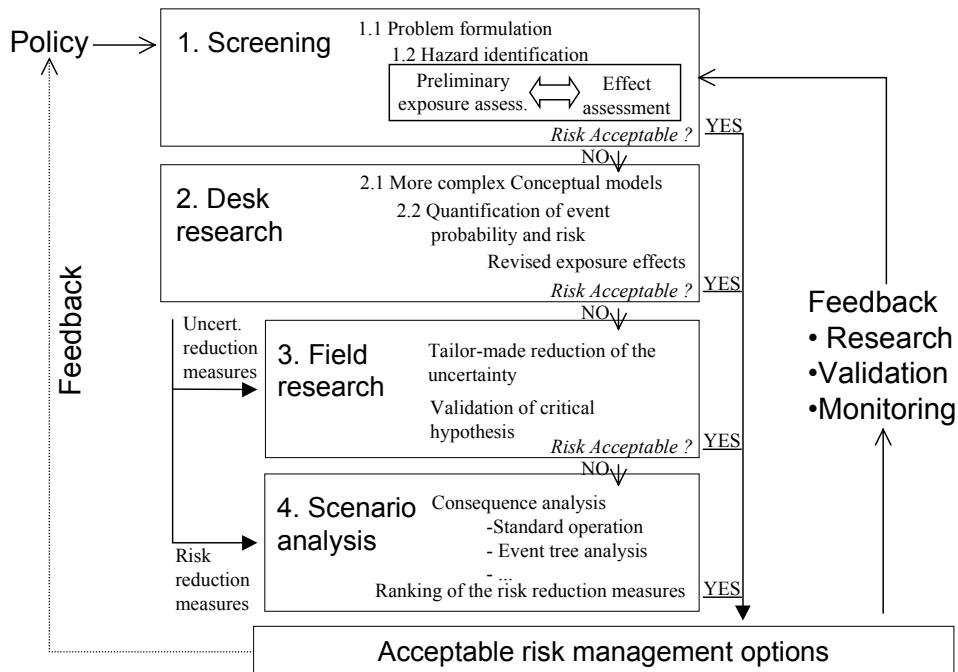


Figure 4.1 A phased approach to model-based risk characterization

Phase 1 — screening

The screening phase includes a formulation of the problem, an initial identification and ranking of hazardous substances, a list of possible risk management alternatives (uncertainty reduction and/or risk reduction technologies) and a “feeling” about possible worst-case outcome, however rough the estimation may be.

Problem formulation

Problem formulation answers the question “What are we trying to assess and why?” It involves setting general objectives such as acceptable levels of risk (e.g. probabilities of infection) and translating them into mathematical terms or objective functions so that they can be matched to technology appraisal or policy-making (e.g. nuisance minimization versus cost). Examples of objective functions for water quality planning are provided by Haimes and Hall (1975). Acceptable risk is very location-specific and therefore does not fit within international guidelines (Hunter & Fewtrell, 2001). Also, local legal and regulatory considerations influence how certain risks are regarded. For example, some local regulations consider an acceptable risk for pathogens to be 1 infection in 10,000 people a year, whereas others refer to the presence of microbial indicators.

Hazard identification

Hazard identification answers the question “What hazard should we include in the assessment?” This involves identifying agents that can potentially cause harm to human health, such as pathogens and harmful chemicals. Existing local regulatory standards or policy frameworks may determine which worst-case end-points are significant. Hazard identification cannot be limited to the evaluation of the substances at the source. For example, in the case study described above in Section 6, nitrite can potentially be hazardous. Although the source — domestic wastewater — contains virtually no nitrites, this chemical can be generated during activated sludge treatment if the denitrification process is incomplete. Therefore, hazard identification is intimately linked with the other steps of the analysis, particularly possible risk-management alternatives.

