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# State of the Art Report
# Health Risks in Aquifer Recharge
# Using Reclaimed Water

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## Foreword

Freshwater is an important resource and will become more so in the future as population increases. Within the next fifty years, it is estimated that 40% of the world's population will live in countries facing water stress or water scarcity. The figure may actually be higher than 40%, because these data are calculated on a national basis, and therefore often do not take into account uneven distribution of water within national boundaries.

In many areas of the world, aquifers that supply drinking-water are being used faster than they recharge. Not only does this represent a water supply problem, it may also have serious health implications. Moreover, in coastal areas, aquifers containing potable water can become contaminated with saline water if water is withdrawn faster than it can naturally be replaced. The increasing salinity makes the water unfit for drinking and often also renders it unfit for irrigation.

To remedy these problems, some authorities have chosen to recharge aquifers artificially with treated wastewater, using either infiltration or injection. Aquifers may also be passively recharged (intentionally or unintentionally) by septic tanks, wastewater applied to irrigation and other means.

Aquifer recharge with treated wastewater is likely to increase in future because it can:

- restore depleted groundwater levels
- provide a barrier to saline intrusion in coastal zones
- facilitate water storage during times of high water availability.

If aquifer recharge is haphazard or poorly planned, chemical or microbial contaminants in the water could harm the health of consumers, particularly when reclaimed water is being used. Wastewater may contain numerous contaminants (many of them poorly characterized) that could have health implications if introduced to drinking-water sources.

Ensuring that the use of treated wastewater for aquifer recharge does not result in adverse health effects, a systematic science-based approach is needed, designed around critical control points, as used in the hazard analysis critical control point (HACCP) approach. Such an approach to potable aquifer recharge requires a thorough evaluation of the best practices that will protect public health, and consideration of environmental and sociocultural concerns.

The Mediterranean region has high levels of water stress but only limited water recycling. During the Barcelona Convention[1], the contracting parties to the convention agreed to reconsider the state of wastewater reuse by assessing current practice. The Barcelona Convention is part of the legal framework supporting the Mediterranean Action Plan, which involves 21 countries bordering the Mediterranean Sea as well as the European Union. The plan was established to protect the Mediterranean region by addressing environmental degradation and linking development with sustainable resource management. The World Health Organization (WHO) supports the Mediterranean Action Plan through a permanent presence in the joint Secretariat administering the plan.

At about the same time as the Barcelona Convention and the establishment of the Mediterranean Action Plan, the WHO Regional Office for Europe received a growing number of expressions of concern from member states that were experiencing increasing levels of water stress. The regional office decided, in consultation with the WHO Water, Sanitation and Health Unit, to hold an expert

[1] The Convention for the Protection of the Marine Environment and the Coast Region of the Mediterranean (known as the The Barcelona Convention) was adopted on 16 February 1976 in Barcelona, Spain, and entered into force on 12 February 1978. There were 21 parties to the convention, including the European Community.

consultation on health risks associated with recharge of aquifers by means of treated wastewater. The meeting was eventually hosted at the Fodor Joszef Institute for Environmental Health in Budapest, Hungary, from 9 to 10 November 2001. The meeting was attended by experts from nine different countries and six different organizations. The full report of the meeting is available online.[2]

Following the meeting, WHO decided to invite selected experts to contribute to a state of the art report to capture their experience. The present volume contains this report. Its aim is to compile examples of the current state of research into aquifer recharge and, more specifically, to highlight how the important issue of assessing and managing health risks has been addressed by different experts, either in academic research or in practical application. The report does not present formal WHO guidelines, and should not be interpreted as such.

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[2] http://www.euro.who.int/document/wsn/WSNgroundwaterrpt.pdf

## Abbreviations and acronyms

| | |
|---|---|
| ADI | **ACCEPTABLE DAILY INTAKE** |
| AIDS | aquired immunodeficiency syndrome |
| APEC | alkyl phenol polyethoxy carboxilates |
| APT | advanced primary treatment |
| ASR | aquifer storage recovery |
| AWT | advanced wastewater treatment |
| BOD | biochemical oxygen demand |
| BW | body weight |
| C | daily drinking-water consumption |
| CCA | critical component agency |
| CCL | contaminant candidate list |
| CFC | chlorofluorocarbon |
| CFU | colony forming units |
| COD | carbon oxygen demand |
| DAEC | diffusively adherent *Escherichia coli* |
| DALY | disability adjusted life years |
| DDT | dichlorodiphenyltrichlorethane |
| DHS | Department of Health Services (California) |
| DNA | dexyribonucleic acid |
| DOC | dissolved organic carbon |
| DS | dissolved solids |
| EDC | endocrine disrupting compound |
| EDTA | ethylene diamine tetracetic acid |
| EHEC | enterohemorrhagic *Escherichia coli* |
| EIA | environmental impact assessment |
| EIEC | enteroinvasive *Escherichia coli* |
| EPEC | enteropathogenic *Escherichia coli* |
| ETEC | enterotoxigenic *Escherichia coli* |
| EU | European Union |
| FMECA | failure mode, effects and criticality analysis |
| GAC | granular activated carbon |
| GIS | geographical information system |

| | |
|---|---|
| HAA | haloacetic acid |
| HACCP | hazard analysis critical control point |
| HIV | human immunodeficiency virus |
| HO | helminth ova |
| HPC | heterotrophic plate count |
| IARC | International Agency for Research on Cancer |
| ID | infective dose |
| JECFA | Joint Expert Committee on Food Additives (WHO) |
| JMPR | Food and Agriculture Organization of theUnited Nations (FAO)/WHO Joint Meeting on Pesticide Issues |
| LAEC | locally adherent *Escherichia coli* |
| LOAEL | lowest-observed-adverse-effect level |
| MAC | *Mycobacterium avium* complex |
| MCL | maximum contamination level |
| MF | microfiltration |
| MPN | most probable number |
| na | not applicable/not available |
| NAA | nitric acetic acid |
| NAS | National Academy of Science (USA) |
| NDMA | N-nitrodimethylamine |
| ns | not specified |
| NF | nanofiltration |
| NOAEL | no-observed-adverse-effect level |
| NRC | National Research Council (of the USA) |
| NTU | nephelometric turbidity unit |
| OSPAR | Protection of the Marine Environment of the North-East Atlantic |
| PAH | polycyclic aromatic hydrocarbon |
| PCB | polychlorinated biphenyl |
| PFU | plaque forming unit |
| POP | persistent organic pollutant |
| RFD | reference dose |
| RNA | ribonucleic acid |
| RO | reverse osmosis |

| | |
|---|---|
| RWC | recycled water contribution (%) |
| SAT | soil aquifer treatment |
| SS | suspended solids |
| TC | total coliforms |
| TDI | tolerable daily intake |
| TDS | total dissolved solids |
| TEQ | toxicity equivalent |
| THM | trichlorohalomethane |
| TOC | total organic carbon |
| TOX | total halogenerated organic compounds |
| TSS | total suspended solids |
| UF | ultrafiltration |
| UK | United Kingdom |
| UOSA | Upper Occoqan sewage plant |
| US EPA | United States Environmental Protection Agency |
| USA | United States of America |
| UV | ultraviolet |
| VOC | volatile organic compound |
| WF-21 | Water Factory 21 |
| WHO | World Health Organization |
| WWTP | wastewater treatment plant |
| YLD | years lived with a disability |
| YLL | years of life lost |

# 1 Introduction

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## 1.1 Background

### 1.1.1 The need for aquifer recharge

Increasing demand for water, particularly in arid and semi-arid regions of the world, has shown that the extended groundwater reservoirs formed by aquifers are invaluable for water supply and storage. Natural replenishment of this vast supply of groundwater is very slow. Therefore, exploiting groundwater at a rate greater than it can be replenished causes groundwater tables to decline and, if not corrected, eventually leads to mining of groundwater. Artificial recharge as a means to boost the natural supply of groundwater aquifers is becoming increasingly important in groundwater management.

Groundwater can have a wide range of beneficial uses. For example, it can be used for irrigation of parks or agricultural land, industrial application, or to provide a potable water supply (i.e. one that is suitable for drinking).

### 1.1.2 Types of recharge

Artificial recharge involves augmenting the natural movement of surface water into underground formations. The recharge can be either direct or indirect. In direct recharge, water is introduced into an aquifer via injection wells. The injected water is treated to ensure that it does not clog the area around the injection well. Indirect recharge involves spreading surface water on land so that the water infiltrates through the vadose zone (the unsaturated layer above the water table) down to the aquifer. Methods for spreading water include over-irrigation, creating basins using construction methods, or making artificial changes to natural conditions (e.g. modifying a stream channel) (Asano, 1998). An advantage of indirect recharge is that the vadose zone acts as a filter, treating, and therefore improving the quality of, the water percolating through the soil. This process is referred to as soil aquifer treatment.

Recharge can be intentional or unintentional. Injection of treated wastewater would be an example of direct intentional recharge; whereas, infiltration of water used for agricultural irrigation would generally be an example of unintentional indirect recharge. Unplanned indirect reuse for potable water supplies is increasing, due to municipal water intakes located downstream from wastewater discharges or increasingly polluted rivers and reservoirs.

### 1.1.2 Wastewater as a resource for recharge

Various sources of water are available for groundwater recharge but, in recent years, the use of nonconventional water resources such as recycled municipal wastewater, has received increasing attention. The primary reasons for considering use of recycled water in groundwater recharge are that recycled wastewater is available for reuse at a relatively low cost and that it provides a dependable source of water even in drought years.

As artificial recharge has increased in popularity, managers have begun to search for additional sources of recharged water. A critical question is whether waters of impaired quality should be used for this purpose, and whether the water recovered from such systems is suitable for potable uses as well as nonpotable ones. Obviously, water of impaired quality could only be used with appropriate pre and post-treatment, and treatment gained from soil and aquifer processes.

### 1.1.3 Challenges to using wastewater for recharge

Using recycled municipal wastewater for artificial recharge of groundwater presents a wide spectrum of technical and health challenges. A major consideration is the possible presence of chemical and microbiological agents in the source water that could be hazardous to human health and to the environment. Concerns about hazardous agents in the water apply particularly to potable reuse. Although nonpotable uses such as irrigation can result in human exposure to hazardous agents, there is less potential for exposure and the risks are therefore significantly lower (NRC, 1994).

Two possible constraints that may limit the use of recycled municipal wastewater for groundwater recharge are concerns over the impacts of emerging contaminants on long-term human health, and public perception on potable reuse. Minimizing the health risks caused by water that people drink is of great importance worldwide.

A further concern is how risk resulting from aquifer recharge will be defined accurately in terms of environmental and health issues (e.g. preventing the degradation or impairment of water quality in groundwater basins that are, or could be, used for domestic water supplies). Detailed information on the processes governing the fate of pathogens and chemicals is required in order to develop appropriate models for determining risk assessment.

Four water quality factors are significant in groundwater recharge with recycled water:

- human pathogens
- mineral content
- heavy metals
- trace organic compounds.

Human pathogens and trace organic compounds are of particular concern when groundwater recharge involves aquifers supplying domestic water (Tsuchihashi, Asano & Sakaji, 2002).

The need for definitive information on the extent of contaminant removal by the soil and underlying geological formations, and on the fate of pollutants during groundwater recharge, has been recognized and is being studied extensively. Much of the research on groundwater recharge and potable water reuse is becoming as relevant as research on unplanned indirect reuse for potable supplies. Tapping polluted sources has potential effects that go beyond the increased cost of additional treatment. Incidental or unplanned indirect potable reuse of polluted water may expose people to health risks not associated with protected sources. The health concerns associated with drinking-water drawing upon polluted sources apply even more forcefully to wastewater recycling and reuse for potable purposes.

## 1.2 Aim and structure of this report

### 1.2.1 Aim

The aim of this report is to contribute to improving practices in intentional groundwater recharge, and to introduce a precautionary approach in other practices (e.g. agricultural irrigation with raw wastewater and land disposal of wastewater), which result in unintentional recharge, to reduce the risks to acceptable social, economic and environmental levels. The report is a first step on the path to developing simple, flexible and practical health-related guidelines that will help to improve practices in both direct and indirect groundwater recharge.

This report is written for civil and sanitary engineers, agricultural engineers, hydrologists, environmental scientists and research scientists. It will also be a useful reference for public works officials, consulting engineers, agriculturists, industrialists, academics and students.

Water is precious. Managing water is a global challenge that affects the environmental, social, economic and political cornerstones of our existence on Earth. Artificial recharge provides an opportunity for sustainable management of water resources, conservation and improvement of quality, to face future water demands. Recharge concepts are simple but practitioners know that we have much to learn about issues relevant to sustainability of managed recharge of groundwater, especially with regard to public health and environmental aspects.

### 1.2.2 Structure

The remaining chapters of this report look at various aspects of the health and environmental risks of aquifer recharge using reclaimed water.

Chapter 2 is an overview of aquifer recharge. It presents an approach to recharge that takes into account the intended use of the groundwater; it also considers existing guidelines. Issues such as differentiation between potable and nonpotable aquifers, and direct and indirect reuse are critical when implementing aquifer recharge projects. The water withdrawn from the recharged aquifer should not require supplementary to meet existing quality standards for its intended use. Moreover, the treatment that applied wastewater effluent undergoes in the vadose zone should not be taken into consideration. Although the effect of such treatment is well documented, it should be considered only as an additional barrier in case of failure of the basic treatment. In contrast, percolation of recycled water through the unsaturated zone can be considered as an additional treatment.

Chapter 3 reviews recharge of groundwater in both developing and developed countries. In the case of developed countries, social acceptance is the main limitation on indirect intentional reuse. Consequently, intense communication campaigns are needed, aimed at persuading society to accept the concept of intentional reuse, especially with regard to public health and environmental protection issues. In addition, economic aspects (levels of initial investments and operation and maintenance costs) should be considered in such campaigns. This chapter also suggests that developed countries undertake epidemiological studies to substantiate their arguments for artificial recharge of groundwater.

In developing countries, especially those in arid or semi-arid regions, artificial recharge of groundwater is particularly important. A shortage of water in these countries has forced a situation in which water of a less than desirable quality is being used. Chapter 3 takes into account both this situation, and the fact that relatively cheap improvements in indirect intentional reuse practices could significantly reduce health risks, especially those concerning gastrointestinal diseases. The guidelines provided in Chapter 3 will be useful in the design and implementation of such improvements, in systematizing epidemiological studies and in undertaking toxicological studies to guard against the effects of toxic pollutants. The guidelines cover cases where disposal of wastewater in soils results in incidental recharge.

Chapter 3 also gives extensive consideration to pathogens present in waters and in wastewater, waterborne diseases and microbiological indicators, and describes treatment processes in both the developing and developed world. This chapter reviews various types of aquifer storage recovery systems and considers regulatory issues, emphasizing aspects relevant to setting regulations for recycled water intended for human consumption via aquifer recharge. Important regulatory concerns include treatment practices and technologies, evaluation and use of the “best” indicator organism, selection monitoring parameters, definition of sampling (taking into account epidemiological studies and toxicological tests) and use of a risk evaluation model. Finally, Chapter 3 describes examples of legislation and proposes guidelines to be applied to indirect potable reuse of aquifers recharged by spreading and by injection.

Chapter 4 looks at legislation concerning drinking-water production and considers the main contaminants of concern for aquifer recharge. It considers two approaches to health risk assessment of artificial recharge of groundwater. The first method, known as the parameter approach, involves estimating the health risk using as a reference either drinking-water standards or (in a quantitative risk assessment) toxicological data, and data on infectious doses and acceptable risk (chemical and biological). The alternative approach is to study the effect of recharge on test organisms or on the population, and estimate or calculate the risk posed by chemicals and pathogens. The chapter reviews several biotests (e.g. bioassays, genotoxicity tests and effect-specific tests) that are used to understand and assess the effect of specific contaminants, such as endocrine disrupting compounds.

Chapter 4 also discusses analytical and probabilistic models for predicting economic, ecological and human health risk assessment for artificial recharge of groundwater.

Chapter 5 describes an integrated approach to assessing the impact of aquifer recharge, where territorial implementations for safe use are interconnected with social, economic and environmental issues. Where groundwater is the main water resource covering potable, washing and irrigation requirements, it is an important contributing factor to exposure risk for consumers or users. Initially, a preliminary environmental health assessment is required for defining policy options. Chapter 5 describes a procedure that includes the following stages:

- outline policy goals
- identify options
- assess the uncertainties of each option
- evaluate costs and benefits of each option
- identify the favored policy.

In a latter stage, the environmental health impact of aquifer recharge with recycled water must be assessed. This stage includes identifying chemicals and pathogens, and assessing risks and health impacts. Perspective data collection with the use of geographical information system (GIS) and environmental mapping for baseline definition and health surveillance is highly recommended.

Chapter 6 reviews health and regulatory aspects of groundwater recharge with recycled municipal wastewater, and provides practical recommendations to guide decision makers. At present, uncertainties about health risk considerations have limited the use of recycled municipal wastewater for groundwater recharge. Groundwater containing a large portion of recycled wastewater may affect the drinking-water supply. This chapter presents two case histories for soil aquifer treatment and nonpotable water recycling and reuse, with less stringent water quality goals than would be required if potable reuse was being considered. It also discusses the proposed State of California (USA) criteria for groundwater recharge as an illustration of a much more conservative approach.

Chapter 7 presents the basic principles of risk communication and sets out the main roles of communication in implementing policy options.

The report also contains four appendices. Appendix A describes the “Stockholm framework”, a harmonized framework for the development of guidelines and standards, in terms of water-related microbiological hazards (Bartram, Fewtrell & Stenström, 2001). Appendix B describes the Balearic Islands (Spain) as an example of regulation of wastewater reuse. Appendix C describes the process used to develop the WHO guidelines covering chemical aspects of drinking-water quality.

Appendix D summarizes the proposed State of California criteria for groundwater recharge and reuse projects.

## 1.3 Conclusions

The main conclusions drawn from this report are the following:

- Development of guidelines to minimize the microbial health risks associated with groundwater recharge with wastewater should take into account other water-related exposures (i.e. through drinking-water, recreational/bathing water contact, and through the food-chain). The management of water-related disease needs a comprehensive approach such as that outlined in the "Stockholm framework" (see Appendix A).
- Various models (analytical and probabilistic) are used for predicting human health risk. Microbiological risk assessment tools will play a critical role in developing future criteria based on the epidemiological studies. However, alternative strategies for minimizing risks in four dimensions (technical, economic, environment and health) using multicriterion decision analyses should be practised.
- Appropriate preapplication treatment technologies and site characteristics (soil permeability, groundwater depth etc.) should be tested to eliminate the potential threats posed by chemicals and pathogens. Establishing these factors ensures that a reliable minimum degree of treatment can be set, that must be adhered to before the recycled water is used. However, the regulatory issue is whether permitting the discharge of chemicals not present in any measurable quantity in a groundwater basin would constitute degradation or impairment of groundwater basin.
- Biotests are biological tests in vitro or in vivo. They can be used to assess the health risk associated with the use of a certain type of water or to monitor the quality of the water produced. The major advantage of biotests is that the water is considered as a mixture. Biotests using endogenous estrogen equivalents should be evaluated, to help to develop a basis for tracing organics in recycled water.
- Future research should be directed toward defining the limits of physical, chemical and biological agents that should be used to establish safe and sustainable groundwater recharge practices. Some topics for future research are how to more precisely establish:
  - microbiological risks;
  - presence, concentration and health significance of pathogens and toxic substances in recycled water by region;
  - fate of micropollutants, including pathogens in the soil and underlying geological formations;
  - residence times (using models),
  - extraction distances and chlorination alternatives;
  - soil and aquifer attenuation.

## 1.4 References

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# 2 Groundwater recharge with recycled municipal wastewater: criteria for health related guidelines

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## 2.1 Introduction

The quality of the water of a recharged aquifer is a function of:

- the quality of the recharge water;
- the recharge method used;
- the physical characteristics of the vadose zone and the aquifer layers;
- the water residence time;
- the amount of blending with other sources;
- the history of the recharge.

It is important to determine at which point water should be submitted to regulations and guidelines: at point of use, at the point of withdrawal from the aquifer or before recharge?

The aim of this chapter is to highlight some principles related to aquifer recharge with recycled water, and to propose a simple approach to health related guidelines that takes into account existing water regulations and guidelines, but avoids overlapping with them.

## 2.2 Features of aquifers

Aquifers have some specific features that may influence guidelines on aquifer recharge with recycled municipal wastewater.

When pollutants are introduced into an aquifer with the recharge water, they will either move with the water, as nitrates do, or be retained on the solid matrix, as generally happens to cations and organic matter. If pollutants that are retained do not break down, they will accumulate within the aquifer. Pollutant removal is regarded as a positive impact. However, despite promising findings (for example, the work of Fox et al., 2001, which provided evidence of organic compound removal in SAT), uncertainties remain about the fate of most contaminants. Questions raised about retained pollutants include the following:

- Is there a risk that the pollutants may appear in the abstracted water due to changes in the physical and chemical conditions prevailing in the aquifer or due to limited adsorption capacities?
- How long can microorganisms survive?
- To what extent are toxic pollutants degraded?

The most attractive aquifers for recharge projects are large aquifers that allow long-term storage and a long water retention time. Long retention time is an advantage, because it favours contaminant removal, but also a disadvantage, because contamination of the water can have a long-term impact, particularly for pollutants that are not efficiently removed. Reclaiming an aquifer that has been polluted is a difficult, long and expensive process; therefore, a prerequisite of artificial recharge is that it should not risk jeopardizing the groundwater resource.

Aquifers are exploited through pumping wells (private and public), which serve a range of different purposes, such as potable water supply, irrigation and industrial uses. The water quality required depends on the use, giving two main options for recharge. The first option is to plan and operate the recharge so that the quality of the water in the aquifer either meets the most stringent requirements or is not degraded. The second option, which is more sophisticated, is to use sector-based management — pumping water of different quality from different areas for different purposes. However, because an aquifer is a continuum, sector-based management requires in-depth knowledge of the aquifer and close monitoring of water quality . This type of management also requires stakeholders' agreement and a control of the withdrawals; for example, to ensure that farmers do not pump water that is fit only for irrigation for a potable water supply. The extension of the plume of injected water should be monitored, which can be achieved through the measure of the content of wastewater tracers such as chloride (in freshwater aquifers), sulfate, boron and gadolinium anomaly. Furthermore, as retention times in aquifers are long, an exploitation policy, once adopted, cannot easily be changed in the short term.

Aquifers are complicated heterogeneous, multilayered systems, often with poorly defined boundaries. Reliable predictions of groundwater flow are possible only if the aquifer system is well known, which means that sufficient data need to be available to work out well-calibrated hydrodynamic numerical models. Flow patterns are relatively easy to predict and control in granular media aquifers; however, due to their discontinuous and anisotropic porosity, the situation is quite different in fractured rocks and karst formations, where, despite recent improvements, modelling solute transfer is, and will long remain, difficult to achieve.

### 2.2.1 Considerations for regulating water quality

In most countries, the quality of water intended for human consumption is regulated through standards set at the regional, national or international level. Examples of such standards are the European Drinking Water Directive 98/83/EEC (Commission of the European Communities, 1998) , the United States Environmental Protection Agency Drinking Water Standards (US EPA, 1993) and the World Health Organization *Guidelines for Drinking-water Quality* (WHO, 2003). In most countries, other water uses, such as agricultural and landscape irrigation, urban and industrial uses, are not submitted to regulations. However, there are regulations and guidelines covering water for nonpotable uses that is known to have originated in wastewater (e.g. WHO 1989; US EPA 1992). Several Mediterranean countries have national regulations and guidelines; for example, France (Conseil Supérieur d'Hygiène Publique de France, 1991), Israel (Halperin, 1999), Italy (Ministero dell'Ambiente e della Tutela del Territorio, 2003), Spain (Salgot & Pascual, 1996) and Tunisia (Government of Tunisia, 1989).

As the global population increases, water pumped for potable supply from rivers, lakes and even aquifers is increasingly polluted with wastewater. Due to advances in research and analytical techniques, new threats to public health from microorganisms and chemicals are constantly being discovered, and many of them are conveyed by wastewater. Potable water regulations must be adapted to integrate new knowledge, but must not impose too great a cost burden on less wealthy countries. Whatever the source of potable water, its quality should comply with the same regulations; however, monitoring of water quality should take into account the source and any harmful substances that have been identified.

Given that the different uses of water abstracted from aquifers recharged with recycled municipal water are subject to regulations and guidelines, is it necessary to impose more requirements on aquifer recharge? This issue has been indirectly addressed by the establishment of a framework for European Community action in the field of water policy (Commission of the European Communities, 2000). The directive states that member states shall:

- implement the measures necessary to prevent or limit the input of pollutants into groundwater and to prevent the deterioration of the status of all bodies of groundwater, …
- protect, enhance and restore all bodies of groundwater, ensure a balance between abstraction and recharge of groundwater, …
- implement the measures necessary to reverse any significant and sustained upward trend in the concentration of any pollutant resulting from the impact of human activity in order progressively to reduce pollution of groundwater;
- ensure the necessary protection for the bodies of water identified with the aim of avoiding deterioration in their quality in order to reduce the level of purification treatment required in the production of drinking-water.

The main aim of these statements is to avoid and reverse any significant and sustained degradation of either the quality or quantity of aquifer water. Though these measures are primarily intended to address diffuse pollution resulting from agricultural activity, they can be applied to artificial recharge. Artificial recharge must not lead to a supplementary treatment of the groundwater pumped for drinking-water supply. Such goals are targeted by the State of California criteria for groundwater recharge (State of California, 1992). Recharge projects must be designed in such a way that they do not jeopardize the public water supply systems, including use of groundwater for potable water supply (Asano, 1992).

A similar approach could be taken for aquifers in which the water quality is too degraded for the supply of potable water, either now or in the future. The most widespread examples of such aquifers are the overexploited coastal aquifers invaded by seawater. Aquifers can be artificially recharged with recycled water to serve several water needs (e.g. agricultural and landscape irrigation, urban and industrial uses) that do not require potable water quality. The recharge should be implemented in a way such that the groundwater quality is improved and meets, on a long-term basis, the most stringent standards related to the intended water applications. Thus, recharge can improve the status of an aquifer and provide groundwater that can be useful for a range of purposes.

Australia, in its Water Quality Guidelines for Fresh and Marine Waters (NWQMS, 1992), has accepted a differential protection policy. In these guidelines, the level of protection offered at sites where recycled water is injected will depend on the potential environmental values of ambient groundwater and, therefore, on its current water quality (Dillon et al., 2001).

There is general agreement that recharge should not create a need for supplementary treatments after withdrawal for the water to meet the standards related to its intended application. Meeting the standards at the point of use is not enough; qualitative requirements have to be satisfied within the aquifer.

The status of the aquifer may not be the same throughout its whole extension. For example, large coastal aquifers contain saline water near the shoreline and high-quality fresh water inland; in which case, different policies could be adopted near the coastline and further inland. Areas invaded by seawater can be recharged by lower quality water in order to accumulate water fit for irrigation; whereas, further inland, potable water can be extracted and high quality water should therefore be maintained. However, such sector-based management is only possible if the expansion of low-quality water can be, and is, controlled.

## 2.3 Aquifer recharge requirements

### 2.3.1 Recharge for indirect potable reuse

Artificial recharge for indirect potable reuse is an attractive option that has been considered for years and has already been implemented in several countries. Most if not all well-documented cases of reclaimed water recharge for indirect potable reuse are in the United States of America (USA) (e.g. West Basin and Orange County, CA; Mesa and Tucson in AZ). The recharge should not degrade the quality of the groundwater nor impose any additional treatment after pumping. Apart from those in Australia (NWQMS, 1995), regulations concerning aquifer recharge do not rely on the capability of the aquifer to remove pollutants to meet the water quality required within the aquifer. In practice, the recharge water reaching the saturated zone of the aquifer should have previously acquired the quality acceptable for drinking-water.

If the recharge is direct, then the injected water should be potable and should, as a minimum requirement, meet the standards enforced in the country or contained in the WHO *Guidelines for Drinking-water Quality* (WHO, 1996). Moreover, the injected water should be treated to prevent clogging around the injection wells, long-term health risks linked to mineral and organic trace elements, and the degradation of the aquifer. The capacity of the aquifer to remove pollutants provides an additional barrier protecting the abstracted water quality.

Setting requirements for indirect recharge is not an easy task. The quality of infiltrated water may be dramatically improved when percolating through the vadose zone, thanks to retention and oxidation processes. These processes affect organic matter, nutrients, microorganisms, heavy metals and trace organic pollutants. However, though much is known about these processes (Bouwer, 1996; Drewes & Jekel, 1996), forecasting the efficiency of the treatment provided by infiltration through the vadose zone and lateral transfer in the saturated zone is hardly feasible. Performances depend on a number of factors such as depth of the unsaturated zone, physical and mineralogical characteristics of the soil layers, heterogeneity, hydraulic load, infiltration schedule and infiltrated water quality.

Therefore, when transfer through the vadose zone is part of the treatment intended to bring injected water up to potable water quality, a case-by-case approach is highly recommended. For each project, pollutant removal tests should be performed, at the laboratory and onsite. Every category of pollutants of concern should be considered. The example of the Dan Project in Israel shows that submitting secondary effluents to a soil aquifer treatment system in a dune sand aquifer can result in the production of a nearly potable water (Sack, Icekson-Tal & Cikurel, 2001). However, recharging potable water aquifer with secondary effluents through such treatment would not be recommended; further treatment, including microbial decontamination, would be needed to reliably obtain potable quality in the aquifer. Furthermore, relying on water transfer in the unsaturated zone to meet potable water quality would not be recommended in heterogeneous soils.