Screening modelling

As the potential hazards may be numerous, in the screening stage of the analysis the risk analyst might prefer to use relatively simple models to help identify potential hazards and promising options for more detailed study, as well as to eliminate options that are clearly inferior. For example, a screening of possible hazards may consist of simple release and exposure modules based on removal percentages, or semi-empirical modelling starting with the information available at hand, however imprecise that information may be. The assessment described in the case study of quantitative health risk assessment applied to household wastewater (Section 4.6, above), is an example of such screening. If the outcome, using worst-case default values, shows that the substance is not of concern, the assessment for that substance can be stopped. However, if the outcome shows that the substance is of concern, the exposure assessment needs to be refined and based on more realistic values.

Phase 2 — desk research

Once the possible hazards and risk reduction measures are clear, it may be helpful to obtain more information to reduce the uncertainty or narrow the range of technologies to be appraised by field testing (which is generally expensive).

Because they include so many simplifications, the models used in the initial screening phase cannot be expected to provide a truly optimal solution to risk reduction, particularly in view of the complex socioeconomic and institutional concerns involved. Uncertainty can be reduced by adopting more sophisticated conceptual models or by reducing uncertainty in the variables and parameters fed into the models:

Model complexity

The model structure, which could be anything from empirical to deterministic predictive, should be the simplest possible that can account for all the important factors involved in the risk management issue. Empirical models are simple, but have a high degree of indeterminacy. These models work by similarity from one case to another, which may be a problem because aquifer recharge cases may not be similar, so that only a long series of field experiments would be able to confirm or reject the results.

Deterministic predictive models involve developing an understanding of all elements in the system, so that the performance of the system can be predicted. Disadvantages of this type of model are their complexity and the fact that at present there are definite limits to what we can expect predictive models to achieve; in certain fields, the best we can hope for is to ascertain patterns.

The choice of an appropriate level of model complexity depends on multiple factors such as:

- local regulations;
- site-specific conditions (e.g. soil characteristics);
- possible technological alternatives (e.g. membranes, reverse osmosis and soil infiltration);
- the availability of, and gaps in, scientific knowledge relevant to the site-specific conditions;
- the availability of, or ease of acquisition of, field data.

Based on the precautionary principle¹¹, the most conservative conceptual model should be used in the analysis. For example, modelling of the bacteriophages MS2 or PRD1 is generally used to model the removal of pathogenic viruses, because these bacteriophages generally show the highest

¹¹ The precautionary principle is that where there are threats of serious or irreversible damage, lack of full scientific certainty should not be used as a reason for postponing cost-effective measures to prevent environmental degradation.

attachment, detachment and inactivation values (Schijven & Hassanizadeh, 2000). Depending on which model is applied, different results will be obtained.

Input and variables uncertainty

The calculations upon which the deterministic simulations are based require estimates of a large set of parameters. However, in actual projects, budget and time limitations may mean that only limited information is available. Many estimates, however inaccurate they may be, do not affect the effluent prediction; and some parameters vary slightly from plant to plant. Therefore, the evaluation of uncertainty is an important part of rationalising information acquisition in the field. One way in which uncertainty can be evaluated is by a sensitivity analysis with the not yet calibrated models, assigning “best guesses” for the probability distributions of the parameters and variables. Cost risks must also be considered. A risk–cost–benefit analysis may provide the economic justification for increasingly detailed and expensive investigations.

Because of the large uncertainties, a probabilistic approach may be more helpful. For example, Monte Carlo analysis can be used to assimilating the various input uncertainties of the different modelling steps involved in quantitative risk analysis and to produce a realistic appreciation of the total exposure uncertainty and its consequences.

Monte Carlo analysis allows relevant in-depth information from a historical data series to be retained in the risk–cost–benefit analysis. This point is illustrated below, in the example of analysis of nitrate and nitrite exposure for wastewater reclamation plants for aquifer recharge (Figure 4.2). The figure shows the regressive relationship between the wastewater treatment plant influent nitrogen concentration and the daily flow. The influent loading is important for predicting the nitrite concentrations generated by activated sludge treatment and subsequent treatment steps.