### 2.3.2 Recharge for nonpotable reuse

The quality of the water extracted from the aquifer should meet the most stringent standards related to the intended water use. In health-related standards applying to wastewater reuse, microorganisms are the main concern. For irrigation, limits can be set for other parameters such as organic matter and heavy metals . Trace organic elements are not likely to present a major harmful impact. As with potable aquifer recharge, relying on the saturated zone of aquifers to improve the recharged water quality is not recommended, even if there is no doubt that filtration effects exist. The saturated zone should only be considered as an additional barrier.

When recharge is direct, the recycled water should have been upgraded to meet the standards and limits required for the intended applications. Also, suspended solids and organic matter should have been drastically reduced to avoid clogging around the injection wells.

Indirect recharge requires a less treated injectant and is easier to implement. Soil aquifer treatment is an appropriate treatment to meet the required water quality, provided it is properly designed and managed. Prediction of the quality of the percolating water when it reaches the saturated zone is generally difficult, mainly because of the high heterogeneity of soil layers. Therefore, a detailed investigation of the hydraulic characteristics of the soil layers below the infiltration site is useful. Onsite performance tests are necessary, except in the case of dune sand layers, which are often homogeneous. When highly permeable or heterogeneous onsite soils are not able to provide the required treatment, infiltration percolation through calibrated sand beds filling pits excavated at the soil surface can be used as a treatment before infiltration through onsite soil layers (Brissaud et al., 1999).

The quality required of the recycled water applied in infiltration facilities should depend on the site, the hydraulic load, the infiltration schedule and the quality to be reached in the aquifer. A secondary treatment is a minimum. Each project must be tailored according to the local context and the water quality to be reached.

## 2.4 Conclusion

This chapter proposes a simple approach to health related guidelines, which takes into account existing water regulations.

Introduction of pollutants into aquifers may have long-term impacts; therefore, avoiding jeopardizing groundwater should be a prerequisite of any aquifer recharge project. Also, recharge should not create the need for supplementary treatments after withdrawal to meet the standards related to the intended water uses.

Distinction between potable and nonpotable aquifers is essential and will allow development of aquifer recharge and saving of water resources. Distinction is also essential between indirect recharge, using surface spreading and direct recharge through injection wells.

Several approaches, with the important exception of that used in Australia, assume that although the saturated zone can improve water quality, this factor should not be taken into account when setting regulations. The conservative approach is to consider transfer into the aquifer as an additional barrier. Therefore, when direct recharge is performed, the quality of the injected water should meet the quality required from the water that will be subsequently withdrawn from the aquifer. The implication of this is that only potable water should be injected into potable aquifers; and that when aquifer water is to be pumped for unrestricted irrigation, the injected water should meet the standards established for the reuse of wastewater for unrestricted irrigation.

In contrast to the situation with direct recharge, water quality improvement due to percolation through the unsaturated zone is taken into account for indirect recharge. However, because this improvement varies with a number of factors, the recharge design should be tailored case-by-case, after in-depth investigations and preliminary in situ tests.

Sector-based management of aquifers is appropriate in some situations, but must be accompanied by the implementation of consistent monitoring programmes.

The regulations developed by the Balearic Islands provide an example of a Mediterranean attempt to address health-related issues; they should not be regarded as a model.

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# 3 Health risk in aquifer recharge with recycled water

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## 3.1 Introduction

Water specialists in developed and developing countries work under very different conditions. Both have a shared responsibility — to minimize the health risks caused by water that people drink. Both also have a common reality — that drinking-water of acceptable quality is becoming increasingly scarce, even in countries where average water availability indices are well above scarcity thresholds. Most countries have areas where demand is higher than availability, and hardly any country can boast that all of its water is potable.

In preparing this chapter, the author has endeavoured to incorporate the most recent knowledge and the most advanced technology with experiences and best international practices in aquifer recharge. The chapter attempts to combine science with pragmatism, based on the recognition that each country (and in many cases regions within a single country) has unique hydrogeological and socioeconomic realities. In view of this situation, it is unwise to attempt to issue universal guidelines for minimizing the health risks arising from aquifer recharge using recycled water. The aims are therefore more modest: to contribute to improving practices in intentional recharge of groundwater and to introduce a precautionary approach in other activities (notably agricultural irrigation with untreated sewage and rural wastewater disposal) that result in incidental or unintentional recharge, to reduce the risks to acceptable levels for the society in question, at costs it is able and willing to assume, through actions that can be implemented via existing institutional arrangements.

### 3.1.1 Origins of the current reuse situation

In developed countries, environmental protection was the original motivation for water reuse, with technologies developed for the safe reintroduction of treated water into the environment. Subsequently, the increasing costs of supplying water from conventional sources led to the idea of using recycled water to complement primary sources for human consumption. The practice of intentional indirect reuse in the developed world has influenced the evolution of increasingly sophisticated and costly methods for treatment, detection and analysis.

There are several striking contrasts between developed and developing countries in relation to water reuse, described below.

*Incidence of waterborne disease*

In the developing world, the main motivation for reuse has been a shortage of water, which is why there are cases of indirect reuse (unintentional of course) for human consumption that are not subject to standards designed to minimize the health risks. The practice of unintentional indirect reuse in developing countries is largely responsible for the approximately 4 billion cases of diarrhoea daily that cause 2.2 million deaths a year, mainly in children under five years of age (WHO, 1999). These figures, which mean the death of one child every 15 seconds and represent 15% of all deaths in developing countries, are in stark contrast to the situation in the developed world, where deaths caused by waterborne diseases are so rare that, when they do happen, they make the headlines.

*Volume of recycled water used*

Another contrast between developed and developing countries is the volume of recycled water used. In the developed world, the total volume of treated and recycled water mixtures from intentional reuse used for human consumption is estimated to be less than 10 $m^3/s$ — barely enough to supply around 2.9 million people with 300 litres daily. The contrasting situation in the developing world is illustrated by the example of Mexico. Despite being the world's eighth biggest economy, Mexico has not been able to meet 100% of its population's drinking-water needs and is considerably backward in terms of wastewater treatment. Downstream of Mexico City, 52 $m^3/s$ of untreated water is used to irrigate 100 000 hectares, which has the effect of artificially recharging the aquifer underlying the irrigated area with 25 $m^3/s$ of raw wastewater. This aquifer has been a source of human consumption for many decades. Forty years ago, wells 50 m deep were needed to extract underground water, but now, due to the amount of infiltrate, water flows out of sources at 100–600 l/s (Jiménez et al., 2001). Nature has taken care of the recovery of the municipal wastewater, which infiltrates into the subsoil after being used for irrigation. The process of natural reclamation is so efficient that intense monitoring campaigns at the source show no evidence of dangerous pollutants, even in toxicological tests applied. Recycled water that has not been diluted with primary water, and has only been treated with chlorine, supplies 300 000 people. Similar unintentional artificial recharge situations have been observed by the British Geological Survey in Amman, Jordan; in Lima, Peru and in Hat Yai, Thailand (Foster, 2001).

*Focus of research*

A third contrast between the developed and developing worlds regarding indirect reuse relates to economic resources and the focus of research on wastewater treatment processes and effects of water consumption on human health. In developed countries, the focus is primarily toxicological: laboratory specimens are exposed to large doses of pollutants, usually one pollutant at a time, and results are extrapolated to anticipate the effects on humans exposed to much lower doses of several compounds. A problem with this approach is that the interactions between compounds that occur in the natural environment are not reproduced under the controlled conditions of the laboratory. Nevertheless, in developed countries, toxicological studies tend to be favoured over epidemiological studies, which are seen to lack statistical validity (in spite of the fact that people who have consumed primary and recycled water mixtures would manifest the effects of the interactions between the components of a water mixture).

Developing countries often do not have the economic or technological resources to undertake costly toxicological studies, and have instead undertaken epidemiological studies. As in the case of Mexico City referred to above, such studies show that simply adding chlorine significantly reduces the health risks of consuming such water mixes.

### 3.1.2 Premises on which guidance is based

The guidance provided in this chapter is based on three premises. The first is that intentional and unintentional indirect reuse should receive equal attention. For example, it is important to consider the potential health risks that may be involved in human consumption of water from aquifers under areas irrigated with groundwater, even if this process does not involve intentional indirect reuse of wastewater (Foster et al., 2002). There are procedures for wastewater reuse that should be applied, to safeguard the quality of groundwater. In other words, it is necessary to start by admitting that reuse exists, and that we should at least aspire to improve the conditions in which it takes place. Direct intentional use of recycled water is outside the scope of this chapter, but in certain conditions of extreme scarcity, such reuse will be necessary; therefore, scientific research should address the issue.

The second premise is that WHO has an obligation to contribute to improving assessment and management of health risks in the practice of indirect intentional and unintentional reuse in both developed and developing countries, and to encourage shared experiences between both groups of countries.

The third premise is that aquifers constitute a storage process, and have generally been replenished either naturally or as a consequence of human activity (mainly in an unintentional way, through agricultural irrigation with low-quality water). Compared to surface storage options, underground storage has a number of advantages:

- it does not impede or change soil use (although land ordinance restrictions are recommended for protecting the quality of groundwater;
- soils and rocky materials that form aquifers generally contribute to the natural reclamation of wastewater that infiltrates the subsoil;
- underground water is usually better protected from pollution than groundwater.

Related to this third premise is the fact that inland cities have no options but to discharge to ground or rivers and lakes that indirectly produce aquifer recharge.

Although the most common way in which unintentional reclamation occurs is through agricultural irrigation, there are also tested technologies for inducing aquifer recharge with treated wastewater. These technologies basically involve infiltrating wastewater (after a certain degree of treatment) through permeable soils that delay flow. In cases of nonpermeable soils, direct injection through wells is used, again after a certain degree of treatment. The advantages and disadvantages of the various technologies available depend on the composition of the wastewater and the geological formations in which the groundwater is held.

In the case of developed countries, the potential for indirect intentional reuse depends mainly on the degree of social acceptance. Consequently, intense communication campaigns are needed, aimed at getting society to accept the levels of investment needed to provide treatment, operations and maintenance that will ensure that health risks are reduced to socially acceptable levels. Developed countries also need to undertake epidemiological studies of wastewater reuse.

In developing countries in arid or semi-arid regions, a shortage of water has forced a situation in which water of a less than desirable quality is used. However, relatively cheap improvements in indirect intentional reuse practices have the potential to significantly reduce the health risks, especially gastrointestinal diseases. The guidelines provided in this chapter are aimed at the design and implementation of such improvements. The chapter also aim to systematize epidemiological and toxicological studies, to guard against the effects of toxic pollutants in wastewater, and includes case studies where wastewater disposal in soils has resulted in incidental recharge.

### 3.1.3 Health risks in water reuse

When water is being used as drinking source, the main aspect to consider in aquifer recharge with recycled water is public health. This chapter considers potentially problematic contaminants in situations of intentional or unintentional reuse for human consumption. Pathogens are considered for the magnitude and speed of their effects, and toxic chemicals for their long-term effects. Given the diversity and variability of the microorganisms that may be involved, each country should give priority to those that have the greatest implications for human health (i.e. high possibility of causing epidemics due to low infectious doses and high levels of persistence and resistance, considering local conditions). Special consideration should be given to certain groups of the population who may be particularly susceptible (e.g. children and the elderly). Similar considerations should be given to chemicals contaminants in recycled water.

## 3.2 Microbiological aspects

The first water epidemics caused by unplanned reuse (until now seen as pollution) illustrated that the main risks to water consumption were pathogens, in terms of the size and speed of the outbreaks. Although the risk from pathogens has been clear for some time, there is continuing uncertainty about how to ensure the microbiological safety of water, especially when the water is recycled. This is because of the many different viruses, bacteria, parasites, fungi, algae and helminths that may be present in wastewater, and thus may eventually contaminate recycled water.

To further complicate the picture, there is a recently coined term — "emerging pathogens" — that is used to describe pathogens that are not actually new, but which for some reason now cause diseases and are related to the consumption of drinking-water. In the United States of America (USA), this group includes the protozoa *Giardia lamblia*, *Cryptosporidium parvum* and *Cyclospora cayetanensis*, the fungus *Blastocystis hominis* and the bacteria *Mycobacterium avium* intracellulare or *M. avium* complex (MAC) (Jawetz, Melnick & Adelberg, 1996). WHO is currently reviewing the health risks of some water-related emerging pathogens.

As is the case for drinking-water, there are three main groups of microorganisms that can be transmitted through water consumption and should be considered in water reuse: viruses, bacteria and protozoa. Although it is possible to become infected with helminths through water consumption, it is not very likely if the recycled water is not turbid.

Table 3.1 lists the pathogens that have been found in wastewater. The table is based on a search of the literature, and shows that different studies produced varied and inconsistent information. Clearly, research is needed into the pathogens likely to cause problems in different countries (both developed and developing), and the prevalence of various microorganisms in different parts of the world.

### 3.2.1 Viruses

Viruses are the smallest infectious agents. They come in different shapes and vary in size from 0.01 to 0.3 μm in diameter. Viruses consist of either deoxyribonucleic acid (DNA) or ribonucleicacid (RNA), surrounded by a layer of protein, which may in turn be surrounded by a lipid membrane. They are obligate intracellular parasites that can only multiply inside the infected host cell; thus, they are harmless outside cells. More than 140 types of enteric virus capable of producing infections or illness multiply in the intestine and are expelled in faeces.

**Table 3.1 List of agents that cause waterborne illness, classification and illness caused**

| **Agent** | **Illness** |
|---|---|
| ***Viruses*** | |
| Adenoviruses (31 to 51 types) | Respiratory illness, conjunctivitis, vomiting, diarrhoea |
| Arboviruses | Arboviral disease |
| Astroviruses (5 types) | Vomiting, diarrhoea |
| Calcivirus or Norwalk agent | Vomiting, diarrhoea |
| Coronavirus | Gastroenteritis, vomiting, diarrhoea |
| Coxsackie A (enterovirus) | Meningitis, fever, herpangina, respiratory illness |
| Coxsackie B (enterovirus) | Myocarditis, congenital heart anomalies, rash, fever, meningitis, respiratory illness, pleurodynia |
| Echovirus (enterovirus) | Meningitis, encephalitis, respiratory illness, rash, diarrhoea, fever |
| Enterovirus 68–71 | Meningitis, encephalitis, respiratory illness, acute hemorrhagic conjunctivitis, fever |
| Flavirus | Dengue fever |
| Hepatitis A virus | Infectious hepatitis |
| Hepatitis E virus | Hepatitis |
| Norwalk virus | Epidemic vomiting and diarrhoea, gastroenteritis |
| Parvoviruses (3 types) | Gastroenteritis |
| Poliovirus (enterovirus) | Poliomyelitis, paralysis, meningitis, fever |
| Reoviruses (3 types) | Not clearly established |
| Rotaviruses (4 types) | Diarrhoea, vomiting, gastroenteritis |
| Snow Mt. agent | Gastroenteritis |
| Small and round viruses | Diarrhoea, vomiting |
| Yellow fever virus | Yellow fever |
| ***Bacteria*** | |
| *Brucella tularensis* | Tularemia |
| *Campylobacter jejuni* | Gastroenteritis, diarrhoea |
| *Escherichia coli enteropathogenic* | Gastroenteritis |
| *Legionella pneumophila* | Acute respiratory illness, legionnaire's disease |
| *Leptospira* spp. (150 types) | Leptospirosis (septic meningitis, jaundice, neck stiffness, haemorrhages in the eyes and skin) |
| *Clostridium perfringens* | Gaseous gangrene, food poisoning |
| *Mycobacterium leprae* | Leprosy |
| *Mycobacterium tuberculosis* | Pulmonary and disseminated tuberculosis |
| *Salmonella* (1700–2400 strains — *parathyphi*, *schottmuelleri*, etc.) | Salmonellosis |
| *Salmonella thyphimurium* | Typhoid fever, paratyphoid or salmonellosis |
| *Shigella* (4 types) | Bacillary dysentery, shigellosis |
| *Treponema pallidum-pertenue* | Yaws (frambuesia) |
| *Yersinia enterocolitica* | Gastroenteritis, yersiniosis |
| *Vibrio cholerae* | Cholera |
| ***Fungi*** | |
| *Aspergillus fumigatus* | Aspergillosis |
| *Candida albincans* | Candidiasis |
| ***Protozoa*** | |

| Agent | Illness |
|---|---|
| *Balantidium coli* | Mild diarrhoea, colonic ulceration, dysentery, balantidiasis |
| *Cyclospora cayetanensis* | Severe infectious, dehydration: diarrhoea, nausea, vomiting |
| *Cryptosporidium parvum* | Diarrhoea and cryptosporidiosis |
| *Entamoeba histolytica* | Amoebic dysentery |
| *Giardia lamblia* | Giardiasis |
| *Naegleria fowleri* | Amoebic meningoencephalitis |
| *Plasmodium malariae* | Malaria |
| *Trypanosoma spp.* | Trypanosomiasis |
| *Toxoplasma gondii* | Congenital or postnatal toxoplasmosis |
| ***Helminths*** | |
| *Ancylostoma duodenale* | Anaemia, ancylostomiasis |
| *Ascaris lumbricoides* | Ascariasis |
| *Echinococcus granulosis* | Hyadatidosis |
| *Enterobius vermicularis* | Enterobiasis |
| *Necator americanus* | Anaemia |
| *Schistosoma* spp. | Schistosomiasis |
| *Strongyloides stercoralis* | Diarrhea, abdominal pain, nausea, strongylodiasis |
| *Taenia solium* | Taenisis, cysticercosis |
| *Trichuris trichiura* | Diarrhoea |
| *Toxocara* | Fever, abdominal pain, nausea |

Adapted from: California Department of Health and Cooper (1975); Asano et al. (1998); Craun (1988);. Kadleck & Knight (1995); Jawetz, Melnick & Adelberg (1996).

Pathogenic viruses, unlike bacteria, are not usually found in the waste of healthy human beings, only those who are intentionally exposed (e.g. through vaccination) or are infected through water and food. The time it takes to expel viruses varies considerably, and expulsion may be constant if the virus is endemic to a given community. In the case of infection, the viruses are found in large quantities, for example rotavirus can be found at concentrations of up to $10^{12}$/g of faeces (Flewett, 1982).

The presence of viruses and their concentration in wastewater vary widely, and are linked to the season and to the age distribution of the population (e.g. concentrations are usually high during summer and low in the autumn). Few studies have identified the composition and type of viruses present in wastewater and recycled water; studies are mostly restricted to enteroviruses, because they are easy to analyse (Leong, 1983).

The enteric viruses most relevant to humans are enteroviruses (polio, echo and coxsackievirus), Norwalk, rotavirus, reovirus, calicivirus, adenovirus and hepatitis A. Enteroviruses are high-risk for a number of reasons: only a relatively low dose is required to cause illness, they are more resistant than most bacteria to the environment and disinfection, and it is difficult to quantitate them using conventional laboratory techniques. Consequently, recycled water should essentially be free of enteroviruses, which is why emphasis is placed on this type of microorganism here and in the sections on treatment for aquifer recharge (Section 3.5) and on regulatory framework (Section 3.6).

Rotaviruses are the biggest cause of infant gastroenteritis worldwide. They are responsible for 0.5–1 billion cases of diarrhoea per year in children under five years in Africa, Asia and Latin America, and cause up to 3.5 million deaths. Usually, 50–60% of cases of gastroenteritis in children that result in hospitalisation are caused by this virus (Jawetz, Melnick & Adelberg, 1996). Rotaviruses are closely related to reoviruses.

Reoviruses and adenoviruses, which are the main causes of respiratory illness, gastroenteritis and eye infections, have been isolated from wastewater. To date, there is no evidence that human immunodeficiency virus (HIV), the virus that causes acquired immune deficiency syndrome (AIDS), can be transmitted via water, although its presence is considered feasible. It is possible that HIV has not been detected in water because of its low concentration (Kadlec and Knight, 1996).

Regarding recharge, viruses that have migrated long distances in aquifers have been isolated. Horizontal migration varies between 3 and 400 m while vertical migration varies between 0.5 and 30 m, depending on soil conditions (Table 3.2).

Low levels of viruses may cause an infection or illness, and wastewater contains thousands of them, some of them much more resistant to disinfection than bacteria. Therefore, monitoring viruses in recycled water is very important. However, there is no consensus as to the meaning of public health when recycled water contains low levels of viruses, even when there is information about how many remain after different stages of treatment.

**Table 3.2 Viruses: transportation in places where wastewater is applied**

| Location | Type of: virus | wastewater | soil | Distance transported (m): Vertical | Horizontal | Reference |
|---|---|---|---|---|---|---|
| Canning Vale, Australia | Echovirus 11 | Secondary effluent | Bassen dean sand | 9 | 14 | Janson et al. (1989) |
| | Echoviruses 14,24,29,39, | | | 3 | | |
| | Coxsackie virus B4; adenovirus 3 | | | 2 | | |
| | Poliovirus 1 | | | 1 | | |
| | Echoviruses 6; poliovirus 2; coxsackie virus B5 | | | 0.5 | | |
| Tucson, AZ | Bacteriophages MS2 and PRD | Secondary effluent<br>Tertiary effluent | Coarse alluvial sand and gravel | 4.6<br>6.1 | | Powelson, Gerba & Yahua (1993) |
| Gainsville, FL | Coxsackie viruses B4, polioviruses 1,2 | Secondary effluent | Sand | 3 | 7 | Wellings, Lewis & Mountain (1976) |
| Tucson, AZ | Bacteriophage PRD1 | Secondary chlorinated effluent | Coarse alluvial sand and gravel | 6.1 | 46 | Gerba et al. (1991) |
| Kerville, TX | Enteroviruses | Secondary effluent | Loam to clay | 1.4 | | Duboise et al. (1976) |
| East Meadow, NY | Echoviruses 12 | Secondary effluent | Coarse sand and fine gravel | 11.3 | 3 | Vaughn et al. (1976) |
| Holbrook, NY | Echoviruses 6, 21,24, 25 | Tertiary effluent | Coarse sand and fine gravel | 6.1 | | Vaugh et al. (1978) |
| Fort Devens, MA | Bacteriophage12 | Secondary effluent | Silty sand and gravel | 28.9 | 183 | Schaub & Sorber (1977) |
| Phoenix Az | Coxsackie virus B3 | Secondary effluent | Fine loamy sand over coarse sand and gravel | 18.3 | 3 | Gerba, (1982) |
| Colton, CA | Enteroviruses | Secondary effluent | Coarse sand | 24.4 | | Feachem et al. (1993) |
| Lubbock, TX | Coxsackie | Secondary | Loam | 1.4 | | Goyal et |

| Location | Type of: virus | wastewater | soil | Distance transported (m) Vertical | Horizontal | Reference |
|---|---|---|---|---|---|---|
| | virus B3 | effluent | | | | al. (1984) |
| San Angelo, TX | Enteroviruses | Secondary effluent | Clay loam | 27.5 | | Goyal et al. (1984) |
| Muskegon, MI | Enteroviruses | Chlorinated aerated effluent | Rubicon sand | 10 | | Goyal et al. (1984) |
| 12 Pines, NY | Poliovirus 2 | Tertiary effluent | Coarse sand and fine gravel | 6.4 | | Vaugh et al. (1981) |
| Northe Massapequa NY | Echovirus 11, 23, coxsackie viruses A16 | Storm water | Coarse sand and fine gravel | 9.1 | | Vaughn and Landry (1977) |
| Vineland, NJ | Poliovirus, echovirus, coxsackie virus B3 | Primary effluent | Coarse gravel and sand | 16.8 | 250 | Koerner and Haws (1979) |
| Montebello, Forebaay CA | Male specific coliphages | Tertiary effluent | Alluvial sand | | 7.6 | Yanko (1994) |

Source: Yates & Gerba (1998)

Tertiary treatment processes consisting of coagulation, flocculation, sedimentation, filtration and disinfection are effective at removing viruses. Filtration removes approximately 4.7 logs and disinfection is approximately of 5.2 logs when the residual chlorine concentration is from 5 to 10 litres per hour, with a 2 hour contact time, (CSDOLAC, 1977; Tanaka et al., 1998).

### 3.2.2 Bacteria

Bacteria are microorganisms of between 0.2 and 10 µm in length, that are single celled, and can reproduce and grow in appropriate conditions (e.g. of temperature, salinity and pH). Bacteria are ubiquitous; species form intestinal colonies and are expelled in large numbers (> $10^{12}$ per gram of faeces), although most are not pathogens. Bacteria that are enteric (i.e. that live, or can live, in the intestines) and pathogenic pose the biggest risk.

Pathogenic bacteria are present in the faeces of infected individuals. One of the most common pathogens found in municipal wastewater is the genus *Salmonella*. The *Salmonella* group contains a wide variety of species harmful to man and animals, and an infected individual can expel up to $10^9$ of these organisms per gram of faeces (Bitton, 1994). The most severe form of salmonellosis is typhoid fever, caused by *Salmonella typhi*. A less common relative is *Shigella*, which produces bacillary dysentery or shigellosis, through swimming in polluted water.

Thermotolerant coliforms are commonly used as indicators of faecal contamination and the potential presence of pathogens. This group responds in a similar way to the environment and treatment as most bacteria, but is unable to simulate the behaviour of viruses or protozoa. In particular, thermotolerant coliforms may be absent in water where the protozoa *Giardia lamblia* and *Cryptosporidium parvum* are present. In relation to aquifer recharge, bacteria are easily retained in the soil; some studies suggest that only 8 cm is required for them to be separated from water during infiltration (Feachem et al., 1983). The section below discusses the pathogenic bacteria relevant to water reuse.

*Escherichia coli*

Traditionally, *E. coli* strains implicated in diarrhoea have been divided into four groups:

- enteropathogenic *E. coli* (EPEC);
- enterotoxigenic *E. coli* (ETEC) — strains that produce heat-labile or heat-stable enterotoxins;
- enteroinvasive *E. coli* (EIEC) — strains capable of invading the intestinal mucus lining (e.g. *Shigella*);
- enterohemorrhagic *E. coli* (EHEC) — strains that produce a similar toxin to Shiga ("Shiga coli") that can cause hemorrhagic colitis.

Recently, a new functional classification has been introduced. Under this scheme, *E. coli* strains that cause diarrhoea and those that do not produce toxins have been grouped according to their adherence to Hep-2 cells, being classified as locally adherent *E. coli* (LAEC) or diffusely adherent *E. coli* (DAEC) (Sansonetti, 1992).

The different groups of *E. coli* can cause gastroenteritis in both animals and humans, and pose a particular risk to newborns and children under five years of age. ETEC is the common cause of traveller's diarrhoea, which is liquid and profuse with some mucosity; symptoms also include nausea and dehydration. The main problem lies in the fact that small doses ($10^2$ organisms) are infectious, and these organisms could cause a problem in recycled water.

*Pseudomonas*

Some species of *Pseudomonas* are opportunistic pathogens; for example, *P. cepacia* have been cultivated from patients suffering from cystic fibrosis, and *P. mallei* causes a fatal infection in humans. The illness begins as skin ulcer or in mucus followed by lymphagitis and sepsis. Inhaling these microorganisms can cause primary pneumonia (Jawetz, Melnick & Adelberg, 1996).

*Campylobacter jejuni*

*Campylobacter* has an incubation period of 2–5 days and affects mainly children and young people. The organisms are typically bacillary curved shapes measuring 2–5 µm in length and consisting of 2–6 coils (Nachamkin, Blaser & Tompkins, 1992).

*Campylobacter jejuni* is usually a pathogen in animals, but has been identified as a cause of diarrhoea in humans (Craun, 1988). Worldwide, it is one of the most common causes of severe gastroenteritis and in Europe is the main cause of gastroenteritis. The main source of infection is nonchlorinated water supplies.

*Salmonella*

*Salmonella* are gram-negative bacilli that are abundant in different environments. They are perhaps the most relevant group of pathogens for both humans and animals, due to great many strains that exist. Typical symptoms of salmonellosis are chronic gastroenteritis, with diarrhoea, stomach cramps, fever, nausea, vomiting, headache and, in severe cases, collapse and death. The incidence in humans is lower than in animals, and the seasonal variation is different. Certain strains are harmful to man and their frequency varies from one year to another and from one country to another. In developing countries, food or water may contain high numbers of *Salmonella*. An infective dose of *Salmonella* varies between $10^5$ and $10^8$ microorganisms, although there have been cases of infections caused by average concentrations of $10^3$ (and sometimes as few as $10^2$), especially in the case of *Salmonella typhi* (Lima & Lima, 1993).

*Shigella*

This bacterium is similar to *Salmonella*, except that only rarely does it infect animals and it does not live long in the environment. *Shigella* are gram-negative, thin rods that eventually take on cocobacillary shapes. There are more than 40 different strains, but *S. sonnei* and *S. flexeneri* represent almost 90% of total wastewater isolations. Shigellosis often begins with light watery diarrhoea that can develop into full-blown dysentery. The symptoms are usually limited to the infected person; however, shigellosis can become serious and complicated in children and adults. Fever, nausea, vomiting and abdominal pain, migraine and myalgia are frequent manifestations of infection with this bacterium. The classic form of dysentery caused by *Shigella* is characterized by the expulsion of faeces containing blood, with or without mucus. An infection caused by *Shigella* can easily be passed on. The infectious dose is much lower than for *Salmonella*, being less than $10^3$ microorganisms (Sansonetti, 1992; Jawetz, Melnick & Adelberg, 1996).

*Mycobacterium tuberculosis*

*Tuberculosis* bacilli are about 0.4 × 3 µm (Jawetz, Melnick & Adelberg, 1996). In artificial environments they form cocci and filaments with a morphology that varies from one species to another. *M. tuberculosis* has been isolated from wastewater and in known to cause illness in people that swim in polluted water (California Department of Health and Cooper, 1975). *M. tuberculosis*, *M. balnei* (marinum) and *M. boris* cause pulmonary and disseminated tuberculosis. In the case of *M. tuberculosis*, contaminated water is the main source of infection.