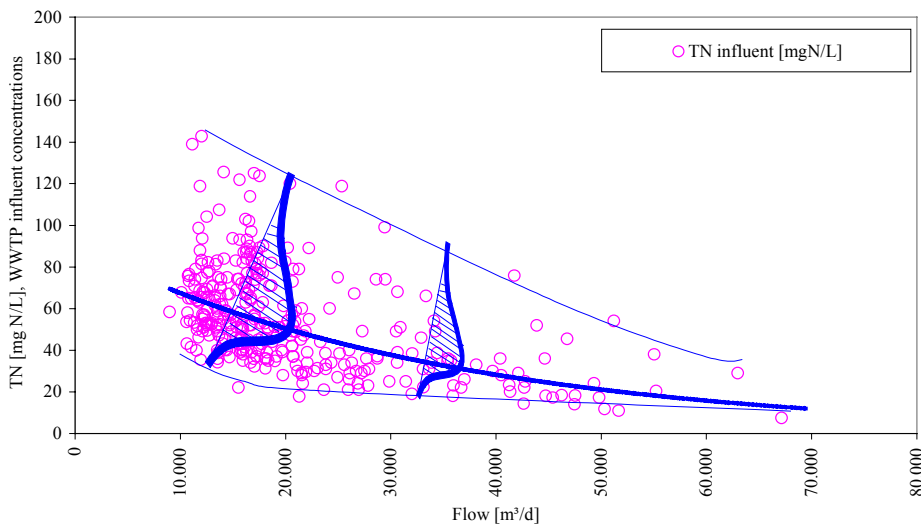


Figure 4.2 Influent total nitrogen concentrations versus daily flow at a wastewater treatment plant inlet (400 data points)

The Monte Carlo technique can retain information about low, median and high-boundary regression curves, as well as their probability distribution (as shown in Figure 4.2). Therefore it avoids compounding high-end boundary regression curves (dynamic modelling) and values (static modelling).

Phase 3 — field data acquisition

Field testing can help to validate or reject the arbitrary assumptions made in the previous stage, reduce uncertainty, and complement and further validate relevant historical data.

Obviously, the quality of the data input into the models is important. Issues in the acquisition of field data include sample representativeness, lack of standardized procedures and measurements at the limit of detection of commercially available instruments. These issues are reviewed by Parker (1994) and Loaiciga et al. (1992).

Phase 4 — risk management scenario analysis

Scenarios should be analysed for processes under optimal operational conditions, suboptimal conditions and “unexpected” events (“unexpected” here means outside the simulation assumptions). In some situations (e.g. breakdown of a membrane or failure of a pump), this can, at best, endanger the compliance of the actual results with the required quality standards, or, at worst, cause an epidemic. Clearly, an evaluation of the available safety margin is needed. This can be achieved in various ways; for example, by a detailed event tree analysis, which provides an explicit means of examining the process configuration’s vulnerability to suboptimal operation or unexpected conditions by evaluating the set of very restrictive assumptions initially made in the model-based analysis.

Once the probability of exposure to a hazardous agent is established, dose–response models (in the case of regulatory agencies) or quality standards (in the case of water utilities) can be applied to establish the severity of consequences for a specific population, caused by the release of a particular agent. As an example, Figure 4.3 illustrates the cumulative probability distribution of nitrite at a wastewater treatment plant outlet during a representative summer period. The 5, 50 and 95 percentile uncertainty boundary profiles are plotted.

Figure 4.3 shows that, with 50% certainty, the secondary effluent is below the NOAEL value indicated by US EPA (1985) for 90% of the time and, with 95% certainty, it is below the NOEAL value for more than 50% of the time (points 1 and 2, respectively). In that example, tertiary treatment (reverse osmosis) before infiltration was adopted, because the local regulations indicate that further treatment or action should be undertaken.