*Vibrio cholera*

*Vibrio* species are curved gram-negative bacilli in the form of comma; they are 2–4 µm long and move using a polar flagellum. These aerobes and facultative anaerobes form part of the water environment; their presence depending on temperature and the degree of salinity. Gastroenteritis caused by *Vibrio* can be choleric or noncholeric. Epidemics mainly affect infants and are caused by *V. cholerae* strain groups O1 and O139, and some *V. cholerae* non-O1 strains. The main clinical symptoms are abundant liquid diarrhoea, with significant loss of hydroelectrolytes and severe dehydration associated with vomiting. *V. cholerae* is rare in developed countries but frequent in the developing world. Humans are the only known hosts and the most frequent vehicle for transmission is water, either through direct consumption or through products irrigated with dirty or polluted water.

*Helicobacter*

Organisms in the genus *Helicobacter* are helicoidal, curved or straight unbranched gram-negative rods, 0.3–1.0 µm wide and 1.5–5.0 µm long. There are at least nine species within the genus *Helicobacter*, as detailed in Table 3.3. The faeces of birds and pigs contain helicobacter-like organisms, which may be designated new species after further evaluation. The major habitat of *H. pylori* is the human gastric mucosa. Three species of *Helicobacter* are significant human pathogens: *H. pylori* (previously named *Campylobacter pylori* and *C. pyloridis*), *H. fennelliae* (previously named *C. fennelliae*), and *H. cinaedi* (previously named *C. cinaedi*). *H. pylori* is found worldwide and appears to be acquired by the faecal–oral or the oral–oral route (Graham et al., 1991). The prevalence of *H. pylori* appears to increase with age (Al-Moagel et al., 1990; Graham et al., 1988), but there is little information on the prevalence of this organism in any defined population, or on the factors that may influence the pattern of distribution.

In developing countries, *H. pylori* is acquired early in childhood, withup to 90% of children infected by 5 years of age (Thomas et al., 1992). In the USA, few infections occur during childhood, and the organisms have a projected incidence of about 0.5–1.0% per year, with about 50% incidence at age 60 (Blaser, 1991). The highest incidence in both developing and developed countries is in poorer, lower socioeconomic groups, for which crowding and poor sanitation are risk factors.

**Table 3.3 *Helicobacter* species and related organisms**

| Species | Hosts | Source or habitat |
|---|---|---|
| *H. pylori* | Humans | Gastric mucosa |
| *H. mustelae* | Ferrets | Gastric mucosa |
| *H. felis* | Cats, dogs | Gastric mucosa |
| *H. nemestrinae* | Macaque monkeys | Gastric mucosa |
| *H. murderer* | Rats, mice | Intestinal mucosa |
| *H. acinonyx* | Cheetahs | Gastric mucosa |
| *H. cinaedi* | Humans, rodents | Intestinal mucosa |
| *H. fennelliae* | Humans | Intestinal mucosa |
| *"H. rappini"*[a] | Sheep, dogs, humans | Liver (sheep) Stomach (dogs), Faeces (humans) |
| *Gastrospirillum hominis*[b] | Cheetahs, humans | Gastric mucosa |

[a] Previously *Flexispira rappini*. A proposal to place the organisms within the genus *Helicobacter* has been made.
[b] Unculturable helix-shaped bacteria found by McNulty et al. (1989) in stained gastric biopsy specimens. RNA sequencing places the organism in the genus *Helicobacter*, and the name *H. heilmanii* has been proposed
Source: Jerris (1995)

*H. pylori* may be a risk factor for gastritis, duodenal (peptic) ulcers, gastric ulcers and cancer. *Helicobacter* has a $C_T$[3]$_{99}$ of 0.12 mg/min.$l^{-1}$ of chlorine, whereas that for *E. coli* is 0.1–0.2 mg/min.$l^{-1}$ (Johnson, Rice & Reasoner, 1997). Therefore, thermotolerant coliforms can be considered as an indicator of *Helicobacter*, although there is some disagreement on this (Mazari-Hiriart et al., 2001).

### 3.2.3 Protozoa

Protozoa are the parasites most commonly associated with diarrhoea. They are single-celled organisms, 2–60 μm in size, that develop as either trophozoites or cysts. Infection results from the consumption of a mature cyst, which is resistant to the gastric juices. The cysts break down in the small intestine, where they are transformed into trophozoites that become imbedded in the wall of the intestine, feeding on bacteria and dead cells. The trophozoites can become cysts again and can then be expelled in the faeces. The infected person may have symptoms of the illness or may have no apparent symptoms.

Like viruses, protozoa do not reproduce in the environment. However, they are able to survive and remain active for weeks, months or even years, depending on environmental conditions (Bausum et al., 1983). Most intestinal protozoa are transmitted through water and contaminated food. In well-treated recycled water, protozoa are unlikely to occur, but they may be found in cases of unplanned wastewater infiltration.

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[3] $C_T$ is the disinfectant concentration (C) that needs to be applied for a certain time (t) to achieve 99% destruction of the target microorganism.

*Amoebae*

One of the most important protozoan parasites detected in municipal wastewater is *Entamoeba histolytica*, a eukaryote with single-celled trophozoites of 20–40 µm in diameter and cysts of 10–16 µm. Amoebae are usually present in the large intestine, but occasionally they penetrate the intestinal mucus and spread to other organs. They are the cause of amoebic and hepatic dysentery. *E. histolytica* is present in 10% of the world's population, resulting in approximately 500 million infected persons, 40–50 million cases of invasive amebiasis a year and up to 100 000 annual deaths (placing it second only to malaria in mortality caused by protozoan parasites). Of these cases, 96% occur in developing countries, especially in the Indian subcontinent, West Africa, the Far East and Central America. Prevalence depends on culture, age, cleanliness, community size and socioeconomic conditions (WHO, 1997; Tellez et al., 1997; Ravdin, 1995). In developed countries, if dysentry occurs it is mainly among immigrants (Salas, Heifetz & Barret, 1990).

*Cryptosporidium*

Oocysts of C*ryptosporidium* are round and measure 4–7 µm, making them difficult to remove from water through conventional processes. Although the infectious dose varies from 1 to 10, outbreaks have always been associated with high concentrations of *Cryptosporidia* in water. The main symptoms of cryptosporidiasis are stomach cramps, nausea, dehydration and headaches.

This parasite is widespread in nature. It infects a wide range of farm animals and pets, and was recently discovered to be a human pathogen. The first known case of cryptosporidiosis occurred in 1976 (Gray, 1994). *Cryptosporidium* forms an oocyst, which enables the parasite to survive for long periods of time until a host is found. The oocyst is also capable of completing a cycle within a host and causing reinfection. An infected individual can carry the parasite for life and can be subject to relapses. In England, *Cryptosporidium* is thought to be responsible for 2% of all cases of diarrhoea, and the organism is common in groundwater (Gray, 1994). The first outbreak of *Cryptosporidium* in the USA occurred in 1984 in Texas and caused 2000 cases. The source of the outbreak was an old well, which illustrates the possible dangers from poorly managed unintentional recharge.

Reactions to infection with *Cryptosporidium* vary; for example, in 1993m the organism caused an epidemic of 400 000 cases in the USA. This outbreak was probably due to contaminated water, although there are no quantitative data on *Cryptosporidium* oocysts in the water (Goldstein et al., 1996). The presence of these protozoa in the water supply was not demonstrated until a year after the outbreak (after analysing a sample from a lake taken from a depth of 40 m). In contrast, *Cryptosporidium* has been detected in the water supply in Mexico City but no associated illness has been reported; nor has the mortality rate from intestinal illness increased above the national average (Cifuentes et al., 2002).

***Giardia***

Like *Cryptosporidium*, *Giardia* is present in the intestines of a large number of animals, where it lives as a trophozoite. Cysts can survive in water bodies for long periods of time, especially in winter. *Giardia* cysts are ovals, 8–14 µm long and 7–10 µm wide.

Giardiasis is a worldwide endemic with prevalence rates of 10% in developed countries and 20% in some developing regions, where it especially affects children under 5 years who are malnourished. The total number of infected people is 1.1 billion, with 87% living in developing countries. According to WHO (1997), the number of infections has been increasing in recent years.

*Giardia* is the most common parasite in man, even though water is not necessarily the main means of dissemination. Between 1980 and 1985, there were 502 outbreaks of giardiasis, of which 52% were due to *Giardia*. The infection has an incubation period of 1–4 weeks. The disease is characterized by very liquid and smelly explosive diarrhoea, stomach and intestine gases, nausea and loss of appetite. Unlike *Cryptosporidium*, *Giardia* can be treated with various drugs, but the only way to prevent infection is through adequate water treatment. New safe drinking-water regulations in the USA now require that groundwater be filtered and disinfected in order to control both pathogens.

In developed countries, the problems in drinking-water caused by *Giardia* and *Cryptosporidium parvum* have not involved recycled water (Yates & Gerba, 1998). Cysts (of *Giardia lamblia*) and oocysts (of *Cryptosporidium parvum*) are more difficult to deactivate with chlorine than viruses and bacteria.

### 3.2.4 Helminths

Helminths are not normally transmitted in drinking-water, but are considered here because of their very high incidence and the problems they could potentially cause in drinking-water sources, particularly in countries with poor sanitation. Helminths are multicellular organisms. The free-living larvae are not usually pathogenic. Those present in wastewater are pathogenic, although they are of little health significance to intentional water recycling and reuse, because their large size is associated with the existence of particles expressed as suspended solids or turbidity (Jiménez & Chávez, 1999). Intestinal parasites cause anaemic malnutrition and can retard growth. In the case of unintentional or deficient recharges, they could pose a problem if the water is turbid.

Although there are different types of helminths, whose relative frequency depends on regional conditions, *Ascaris* almost always dominates (Fig. 3.1). Water treatment studies frequently use *Taenia saginata* as a model, despite it being a rare genus of helminths(Feachem et al., 1983; Pike, Morris & Carrington, 1983). In developing countries, levels of helminthiasis can be as high as 25–33% of the population (Bratton & Nesse, 1993; Wani & Chrungoo, 1992). As Table 3.4 illustrates, helminths are present in around 650 million individuals (Khuroo, Zargar & Mahajan, 1990), whereas in developed countries they are present in no more than 1.5% of people (WHO, 1997). Ascariasis is endemic in areas of Africa, Central America, South America and the Far East, where there is poverty and poor sanitary conditions. Incidence rates reach 90% in some parts of the world (Schulman, 1987; Bratton & Nesse, 1993).

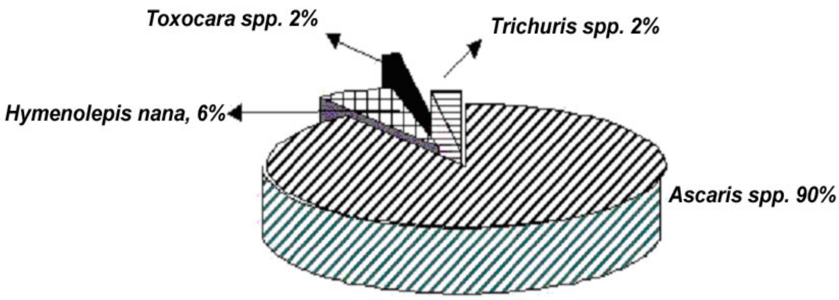


**Figure 3.1 Types of helminths most common in Mexico City wastewater (Jiménez & Chávez, 1998)**

**Table 3.4 Estimate of the size and global scope of intestinal parasites**

| Disease | Number of infected people (million) | Annual cases (million) | Annual deaths |
|---|---|---|---|
| Amebiasis | 500 | 40–50 | 40 000–100 000 |
| Giardiasis | 200 | 0.5 | |
| Ascariasis | 800–1000 | 1 | 20 000 |
| Hookworm infection[a] | 700–900 | 1.5 | 50 000–60 000 |
| Trichuriasis | 500 | 0.1 | |

[a] Includes *Ancyslostoma duodenale* and *Necator americanus* infections.
Source: WHO (1997); Salas, Heifetz & Barret (1990).

Ascariasis is common in the USA, but of the 4 million people infected, the biggest percentage are immigrants from developing countries (Bratton and Nesse, 1993). Figures for the prevalence of pathogenic intestinal parasites in this sector of the population vary between 20% and 60% (Salas, Heifetz & Barret 1990).

The ova and larva of the helminths are resistant to various environmental conditions and traditional disinfection methods, but can be removed by sedimentation, coagulation and flocculation (Jiménez-Cisneros, Maya-Rendòn & Salgado-Velázquez, 2001), filtration (Jiménez et al., 2000), wetlands (Rivera et al., 1994) and stabilisation ponds (WHO, 1989).

### 3.2.5 Infectious dose

A pathogen's ability to infect depends on a large number of factors. For example, both the host and the pathogen are living creatures and therefore do not all respond to environmental factors in the same way.

Data on infectious doses are not very precise, because, for an infectious agent to cause disease it must not only be present in a sufficient number but the individual must also be susceptible to the infection, which in turn depends on factors such as the degree of immunity, genetics and nutrition. Usually, infectious doses for organisms other than enteroviruses are determined by exposing a group of individuals or animals to different doses of microorganisms. Studies on humans normally use groups of young, healthy volunteers; thus, the results may not be applicable to other populations such as children, elderly people and those who are malnourished. A complicating factor is that in developing countries, populations frequently exposed to unhealthy living conditions often have stronger immunity to some types of microorganism (e.g. pathogenic bacteria), in spite of poor living conditions and lack of medical attention.

A further problem in determining infectious doses is the ability of microorganisms to form colonies within agglomerates. Analytical method may measure such agglomerates as if they were a single element instead of several. One aspect that has been given little attention is exposure to a group of microorganisms rather than to just one. Finally, most of the studies are undertaken using pathogens isolated and grown in laboratories, although in nature they are almost always mixed with other organisms and adapted to the environmental conditions.

Data on infectious doses are given in Table 3.5. It shows that there is wide variation between different studies for any given type of microorganism and also between different groups of microorganisms.

**Table 3.5 Infectious dose**

| Organism | Infectious dose | Reference |
|---|---|---|
| Enteric viruses | 1–10<br><10 | Feachem et al. (1983)<br>Kadlec & Knight (1996) |
| *Campylobacter jejuni* | $10^6$ | Kadlec & Knight (1996) |
| *Clostridium perfrigens* | $1–10^{10}$ | Feachem et al. (1983) |
| *Escherichia coli* (enteropathogenic) | $10^6–10^{10}$<br>$10^6– 10^{10}$<br>100 | Crook et al. (1998)<br>Feachem et al. (1983)<br>Gray (1994) |
| *Salmonella typhi*<br>*Salmonella* species | $10^4–10^7$<br>$10^3$<br>$10^5–10^7$ | Kadlec & Knight (1996)<br>Feachem et al. (1983)<br>Cooper & Olivieri (1998)<br>Gray (1994) |
| *Shigella* species | 100– 1000<br>10– 100 | Cooper et al. (1998)<br>Shiaris (1985)<br>Kadlec & Knight (1996) |
| *Shigella flexneri* | 180 | Feachem et al. (1983) |
| *Shigella dysenteriae* | 20<br>10 | Feachem et al. (1983)<br>Crook et al. (1998) |
| *Vibrio cholerae* | $10^3–10^7$<br>$10^8$<br>$10^8–10^9$ | Feachem et al. (1983)<br>Kadlec & Knight (1996)<br>Gray (1994) |
| *Yersinia* | $10^9$ | Kadlec & Knight (1996) |
| *Balantidium coli* | 25–100 | Kadlec & Knight (1996) |
| *Cryptosporidium parvum* | 1–10 | Rose (1990) |
| *Entamoeba histolytica* | 20<br>10–100 | Feachem et al. (1983)<br>Crook et al. (1998)<br>Kadlec & Knight (1996) |
| *Giardia lamblia* | 10<br>< 10<br>25–100 | Feachem et al. (1983)<br>Crook et al. (1998)<br>Kadlec & Knight (1996) |
| ***Helminths*** | | |
| *Ascaris lumbricoides* | 1–10 | Feachem et al. (1983) |
| *Hymenolepis nana* | 1 | Kadlec & Knight (1996) |
| *Trichuris trichiura* | 1 | Kadlec & Knight (1996) |

Evaulation of the risk from waterborne pathogens is complex. In the case of water reuse for human consumption, little is known about the significance of low concentrations of microorganims in mixtures, and the information that is available is often contradictory (Hass, 1983). Clearly, establishing dose-response relationships oversimplifies what occurs at different levels within a population. For example, even when exposing uniform populations of healthy individuals to the same dose, a wide variety of responses can be obtained.

This does not mean to say that water should not be reused simply because of the difficulties of determining the infectious dose. There are many areas where water with a poor microbiology quality is consumed. Based on studies in these areas, it would be possible to obtain data with sufficient precision, related to different sectors of the population, for a wide variety of pathogens.

### 3.2.6 Levels of pathogenic microorganisms in wastewater

Table 3.6 presents data on the levels of different microorganisms in wastewater. The data vary between different studies, but the enormous difference in levels of pathogens between developed and developing countries is clearly evident, which is a reflection of the very different health standards. This can be seen from the headings where the countries have been identified.

### 3.2.7 Survival of pathogens in the environment

If water from intentional or unintentional artificial aquifer recharge is to be used for human consumption, then not only the presence, but also the survival, of pathogens in the environment is important. Survival is very variable for each group and genus and depends on:

- humidity — a dry environment kills microorganisms
- the content of organic matter — its presence favours survival
- temperature — greater resistance to low temperature
- pH — bacteria survive better in alkaline soils than in acid ones
- rain — the presence of water and soil saturation promotes mobility
- sunlight — it disinfects
- foliage protection
- competition between native flora and fauna.

**Table 3.6 Levels of pathogenic microorganisms in wastewater**

| Agent | Content | Country | Reference |
|---|---|---|---|
| *Campylobacter* species | 3700 | Germany | Holler & Waltraud (1988) |
| Thermotolerant coliforms (MPN/100 ml) | $10^7$–$10^9$ | Mexico | Jiménez, Chávez & Capella (1997); Jiménez, Chávez & Hernández (1999) |
| | $10^3$–$10^5$ | USA | Stover et al. (1995) |
| | $10^3$–10 | Japan | Fusishi et al. (1997, 1999) |
| | $10^4$–$10^7$ | Egypt | Stott et al. (1997) |
| | $10^4$–$10^6$ | USA | Berg & Metcalf (1978) |
| | $10^5$–$10^7$ | USA | Geldreich (1978) |
| | $10^6$–$10^7$ | USA | Davis (1979) |
| | $10^7$ | Brazil | Mara & Silva (1979) |
| | $10^5$–$10^8$ | Developing countries | Feachem et al. (1983) |
| | $10^8$ | Bangladesh | Daniel & Lloyd (1980) |
| *E. coli* (MPN/100 ml) | $10^6$–$10^7$ | Scotland | Wheater et al. (1980) |
| | > $10^8$ | Kenya | Evison & James (1973) |
| | $10^4$ | South Africa | Grabow & Nupen (1972) |
| Total coliforms (MPN/100 ml) | $10^7$–$10^{10}$ | England | Geldreich (1978) |
| *Clostridium perfringens* | $10^3$–$10^5$ No./100ml | USA | Feachem et al. (1983) |
| *Cryptosporidium parvum* (oocysts/l) | 0.91–28 | USA | Rose (1988) |
| | 2 x $10^2$ | USA | US EPA (1991, 1992b); sCooper & Olivieri (1998) |
| | $10^3$ | USA | Mayer and Palmer (1996) |
| | 4.1–51 800 | USA | Madore et al. (1987) |
| | 1–$10^3$ | Worldwide | Feachem et al. (1983) |
| | 10–170 | England | Bukhari et al. (1997) |
| | 2.5–8000 | England | Parker (1993); Carrington & Gray (1993) |
| | 13–73 | Kenya | Grimason et al. (1993) |
| *Entamoeba 0histolytica* (cysts/l) | 1–10 No./100ml | USA | Feachem et al. (1983) |
| | 4–28 | Israel | Kott and Kott (1969) |
| | 52 | USA | Wang and Dunlop, 1954 |
| Enteric viruses (vu/l) | 3 x $10^4$ PFU | USA | US EPA (1991, 1992b); Cooper & Olivieri (1998) |
| | $10^3$ | USA | Heyward et al. (1979) |
| | 27–$10^4$ | USA | Fujioka and Loh (1978) |
| | $10^3$–$10^5$ | Israel | Feachem et al. (1983) |
| | 600–$10^6$ | Israel | Buras (1976) |
| Enterococci (No./100ml) | $10^5$–$10^6$ | USA | Davis (1979) |

| Agent | Content | Country | Reference |
|---|---|---|---|
| Faecal streptococci (No./100ml) | $10^4$–$10^6$ | USA | Geldreich (1978) |
| | $10^6$ | Brazil | Mara and Silva (1979) |
| | > $10^7$ | Kenya | Evison and James (1973) |
| *Giardia lamblia* (cysts/l) | 2 x $10^2$ | USA | US EPA (1991, 1992b); Cooper & Olivieri (1998) |
| | 1–$10^3$ | Worldwide | Feachem et al. (1983) |
| | 10–13 600 | England | Bukhari et al. (1997) |
| | 0.095–43 907 cysts/l | England | Parker (1993); Dawson et al. (1994); Robertson, Smith & Patton (1995) |
| | 0.075–14 000 cyst/l | USA | Rose (1988), Sykora et al. (1991); Robertson, Smith & Patton (1995) |
| | 213–6312 | Kenya | Grimason et al. (1993) |
| | 0.13–0.3 | USA | Ongerth (1990) |
| | $10^4$ | USA | Mayer & Palmer (1996) |
| | $10^3$–$10^5$ | USA | Jakubowski & Ericksen (1979) |
| *Pseudomonas aeruginosa* (No./100ml) | $10^3$–$10^4$ | Scotland | Wheater et al. (1980) |
| *Salmonella* species (MPN/100 ml) | $10^6$–$10^9$ | Mexico | Jiménez, Chávez & Capella (1997); Jiménez, Chávez & Hernández (1999) |
| | $10^3$–$10^4$ | USA | US EPA (1992b) |
| | 4 x $10^3$ | USA | US EPA (1991, 1992b); Cooper & Olivieri (1998) |
| | 2 | Finland | Hirn (1980) |
| | 2–41 | South Africa | Grabow & Nupen (1972) |
| | 7–250 | India | Phirke (1974) |
| | 500 | USA | Davis (1979) |
| | 670 | Holland | Kampelmacher & van Noorle Jansen (1970) |
| | 7240 | England | Jones (1977) |
| | 8000 | USA | Davis (1979) |
| | 2.0–8000 | Worldwide | Feachem et al. (1983) |
| *Shigella* species (No./100ml) | 1–$10^3$ | USA | Feachem et al. (1983) |
| Helminth ova (No./l) | 8 x $10^2$ | Syria | Bradley & Hadidy (1981)[a] |
| | 1–8 | USA | US EPA (1992b) |
| | 9 | France | Schwartzbrod et al. (1989) |
| | 6–42 | Egypt | Stott et al. (1997) |
| | 60 | Ukraine | Dojlido & Best (1993) |
| | 166–202 | Brazil | Blumenthal et al. (1996) |
| | 6–380 | Mexico | Jiménez, Chávez & Capella (1997); Jiménez, Chávez & Hernández (1999) |
| | 840 | Marroco | Schwartzbrod et al. (1989) |
| | 1–800 | Mundial | Feachem et al. (1983) |

| Agent | Content | Country | Reference |
|---|---|---|---|
| Poliovirus (No./100ml) | 182 to 492 000 | England | Irving (1982) |
| Protozoa (cysts/l) | $10^3$–$10^5$ | Worldwide | Feachem et al. (1983) |
| Protozoan cysts, (organisms/l) | 978 to 1814[1] | Mexico | Jiménez, Chávez & Capella (1997); Jiménez, Chávez & Hernández (1999) |
| | 28.4[2] | USA | Rose (1988) |

MPN = most probable number
[a] Due to an estimated 42% of the population excreting, on average, 800 000 ova daily per person.

The survival of certain pathogens is shown in Table 3.7.

**Table 3.7 Typical survival of some pathogens at 20–30°C**

| Pathogens | Survival, days | | |
|---|---|---|---|
| | **Fresh and wastewater** | **Cultures** | **Soil** |
| ***Viruses***[b] | | | |
| Enteroviruses | < 120, usually <50 | < 60, usually < 15 | < 100, usually < 20 |
| ***Bacteria*** | | | |
| Thermotolerant coliforms | < 60, usually < 30 | < 30, usually < 15 | < 70, usually < 20 |
| *Salmonella* species[a] | < 60, usually < 30 | < 30, usually < 15 | < 70, usually < 20 |
| *Shigella* species[a] | < 30, usually < 10 | < 10, usually < 5 | < 20, usually < 10 |
| *Vibrio cholerae*[c] | < 30, usually < 10 | < 5, usually < 2 | < 20, usually < 10 |
| ***Protozoa*** | | | |
| *Entamoeba histolytica* cysts | < 30, usually < 15 | < 10, usually < 2 | < 20, usually < 10 |
| Helminths | | | |
| *A. lumbricoides* ova | Several months | < 60, usually < 30 | Several months |

[a] In sea water viruses have a much lower survival rate than bacteria.
[b] Includes polio, echo and coxsackie viruses.
[c] There is a lot of uncertainty about the survival of *Vibrio cholerae* in water.
Source: Feachem et al. (1983).

### 3.2.8 Microbiological analytical techniques

Setting microbiological quality guidelines for recycled water is complicated, because the techniques for identifying and measuring pathogens are complex, time consuming, costly and specialized. Furthermore, the quality and type of microorganisms found in domestic wastewater is so variable that standardizing methods for all purposes may not be useful, and routine monitoring for each organism is not only impractical but impossible. In particular, the time required to analyse a pathogen of interest is so long that measurement is not a useful tool for providing treatment plants with feedback.

*Viruses*

Identification and quantification of viruses in wastewater is complicated by the low level of recovery and by the complex and costly techniques involved. Therefore, very few laboratories can undertake such tests. A laboratory requires, on average, 14 days to determine the presence or absence of a virus in the water and a further 14 days to identify it. Techniques using recombinant

DNA may facilitate virus detection, but the questions of how to study the infectious potential of the particles and how to apply these advances to environmental samples have yet to be answered. The detection limit for viruses is about 0.01 viral units per litre (vu/l). Determination of viruses in recycled and reclaimed water is very difficult. Final concentrations of viruses after advanced treatment are very low (about 0.002–2.3 vu/l). Also, for reclaimed water, virus concentrations are so low that it is almost impossible to quantify them; it is only feasible to determine their presence or absence after 24 m of water infiltration in soil (Asano, 1998).

*Protozoa*

Quantifying protozoa in clean and recycled water also presents enormous difficulties because of the size of the sample that must be filtered (100–500 l) to retain the oocysts and cysts, and to identify the species and their viability.

### 3.2.6 Microbiological indicators

In the future, the use of molecular techniques such as the polymerase chain reaction (PCR) may make it easier to detect and identify pathogens, and thus to identify those that pose a health risk. However, good analytical techniques do not solve the problem of selecting one or several indicators and their significance for health. Microbiological indicators are needed because of the difficulty of measuring pathogens, their wide variety and chance presence in waste and treated water. Generally, an indicator should:

- be present only when there is contamination of a faecal origin;
- have the same or a greater capacity to survive than pathogens for which it is acting as an indicator;
- not reproduce outside the host;
- be easily determined and monitored in environment samples.

There is currently no indicator that has all of these characteristics; however, the most widely accepted indicator organisms, in order of acceptance, are coliforms, faecal streptococci and *Clostridium perfringens*. All of these organisms are used as indicators for drinking-water (supposedly from primary sources) and wastewater, but are currently not accepted when dealing with reclaimed water. Experience on the usefulness of such organisms as indictors is generally limited to drinking-water and wastewater; it is very rare in the case of reclaimed water. There is no consensus about which indicators should be used in the case of water reuse.

The fact that the disease causing agent for waterborne diseases is often not detected makes an adequate selection of indicators difficult. This implies that, independently of the selection of indicators, the treatment processes involved should effectively reduce the amount and type of pathogens originally present in wastewater and a way of ensuring this occurs should be found.

In the following section, we first consider the properties of traditional indicators and then present other options that are currently the subject of debate.

*Traditional indicators*

Conventional indicators are the coliforms (thermotolerant and total) and the faecal streptococci.

**Coliforms as indicators**

Historically, the thermotolerant coliform group has been used to indicate the presence of bacteria of a faecal origin and consequently the possible presence of other pathogenic microbes. Initially, total coliforms were used, but when it was discovered that they were not all of faecal origin, the thermotolerant coliforms were used, detected using incubation at $44.5 \pm 0.5^{o}C$. However, thermotolerant coliforms found in faeces include organisms such as *Klebsiella*, whose significance to health is questionable.

Due to tradition more than anything else, total coliforms have been the indicators generally used to determine whether water is drinkable. However, in the case of recycled water, the United States Environmental Protection Agency (US EPA) recommends using thermotolerant coliforms as an indicator or, if their representation is in doubt, reverting to the specific determination of *E. coli*. This is because, due to the water's origin, there is certainty about whether there are or were thermotolerant coliforms.

A problem with the use of coliforms is that viruses, protozoa and helminth ova are more resistant to disinfection and environmental conditions than thermotolerant coliforms. Also, in effluents not containing chlorine, such as those that may be used in recharge, there is regrowth, so that analysing thermotolerant coliforms in reused water is of little relevance to health. Furthermore, regrowth could be interpreted as the presence of pathogens, even when they are unable to reproduce in recycled water.