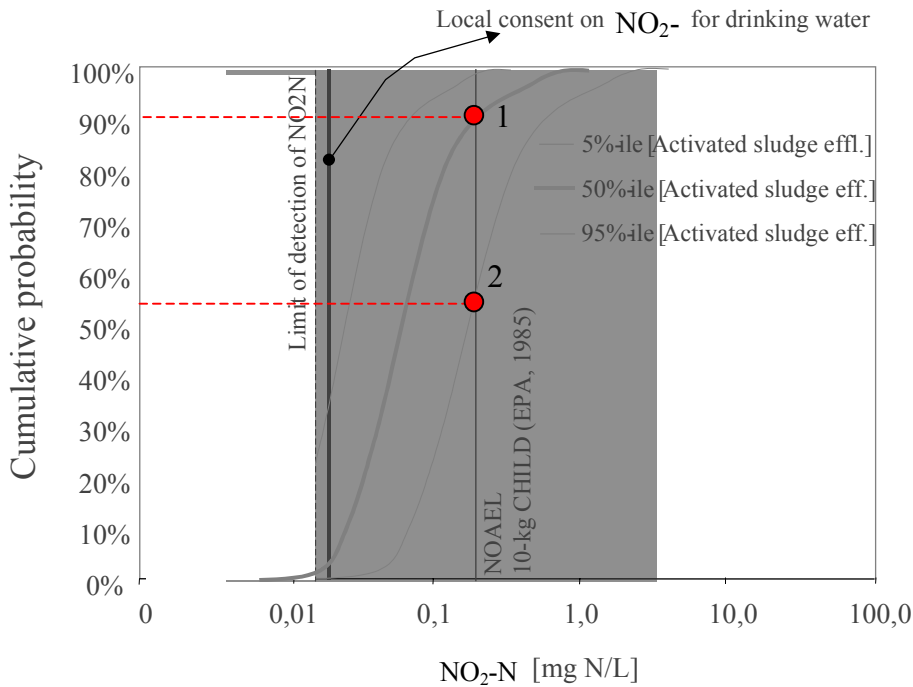


Figure 4.3 Simulation results for nitrites for artificial aquifer recharge at the reuse installation: secondary effluent

With a detailed scenario analysis, probability of morbidity derived from epidemiological risk analysis can be determined directly. The increase in probability of exposure to a health risk can immediately be added to the graph, providing a direct estimation of the factor of safety (i.e. of the relative dangers to the population exposed).

The time-dependence may be important; for example, to provide a more realistic assessment of the risks involving those substances that can cause harm after long-term exposure, or to assess the effect of temporary suboptimal conditions in the event tree analysis. Rainfall and various diurnal and seasonal factors (water consumption, temperature and concentration levels in the recycled wastewater) may vary significantly.

4.7.2 Probabilistic approach

Conceptual models can help to predict general economic, ecological and human health impacts of certain decisions. However, numbers can often be misleading if the model uncertainty and input uncertainty associated with them are not made clear. Probabilistic techniques provide a useful tool for clarifying these uncertainties. Techniques can be as simple as the “method of moments” or as complex as Monte Carlo analysis (Vose, 1996). Monte Carlo analysis is the technique that is most often used in uncertainty assessment (e.g. Medina et al., 1996; Delvecchio & Haith, 1998). The rigor of the Monte Carlo analysis may be necessary because the problem is complex and the decisions are being made in situations of uncertainty and high stakes. Monte Carlo analysis can be coupled to deterministic models in a number of ways; for example, by following these steps:

1. Assign information about the probability distribution of each input parameter and variable in the system.
2. For every calculation, the simulation uses a value for each input parameter randomly selected by the Monte Carlo engine from the probability density function for that variable. In making multiple calculations, the Monte Carlo engine produces a range of values for the input parameters and variables, reflecting the probability density function of each input parameter and variable. Enter the set of samples (‘shot’) into the deterministic model.
3. Solve the conceptual model for each shot, as for any deterministic analysis, static or dynamic.
4. Store the model results and repeat the process until the specified number of model iterations is completed. The output can now be expressed as probability density function or cumulative probability density function (Fig. 4.4).