**Faecal streptococci as indicators**

Faecal streptococci are intestinal bacteria that belong to the Lansfield's group (corresponding to a strain classification) and are found in the faeces of all warm-blooded mammals. Within the faecal streptococci is a subgroup known as enterococci, which is characterized by growing both at 10°C and 45°C, in an environment of 6.5% NaCl and a pH of 9.6. This subgroup has been suggested as a useful indicator for the quality of water for recreational use (Cabelli, 1983; Dufour, 1984), and because of its environmental permanence, it may be useful to water reuse.

*Nonconventional indicators*

In countries where waterborne illnesses pose a significant problem, selecting an indicator can be more complex. Hazen & Toranzos (1990) showed that *E. coli*, the most universal indicator, is a native of tropical waters. Mexico City aquifer supplies water to millions of people. In this aquifer, minor recharge work using reclaimed water is being carried out, and in the southern part direct infiltration of wastewater or septic tank effluents is been performed, due to the lack of sewer systems and the presence of a stony soil. Mazari-Hiriart et al (1999) have carried out a study in this area to determine the usefulness of coliforms and faecal streptococci as indicators. In a one-year survey, they found that the presence of male specific MS-2 coliphages cannot be correlated to the presence of traditional indicators (total and faecal coliforms as well as faecal streptococci). These results suggest that, for reclaimed water, bacterial indicators must be used with a viral indicator. Considering that bacteriophages have a very high prevalence (Snowdown & Cliver 1989: Yahya et al., 1993), Mazari-Hiriart et al (1999) concluded that bacteriophages are the best indicator of faecal pollution. The coliphages can be detected at relatively low cost, using simple analytical methods.

***H. pylori* as an indicator**

The bacterium *H. pylori* (discussed in Section 3.2.2, above) has been found in water supply sources contaminated by wastewater infiltration. As an indicator for use with recycled water it may be of interest in developing countries, given the markedly high prevalence in these countries. Mazari-Hiriart et al. (2001) demonstrated that complying with a set level of thermotolerant coliforms and wastewater chlorine (0.2–1 mg/l) does not necessarily indicate the absence of *H. pylori*.

This organism could also have applications as an indicator in developed countries, because *H. pylori* is present throughout the world. In Sweden, for example, it has been detected in wells, wastewater and water supplies, despite the high level of water treatment in that country (Hulten et al., 1998).

***Clostridium perfringens* as an indicator**

*Clostridium perfringens* is commonly found in faeces and has been used as a recent indicator of faecal contamination, particularly in the United Kingdom (Bisson & Cabelli, 1980). Its usefulness lies in the fact that it is easily quantified and is more resistant to disinfection and environmental conditions than many pathogens. *C. perfringens* form a resistant endospore; therefore, the presence of vegetative cells indicates recent contamination, whereas spores imply past contamination. Grabow (1990) questions the usefulness of this microorganism in water recycling and reuse because, although it is resistant to disinfection, the initial level of contamination is usually low, making detection difficult. Only modern techniques can overcome this problem. What is clear is that there is little information available on the levels of *C. perfringens* in reused and wastewater.

*Viruses as indicators*

A wide variety of viruses may be present in waste and recycled water, and, as with bacteria, it is impossible to measure all of them. Bacterial indicators are not useful for determining the presence of viruses; therefore, a virus indicator would be useful in the case of water reuse. However, any virus of animal origin does not meet at least two of the set requirements for an ideal indicator:

- the viruses of interest are those that infect human beings; thus, they are rarely found and occur only in low concentrations;
- viruses cannot be detected even though other pathogens are present in the water.

Bacteriophages, viruses that infect bacteria, have potential as indicators. Although bacteriophages have not been linked to human diseases, and therefore have no health implications, they are easily detected. In particular, the coliphages (bacteriophages that infect coliform bacteria) have been proposed as viral indicators. The coliphages are always present in wastewater and are fairly abundant; they are relatively cheap and easy to detect, and information on levels of contamination can be obtained within 24 hours. However, the coliphages do not adequately simulate the behaviour of animal viruses.

Another possible candidate as indicators of human enteroviruses are the specific coliphages $F^+$ . They are found in wastewater at levels of 100–1000 per ml and have similar or better resistance to environmental factors and disinfection than human enteroviruses.

*Protozoan cysts and helminth ova as indicators*

The pathogens most resistant to disinfection, and which survive the widest range of environmental conditions are the protozoan cysts and helminth ova. Therefore, the absence of bacteria or enteroviruses does not indicate that protozoa are absent. Protozoan cysts would pose the biggest problem in reused water because of their small size and their resistance. For example, disinfection using ultraviolet (UV)[4] requires 60 mW s/cm$^2$ to inactivate *Cryptosporidium* and 176 mW s/cm$^2$ for *Acanthamoeba* (Maya, Beltrán & Jiminéz, 2002). Currently, there is no protozoan that is considered an ideal indicator, and different protozoa may present a problem in different countries. The main problem associated with protozoa is that detection techniques to determine the presence and viability of these microorganisms are very complex.

Regarding helminths, *Ascaris* ova are used because of their resistance. However, in this case it does not make sense to talk about an indicator organism; rather, the helminths themselves are measured, because basically this requires the same, effort, cost and time.

[4] UV is a disinfection option often preferred for drinking water reuse treatment due to the lack of byproducts formation

### 3.2.7 Risk assessment

The National Research Council (1982) of the USA states that to define the risk involved in the use of recycled water for consumption, the following should be considered:

- the long-term effects of chemical compounds are the main concern;
- the risk from consuming recycled water should be evaluated in comparison to the risk of consuming water from conventional resources;
- the need for an intensive toxic tests program.

These points may be interpreted differently by developed and developing countries. For example, in developing countries, although there are some concerns about long-term effects, microbiological pathogens have effects in the very short term, so many of their inhabitants do not have the opportunity to wait several years to see what other problems may arise in the long term. Also, the second of the above points may be interpreted as an analysis of local conditions because primary sources in many developing countries are so polluted (in particular, with domestic water) that the use of recycled water could result in better quality water. Table 3.8 shows the risk from exposure to different types of reused water.

To establish the risk in a particular region it is necessary to:

- establish the type and quantity of microorganisms present
- know the infectious dose
- define and evaluate a possible route of infection.

These three aspects are not universal or easy to define. For example, the concentrations of enteric viruses found in undisinfected secondary effluent from municipal wastewater treatment plants are highly variable. Thus, in assessing the risk of infection from viruses, the variability in their occurrence must be considered (Tanaka et al., 1998).

**Table 3.8 Annual risks of contracting at least one infection from exposure to recycled wastewater at two different enteric viruses concentrations**

| Viruses | Exposure scenarios | | | |
|---|---|---|---|---|
| | Landscape irrigation for gulf courses | Spray irrigation for food crops | Unrestricted recreational impoundments | Groundwater recharge |
| Maximum enteric viruses concentration of 1.1 vu/l in chlorinated tertiary effluent | | | | |
| Echoviruses12 | $1x10^{-3}$ | $4x10^{-6}$ | $7x10^{-2}$ | $6x10^{-8}$ |
| Poliovirus 1 | $3x10^{-5}$ | $2x10^{-7}$ | $3x10^{-3}$ | $5x10^{-9}$ |
| Polioviruses 3 | | $1x10^{-4}$ | $8x10^{-1}$ | $2x10^{-8}$ |
| Minimum enteric viruses concentration of 0.01 vu/l in chlorinated tertiary effluent[a] | | | | |
| Echoviruses 12 | $9x10^{-6}$ | $4x10^{-8}$ | $7x10^{-4}$ | $5x10^{-10}$ |
| Poliovirus 1 | $3x10^{-7}$ | $1x10^{-9}$ | $2x10^{-5}$ | $5x10^{-11}$ |
| Polioviruses 3 | $2x10^{-4}$ | $1x10^{-6}$ | $2x10^{-2}$ | $2x10^{-10}$ |

Source: Asano et al. (1992)

### 3.2.8 Establishing quality guidelines based on microbiological aspects

The number and type of pathogens varies in terms of space and time depending on the population's health, the disease incidence rate, the time of year, water use, the economic level and water quality.

Likewise, the effects of pathogens are also very variable, ranging from mild gastrointestinal illnesses to more serious infections such as hepatitis, cholera and meningitis.

Each region should therefore determine the pathogens of interest, based on population waterborne disease data (see “Stockholm framework” discussion in Appendix A) This is not an easy task, because in several countries, especially developing ones, identifying the agent of disease is not a common practice. It is often impossible to identify the disease causing agent in many gastrointestinal illnesses supposedly caused by water, either because of a lack of analytical methods or because the opportunity does not present itself. For example, even in the USA, one of the richest countries in the world, the agent is detected in only 50% of cases waterborne disease.

For 589,000 cases of waterborne disease documented between 1970 and 1990 in the USA, C*ryptosporidium* was the most common (74%), whereas for amoebiasis and salmonellosis, which are common in developing countries, frequencies were only < 0.1% and 0.5 % (Sayre, 1988). In contrast, levels of amebiasis and salmonellosis in developing countries are 15% and 4 %, respectively.

## 3.3 Chemical and physical contaminants

### 3.3.1 Organic matter

This section looks at the main types of organic compound that contaminate water, and analytical techniques for their determination.

*Biodegradable organic matter*

Biodegradable organic matter provides food for microorganisms, adversely affects disinfection processes, makes water unsuitable for certain industrial and other uses, consumes oxygen and may cause acute or chronic health effects if recycled water is used for potable purposes. Regulatory agencies typically use biochemical oxygen demand (BOD) as a gross measure of biodegradable organic constituents.

*Refractory compounds*

Refractory compounds are characterized by their resistance to normal treatment techniques. Some are harmful to humans and the environment, so their presence in recycled water for human consumption should be reduced. The most commonly used parameter to evaluate them is total organic carbon (TOC), although this also includes biodegradable matter.

*Nutrients*

Nitrogen, phosphorus and potassium are essential nutrients for plant growth, and their presence in water increases its value for irrigation. In the case of nitrogen, excess amounts in the aquifer limits its use as a supply source because of critical values established for the control of acute toxicity in infants (Sayre, 1988).

*Heavy metals*

Certain heavy metals accumulate in the environment and are harmful to plants and animals. Their presence limits water recycling and reuse. Usually, in terms of recharge, if the water treated is domestic wastewater there are no problems unless drainage contains a high content of industrial discharges.

*Residual chlorine*

Residual chlorine plays a role in the possible formation of chlorinated organic compounds that limit the subsequent use of water because of its potential to cause cancer. It is also toxic for native soil flora and fauna, which contribute to cleaning the water during percolation. It can be difficult to balance the benefits obtained from the application of chlorine with its secondary effects.

### 3.3.2 Compounds of interest or concern in recycled water

Table 3.9 summarizes the types of compound that are of interest or concern in recycled water. The difficulty here is that recycled water will contain very low concentrations of a variety of compounds. The detrimental health effects of these compounds are known, but their presence in water may not be known. Therefore, the maximum TOC value is usually limited to 1–2 mg/l, irrespective of the type of compounds that generate the TOC.

According to Raymond and Rauckamn (1987), on a day-to-day basis, humans are rarely exposed to a single chemical, as occurs during toxicological studies used to determine permitted levels, except perhaps in cases of occupational exposure. In reality, it is imperative to consider all possible sources of ingestion (e.g. food or beverages containing chemical additives, personal care products and the air, especially in polluted cities). This mixture of compounds, together with the varied effects caused by synergetic, neutralizing, potentiating and antagonistic effects between substances that enter the body, makes it difficult to predict the effects on humans.

**Table 3.9 Compounds of possible interest or concern in water reuse**

| |
|---|
| Metals, inorganic components, pharmaceutical |
| Estrogens (natural and synthetic) |
| Surfactants and solvents |
| Perfumes |
| In particular: nonylphenol (endocrine disruptor, a product of the biodegradation of the nonylphenol ethoxylate surfactant), 17-β estradiol (natural human estrogen). Polychlorinated dibenzo-p-dioxyne, Polychlorinated dibenzofurans (which are pentachlorophenol subproducts). |

Source: Van Eyck et al. (2001)

### 3.3.3 Analytical techniques

Significant advances have been made in analytical techniques, such as mass spectrophotometry coupled with gas chromatography, spectrometry, nuclear magnetic resonance and hydrofibric, hydrophilic and volatile separation methods. However, the detection and analysis of a large number of compounds in water has perhaps given rise to more information than can actually be interpreted, because we now know more about what, and how much, there is in the water than about the effects that such compounds produce, especially in conditions outside the laboratory. Usually, the TOC is used to establish the organic quality of water as well as the efficiency of a process. However, this parameter does not define the degree of danger of the water, because the same TOC values can be obtained from samples with differing degrees of toxicity (and even from samples that are nontoxic).

An added problem in evaluating the toxicity of recycled water is a lack of experience in compiling, collecting, concentrating and processing samples. Traditional detection methods often eliminate compounds present throughout the process, and samples need to be concentrated about 500 times. Further research is needed into chemical analysis of recycled water.

## 3.4 Treatment of wastewater for aquifer recharge

This section discusses the treatment processes applicable before aquifer recharge (unintentional and intentional). It includes simple processes aimed at improving existing conditions; for example, it describes the advantages of soil aquifer treatment, a useful technique in unintentional recharge. This section also describes high technology processes, useful in developed countries to encourage and promote the reuse of water, such as direct injection to the aquifer in conjunction with advanced treatment schemes.

### 3.4.1 General considerations

Typical water treatment schemes in developed countries include at least two steps. The primary level is usually based on physical properties, and serves to remove decantable and floating material. The secondary level is generally a biological process, used to remove most of the remaining (dissolved and suspended) organic material. Because of the negative effects of returning secondary effluents to the environment, a third step is often used. This tertiary treatment refines the quality of the water resulting from the secondary treatment, but the techniques involved in this step will depend on the specific problems of a country or region. Generally speaking, the third step is used to remove nitrogen and phosphorus; however, the development of “compact” processes, which eliminate these contaminants in earlier stages, is increasingly raising doubts about the idea of treatment schemes by steps.

In the developing world, the level of treatment barely reaches as much as 15% of municipal wastewater CEPIS/OPS, 2000; US EPA, 1992b). In most cases, treatment consists of a primary step only, or a partial secondary level. Effluents are frequently used for agricultural irrigation or are discharged into soil, rivers or lakes, eventually reaching the ocean. Reuse of water for irrigation is promoted by farmers because of the increase in productivity generated by raw wastewater (Table 3.10). Furthermore, irrigation efficiency in developing countries is frequently low (40–50%), so that much of the unused water infiltrates into the aquifer and recharges it. One of the main concerns of such incidental recharge is the microbiological quality of the underground water.

**Table 3.10 Yield increase where wastewater is used for irrigation in the Mezquital valley, near Mexico City**

| Crop | Yield in tons/ha | | Increase (%) |
|---|---|---|---|
| | Raw wastewater | Fresh water | |
| Maize corn | 5.0 | 2.0 | 150 |
| Barley | 4.0 | 2.0 | 100 |
| Tomato | 35.0 | 18.0 | 94 |
| Oats for forage | 22.0 | 12.0 | 83 |
| Alfalfa | 120.0 | 70.0 | 71 |
| Chilli | 12.0 | 7.0 | 70 |
| Wheat | 3.0 | 1.8 | 67 |

Source: Jiménez, Chávez & Capella (1997).