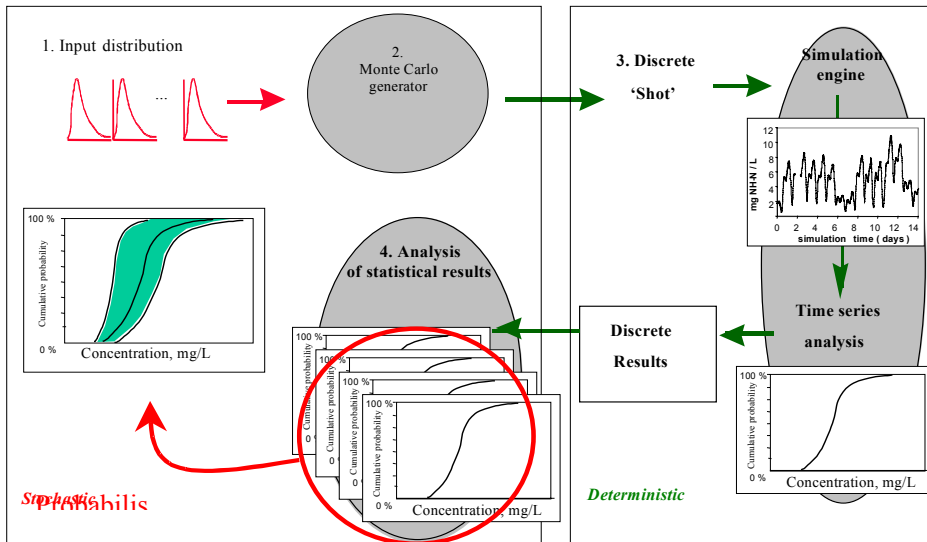


Figure 4.4 Layout of the probabilistic methodology

Uncertainty and variability

The probabilistic simulation takes into account both input and parameter uncertainty, in this way it deals with the difficulties in estimating model parameters and takes into account the inherent uncertainty in specific phenomena. (Variability represents heterogeneity or diversity, which cannot be reduced through further measurement or study. Uncertainty represents ignorance about a poorly characterised phenomenon, which can sometimes be reduced through further measurement or study). In the approach set out in Figure 4.4, the variability is assumed to have been completely captured through the introduction of dynamic mechanistic simulations, and the uncertainty through the Monte Carlo simulation. Therefore, there is no need for a second order Monte Carlo analysis that would simulate variability and uncertainty in two loops, as illustrated in Grum & Alderink (1999).

Analysis of the model’s result

This iterative process generates a probability density function or cumulative density function of the output (Rousseau et al., 2001). Based on the distribution of the output, a risk exposure level representing the high end (e.g. 95th percentile), central tendency (median or mean) or any other desired level of probability can be identified. It is therefore possible to represent uncertainty in the output of a model by generating sample values for the model inputs and running the model repetitively. Instead of obtaining a single value result, as is the case with a deterministic simulation, a set of results is obtained (Cullen and Frey, 1999). This set represents the cumulative effect of the given uncertainties of the input items.

4.7.3 Strengths of the probabilistic approach

The basic strengths and advantages of the probabilistic approach are described below.

Decisions are made with a healthier understanding of the factors influencing that decision. Explicitly incorporating uncertainty and variability in the model-based risk analysis can simplify the risk manager’s decision or increase their “comfort factor”: by pointing out dangers in a more realistic fashion; that is, by highlighting where the uncertainty or indeterminacy actually is, or by providing a kind of guarantee (to the manager and to outside overseers) that an impartial systematic search has been conducted. The procedure is more transparent to stakeholders, because the measure of the risk of a particular course of action is grounded in a rational analysis of the available

imperfect information and because the expert judgement is made explicitly. A transparent process with all the assumptions and parameters clearly stated can help to settle possible controversies between water utilities, government and environmentalists.

The methodology can increase the likelihood that standards will be met and can save money. It shows the degree of conservatism that would result from the compounding of conservative assumptions employed in conventional projects, and also proposes a robust and transparent way of avoiding that situation. In the conventional approach, the poor link between the cause–effect relationship of risk and uncertainty makes it necessary to calculate a conservative or high-end-point estimate of input variables and parameters. However, using this approach, the uncertainty of each input variable and parameter can be explicitly introduced into the model-based analysis; thus, it contributes to the calculation of the overall uncertainty of the system.

The procedure provides systematic evaluation of the uncertainties of variables and parameters, which can be useful in rationalizing the acquisition of the new measurements. Uncertainty can be reduced (at a cost!), but not eliminated. Therefore, quantitative information about the causal link between input uncertainty and overall uncertainty in the system can help to ensure that allocation of limited financial resources results in a decrease in the overall uncertainty in the system. The risk assessor may wish to reduce uncertainty in a particular situation by gathering additional information. This involves reallocating investment resources, as it increases costs for monitoring and for personnel in the early phase of the project. The probability analysis can be used to justify the cost involved in acquiring additional information.

4.7.4 Weaknesses of the probabilistic approach

The concepts introduced here are potentially of great significance in the process of risk management. However, risk analysis is basically a mathematical tool and can only be of practical use if predictive models are available, and quantitative estimates of the probability distribution of input parameters and variables can be made.

Results of the analysis depend on the risk assessor being willing and able to invest time and resources in searching for valid and relevant information. The data available about present and past events play a central role, reducing the input from arbitrary judgement. However, the effective use of the Monte Carlo simulation technique depends heavily upon such information being available.

This approach does not eliminate risks, it helps in identifying and dealing with imperfect information. Because of the high level of complexity and the assumptions, this approach should never be applied in a mechanistic fashion, and any conclusions it suggests must be carefully considered in the light of sound technical judgement and experience. In fact, although this methodology can serve as an objective basis for risk characterization, it does not mean that expert judgement is abandoned altogether! On the contrary, expert input is required and should carry weight when setting an acceptable level of risk.

Where probability drawn from past or present experience is not available, “subjective” probabilities are considered, based on the risk assessor’s own expectations, preferences, experience and judgement. These expectations can provide some assistance, but it is clear that subjective probabilities regarding uncertainty can be dangerous in decision-making. Risk assessors or managers may have an unjustified overconfidence in the modelling approach, and expert judgement must be used with caution (especially considering that experts are often overconfident).

The limitations of modelling must be kept in mind. Models are generally built around some specific narrow boundary conditions. Moreover, mathematical rigor must not be confused with reality, and the limitations of scientific knowledge and real-life technical challenges must not be forgotten. The validity limits of predictive models have to be taken into account, because a documented and experimental validation of the critical hypothesis may follow.

4.8 Quality management approach

The quality of the water produced by a water treatment plant depends on the reliability of the installation, which is the probability that a system can meet established performance criteria consistently over extended periods of time, or the likelihood of achieving an effluent that matches, or is superior, to predetermined standards.

Various factors can affect the quality of the produced water. Relevant in this context are variations in the influent streams and in the performance of the system. Variations in the influent stream are taken into account in the design of a treatment plant. If the influent concentrations are within the limits of the design of the system, the quality of the effluent should fall within the design parameters, provided that the specified plant is operated consistently and maintained adequately. The quality of the effluent can be estimated by modelling, based on frequency distributions of influent parameters. In the most basic modelling approach, the treatment system is seen as a box, and the quality of the produced water is studied as a function of the influent. This type of modelling is based on analysis of influent and effluent samples.

The performance of the system is an equally important influence on the quality of the produced water. The performance is expressed as operational reliability or mechanical reliability (Olivieri et al., 1998). In this scenario, a water treatment system is not a box, but consists of a succession of independent treatment steps, each of which forms a barrier for components to be removed during treatment. Barriers neither perform perfectly nor fail completely; instead they have a performance distribution, which can be described using statistical analysis of the performance of the individual process elements (National Research Council, 1998).

From the perspective of plant reliability, the performance of the most vulnerable or weakest component is the most crucial. The identification of these critical components is therefore essential and forms the first step in a mechanical reliability analysis. Hazard analysis and critical control point (HACCP) is an operational reliability analysis technique based on critical points.

Reliability analysis can go further than the critical points, by looking at the complete statistical data for each independent barrier. Examples of analysis techniques in which performance statistics are used are critical component analysis (CCA) and failure mode, effects and criticality analysis (FMECA), which aims to quantify the risk of noncompliance of the produced water with the standards (Lainé et al., 2000). Performance statistics include (Olivieri et al., 1998):

- reliability statistics —overall mean time between failures;
- maintainability and maintenance statistics — mean time to repair and corrective maintenance person hours per unit per year;
- availability statistics — fraction of time the unit was operating and fraction of time it can be expected to operate, excluding preventative maintenance.

The performance distribution of a single treatment step can be expressed as a function of the influent concentration, under the form of “transfer functions”. A transfer function gives the output concentration as a function of the input concentration for a given treatment. The “nominal transfer function” applies to optimal conditions. The “degraded transfer function” takes the failures into account and is drafted using performance statistics (Lainé et al., 2000).

4.8.1 Hazard analysis critical control point

HACCP has been developed for the food processing industry, with the aim of optimising the end-product by minimizing the risk of contamination. It consists of:

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- identifying the critical control points where hazards such as microbial contamination can occur;
- establishing critical limits;
- establishing controls to prevent or reduce the hazards;
- setting up verification procedures and maintaining records documenting that the controls are working as intended (Food Safety and Inspection Service, 1996).

In the third edition of the WHO *Guidelines for Drinking-water Quality* (WHO, 2003), WHO advocates the use of water safety plans that employ HACCP-like principles to control health risks from the catchment to the tap.

An example of a HACCP analysis is given by Dewettinck et al. (2001). The article lists the hazards, control measures, monitoring and corrective actions for a planned water reuse installation. The main disadvantage of HACCP is that the failures and risks are not quantified, and there are no examples of the application of HACCP that quantify operational reliability. Laîné et al. (2000), explains how FMECA can be used in a water treatment plant, but without going to the level of quantification. The HACCP concept could be completed by weighing the hazards against each other.

Nurizzo et al. (2000) suggest that failures in a water treatment installation have only a minor effect on the water quality, because of the multiple barrier design. The authors also suggest that factors with the greatest influence are variations in influent concentrations, pH and temperature.

Together with quality management approach, it is interesting to study the legislation or guidelines on minimal treatment as a tool for minimizing health risks. In relation to reuse, legislation contains the different parameters and their maximum concentrations in the water, plus detailed descriptions of the minimal treatment to be applied (National Research Council, 1998). In the future, reuse is likely to be increasingly regulated by descriptions of minimal treatment and operational controls that must be carried out.

Health risks should also be managed according to their context of the overall water-related disease burden (see “Stockholm framework” in Appendix A). Tolerable risk can be looked at in the context of total risk from all exposures, and risk management decisions can be used to address the greatest risks first. For example, halving the number of cases of salmonellosis attributed to drinking-water would have very little impact on the disease burden if 99% of the cases were related to food.

4.9 Effects approach

The effects approach involves studying the effect of the water on test organisms (using biotests) or on populations (using epidemiology). This section looks in detail at this approach.

4.9.1 Biotests

The term “biotests” refers to biological testing using test organisms (in vivo) or modified cell tissues (in vitro). Biotests can be used to assess the health risk related to the use of a certain type of water or to monitor the quality of the water produced, for example at a reuse plant (referred to as biomonitoring). The major advantage of biotests is that the water is studied as a mixture; thus, such tests can detect effects that might occur due to synergism between different pollutants. A disadvantage is the fact that each test measures only one effect (e.g. toxicity is measured with one test and endocrine activity with another).

Certain compounds might have different effects on organisms. For this reason, it is necessary to use a battery of biological tests to assess a particular water (Penders & Hoogenboezum, 2001). Much research is being carried out on the development of new biomonitoring tests and the adjustment and standardization of existing tests. The following section describes biotests that are currently of importance in relation to reuse of water.

Bioassays

Bioassays can be used to assess the toxicity of individual compounds in a water sample at different trophic levels (e.g. using bacteria, algae and water fleas) (Penders & Hoogenboezum, 2001). Bacteria are used to study the possible biodegradation of toxic components, algae to investigate effects on photosynthesis and plant life, and water fleas to look at potential effects on water consumers. The choice of bacteria, algae or water fleas for bioassay depends on the aim of the tests. Thus, a set of bioassays used to estimate the effects of a particular water on the environment will differ from that used to monitor drinking-water.

Bioassays can test for acute or chronic toxicity. Tests for acute toxicity show the harmful effect of compounds within a short period (e.g. 72 hours). Tests for chronic effects, necessary to determine long-term effects, require complex procedures and involve high costs. Concentration techniques can allow bioassays for acute effects to be used to estimate chronic effects. Examples of such techniques are solid-phase extraction followed by elution, liquid/liquid extraction, freeze drying and membrane techniques (Penders & Hoogenboezum, 2001).

Genotoxicity

Genotoxicity tests are designed to measure the mutagenicity of samples. One of the first genotoxicity tests developed was the Ames test. This test is laborious and time consuming, having a response time of 3 days. Faster tests have now been developed, such as the UMU-test and the VITOTOX[®] test, both of which have a 4-hour response time, and the Comet test, which has a 2-hour response time (Penders & Hoogenboezum, 2001).

Effect-specific tests

Effect-specific tests can measure different effects of compounds found in water, such as enzyme inhibition (Penders & Hoogenboezum, 2001), bioaccumulation (De Maagd, 2001) and estrogenic activity (Berckmans, 2001). Such tests have been developed to understand and measure the effects of endocrine disrupting chemicals on humans and wildlife populations.

Various methods can be used to estimate estrogenic activity. In vitro tests include: the ER-CALUX[®] assay (Penders & Hoogenboezum, 2001) and the MVLN assay (Berckmans, Vangenechten & Witters, 2001), which are based on human breast cancer cells; and the YES-assay, which is based on yeast cells (Witters, Vangenechten & Berckmans, 2001).

In vivo endocrine activity tests involve fish reproduction studies. Fish are exposed to the water under investigation for 3 weeks; the induction of vitellogenin in the blood of male fish and the ovary and testis size are then examined, as well as the female spawning success and male fertility.

Monitoring of water for endocrine activity using MVLN, YES and fish reproduction bioassays showed that levels of endocrine activity are lowest in surface water used for drinking-water production, slightly higher in wastewater treatment plant effluent and highest in rivers with average pollution (Witters, Vangenechten & Berckmans, 2001; Berckmans, Vangenechten & Witters, 2001).

Further research is needed to develop test methods specifically for water testing in relation to drinking-water reuse and to identify those compounds that have high estrogenic activity.

4.9.2 Epidemiology

Epidemiological studies can complement biotests. In the few cases where wastewater is intentionally reused as drinking-water, epidemiological studies are used to measure the effect of the water on the population. In existing, well-managed reuse schemes, such studies have found no increase in the level of infections, cancer cases or other diseases, suggesting that the increase in risk related to the use of recycled water is negligible when the process is managed correctly. In cases where water is reused for irrigation, epidemiological studies are less frequent, mainly because of the high costs involved, although WHO has been active in this area. Epidemiological studies have some major disadvantages (National Research Council, 1998):

- they are complex and costly
- it is difficult to estimate the real concentrations to which the population is exposed
- it is difficult to estimate simultaneous exposure to different pollutants or from different pollution sources.

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