### 3.4.2 Pathogen removal

Environmental conditions and treatment processes are both hostile for most pathogens, and cause their death (Kadlec and Knight, 1996). The factors involved include temperature, ultraviolet light, water quality, competition with native microorganisms and sedimentation. In Section 3.2, the importance of controlling pathogens in recycled water for human consumption was mentioned. Published studies suggest that traditional secondary and tertiary treatment processes reduce pathogen concentrations to “acceptable” levels. This “popular” knowledge has established a minimum target level, based on disinfection, without taking into account what occurs with lower treatment levels or different process designs (Lorch, 1987).

The literature describes the efficiency of pathogen removal in different treatment steps (Table 3.11). However, these data are mainly from developed countries, where the limited variety and low concentration of microorganisms are unlikely to challenge the capacity of the processes. Developed countries have the technical and economic resources to reduce the type and concentration of pathogens found in wastewater to levels considered safe. However, in the developing world, with its wide variety and high concentrations of microorganisms, the situation is more complex.

**Table 3.11 Pathogen removal by different treatment processes**

| Stage | Enteroviru s | *Salmonella* | *Giardia* | *Cryptosporidium* | Helminths |
|---|---|---|---|---|---|
| Concentration in wastewater | $10^5$–$10^6$ | $10^3$–$10^4$ | $10^3$–$10^5$ | 1–4000 | --- |
| Remaining after primary treatment[a] | $10^3$–$10^5$ | $10^2$–$10^4$ | $10^4$–$10^5$ | | |
| Efficiency of removal | 50–98% | 50–99.8 | 27–64 | 0.7 | 90% |
| After secondary treatment[b] | $10^2$–$10^5$ | $10^0$–$10^3$ | $10^3$–$10^5$ | | |
| Efficiency of removal | 53% to 1 log | 98% to 2 log | 45–97% | | 99.99% |
| After tertiary treatment[c] | $10^{-3}$–$10^2$ | $10^{-6}$–$10^0$ | $10^{-2}$ | $10^3$ | |
| Efficiency of removal | 1–3 log | 2–6 log | 1–4 log | 2–7[d] | |

[a] Primary sedimentation and disinfection
[b] Primary sedimentation, trickling filter/activated sludge, and disinfection
[c] Primary sedimentation, trickling filter/activated sludge, disinfection, coagulation, filtration and disinfection
[d] Filtration only
Sources: Feachem et al. (1983); Leong (1983); US EPA (1991); Yates & Gerba (1998)

### 3.4.3 Primary and advanced primary treatments

For effluents to be used in irrigation or recharge by infiltration, there is not need to remove organic material; therefore, an expensive and highly technical secondary treatment may not be necessary. The same may be true of a tertiary treatment to remove nitrogen and phosphorus, because this process would remove fertilizers that are useful for the soil. In view of these considerations, a simple process at a primary level may suffice. For example, the combination of a high-carbon content effluent and soil aquifer treatment increases the denitrification of nitrates in the soil and helps to degrade refractory synthetic organic compounds through co-metabolism and secondary use of carbon (Bouwer, 1974; McCarty, Rittman & Reinhard, 1985). In this way, the risk of nitrogen dissolving into the underground water is reduced, and agriculture productivity is increased. Primary and advanced primary treatments are useful in countries with limited economical resources where wastewater is already being used for irrigation. This approach is more effective than costly treatment processes that are difficult to implement and whose main purpose is the protection of surface bodies of water. If programs incorporating soil aquifer treatment are properly implemented, they have the advantage of increasing the availability of aquifer water.

Authors such as Adin and Sacks (1991), Shao et al. (1993), Jiménez, Chávez & Capella (1997) and Harleman & Murcot (1999), recommend application of a primary treatment assisted with chemicals — referred to as advanced primary treatment (APT) or chemical enhanced treatment — for reuse of water for irrigation. This treatment removes harmful contaminants, but allows beneficial soluble organic matter to pass into the soil. The process results in a low content of suspended solids or turbidity, which leads to greater disinfection efficiency, either with chlorine or UV light. Also, it means that the water is suitable for use in sprinkling irrigation. The effluent quality is improved by the soil effect, and aquifers can be used as water supply storage.

According to Leong (1983), primary processes involving grit elimination and simple sedimentation remove 5–10% of viruses; however, in practice, values range from 0 to 80%, with a median of 10%. Efficiency is highly dependent on the degree of separation of solids, because viruses adsorb onto particles, where their infectious capacity is not diminished. There are no data available for virus removal by APT, although it can be assumed to be more effective than a simple primary treatment because APT removes 70–80% of suspended solids, compared to 30% for simple primary treatment. Viruses are generally associated with particles of less than 8 µm. Gerba et al. (1975). Between 60 and 100% of viruses are adsorbed on particles suspended in wastewater (Wellings et al., 1976), thus removal of colloidal particles is highly recommended to assist with virus removal.

### 3.4.4 Secondary processes

*Activated sludge*

Activated sludge is more effective than other secondary biological processes for pathogen removal (Table 3.12). For example, it removes 10% more pathogens than trickling filters (Leong, 1983). Both sedimentation and aeration play an important role in this process.

Sedimentation eliminates heavy and large pathogens, while aeration promotes antagonistic reactions between different microorganisms, causing their elimination. As a result of pathogens becoming entrapped in the flocs (which are subsequently sedimented), there is a high removal of small nonsedimentable microorganisms, such as *Giardia* and *Cryptosporidia*, which remain concentrated within the sludge.

Helminths are also eliminated from activated sludge by sedimentation. In the USA, helminths are generally at concentrations too low to be detected following this treatment; while in countries such as Mexico, where the initial concentrations of helminths are higher, values of 3–10 helminth ova per litre are found in the effluent of properly operated treatment plants (Jiménez, Chávez & Capella,1997; Jiménez & Chávez, 1999).

**Table 3.12 Pathogen removal in activated sludge**

| Pathogen | Removal | Reference |
|---|---|---|
| Viruses | 90–99% | Leong (1983); Rao, Metcalf & Melnick (1986) |
| *Giardia* and *Cryptosporidia* | 90% | Casson et al. (1990); Rose & Carnahan (1992) |
| Helminths | 90% | Jiménez, Chávez & Capella (1997); Jiménez &Chávez (1999) |

Irving (1982) found that, although overall virus removal by activated sludge is 60%, each virus type is removed differently. For example, the percentage of organisms removed by the treatment is 92% for enterovirus, 81.5% for adenovirus and 26% for rheovirus. Rotavirus, which is a significant health hazard (see Section XX, above), behaves like rheovirus; therefore, activated sludge is not efficient at removing this pathogen.

### *Stabilization ponds*

Stabilization ponds are very efficient at removing almost all kinds of pathogens (see tables 3.13 and 3.14). The factors involved in the pathogen inactivation or removal include temperature, sunlight, pH, predator microorganisms, adsorption and trapping into flocs. However, the most important factor is sedimentation, because of the lengthy retention time. Shuval et al (1985) found that helminth ova were completely removed after 10 or 20 days hydraulic retention in stabilization ponds. Bacteria and viruses were reduced by 5–6 log with 20 days retention time, and by 2 log with 10 days.

**Table 3.13 Comparison of pathogen removal in stabilization ponds and conventional treatments**

| Excreted pathogens | Removal in | |
|---|---|---|
| | stabilization ponds | conventional treatment |
| Bacteria | Up to 6 log units | 1–2 log units |
| Viruses | Up to 4 log units | 1–2 log units |
| Protozoa cysts | 100% | 90–99% |
| Helminth ova | 100% | 90–99% |

Source: Feachem et al. (1983)

For these reasons, in developing countries with warm climates, use of stabilization ponds to recycle water for agricultural use is promoted, when land is available at a reasonable price (WHO, 1989). To remove helminth ova, a minimum retention time of 8–10 days is set, with at least twice as much time to reduce thermotolerant coliforms to less than 1000/100 ml. To control *Cryptosporidia*, almost 38 days are needed (Grimason et al., 1993); however, practical experience demonstrates that this is hard to achieve when there are hydraulic problems, such as flow bypasses (Camp, Dresser & McKee, 1993; Huntington & Crook, 1993; Yates & Gerba, 1998). In some cases, the use of stabilization ponds is limited if the land is geologically unsuitable for building a pond or no reasonably priced land is available (e.g. where land costs are high due to the production of export crops, or the distortion in the market created in areas where narcotics crops are cultivated). Additionally, care must be taken in arid zones with high evaporation–transpiration rates, because in these areas ponds may represent a net loss of water. For example, in the eastern part of Mexico City, a 920 hectare pond built for agricultural reuse of water has an evaporation rate of 25% of incoming water (700 l/s) in the dry season (when water is needed for irrigation); additionally, salinity increases as a result of evaporation.

**Table 3.14 Bacterial and viral content in raw wastewater and the effluent of five waste stabilization ponds in series in North-east Brazil at 26°C (geometric mean)**

| Organism[a] | RW | A | F | M1 | M2 | M3 |
|---|---|---|---|---|---|---|
| Thermotolerant coliform | $2 \times 10^7$ | $4 \times 10^6$ | $8 \times 10^5$ | $2 \times 10^5$ | $3 \times 10^4$ | $7 \times 10^3$ |
| *Campylobacter* | 70 | 20 | 0.2 | 0 | 0 | 0 |
| *Salmonella* | 20 | 8 | 0.1 | 0.002 | 0.01 | 0 |
| Enterovirus | $1 \times 10^4$ | $6 \times 10^3$ | $1 \times 10^3$ | 50 | 50 | 9 |
| Rotaviruses | 800 | 200 | 70 | 10 | 10 | 3 |

A = anaerobic pond with a mean hydraulic retention time of 1 day; F = a facultative pond with a retention time of 5 days; M1–M3 = maturation ponds with a retention time of 5 days; RW = raw wastewater
[a] Bacterial numbers per 100 l, viral numbers per 10 l
Source: Pearson et al. (1995)

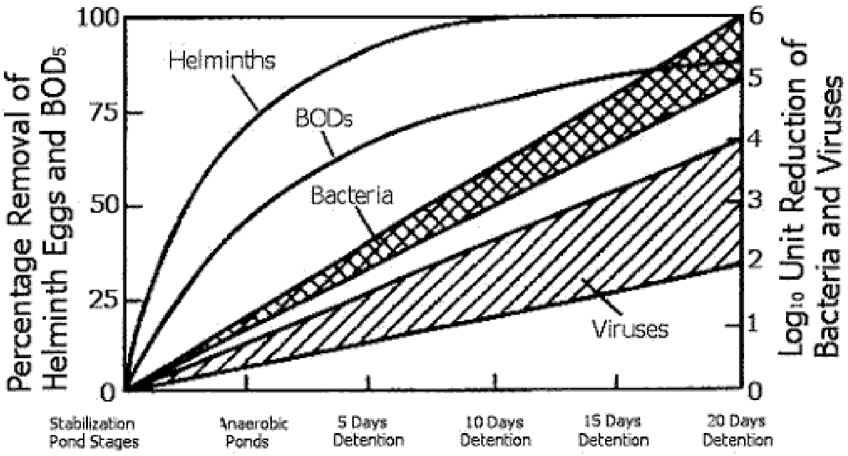


**Figure 3.2. Generalized removal curves for BOD, helminth ova, excreted bacteria and viruses in waste stabilization ponds at temperatures above 20 °C (Shuval *et al.*, 1985).**

Because of their size (20–90 µm), helminth ova are closely related to the suspended solid content (Fig. 3.2); therefore, processes for eliminating suspended solids also remove helminth ova. It is estimated that with levels of 20–40 mg/l of total suspended solids (TSS) in the effluent, the concentration of helminth ova may reach a level of 3–10 and, for values below 20 mg/l, the content may easily drop to 1 or less (Jiménez &Chávez, 1998).

*Slow filtration*

Slow filtration is recognized as an efficient method for microbiological control in producing potable water for rural and low-income communities. However, there is little information available about its use for purifying wastewater effluents (with high organic and turbidity content), and, in particular, about its benefits in preparing water for recharging or irrigation. It may be adviseable to use slow filters after wastewater is stored in reservoirs and before it is used for irrigation, because of their proven capacity to remove pathogens (Adin, 1998). Filtration has also been recommended in combination with soil aquifer treatment and aquifer storage. The few studies carried out on slow filtration of wastewater have demonstrated a removal range of 60–80% of suspended solids and one *E. coli* log, with coarse sand (Farook & Yousef, 1993; Farooq et al., 1993; Adin, Mingelgrin & Kanarek, 1995).

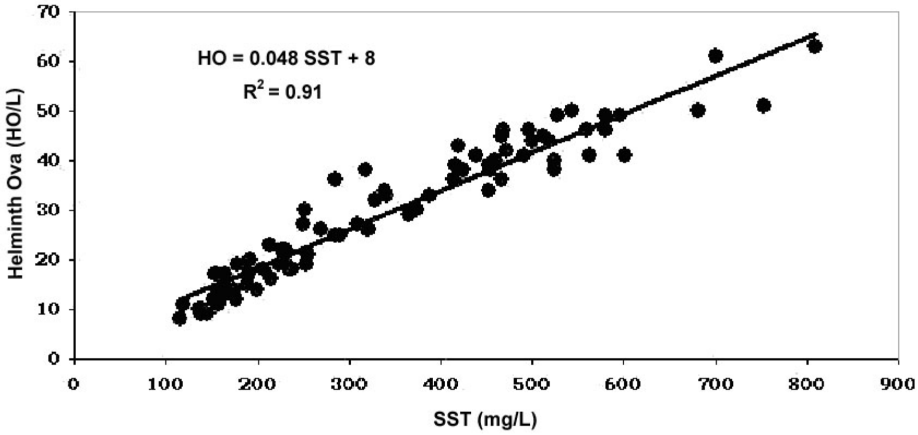


**Figure 3.3. Correlation between TSS concentration and Helminths ova in wastewater from Mexico City (Adapted from: Jiménez and Chávez, 1999; Jiménez *et al.*, 1999).**

In a study by Ellis (1987), 0.95 m-high filters were applied to a secondary biological effluent that had low BOD and TSS (16–22 mg/l), and $10^5$ thermotolerant coliforms. Using filtration rates of 3.5–7.0 m per day, the filters removed 65% of BOD and TSS, and 95% of thermotolerant coliforms, in filter runs of up to 20 days. A horizontal filter with a rate of 2 m per hour and a contact time of 33 minutes gave an efficiency of 82% in TSS. Interestingly, this research demonstrated that using two distinct sizes of sand (0.3 and 0.6 mm) makes no difference to the outcome, and that it is possible to obtain an adequate effluent in both cases.

*Constructed wetlands*

Interest is growing in the use of constructed wetlands as a secondary or tertiary treatment method, because this method does not generate by-products (Table 3.15). Constructed wetlands, which imitate nature, have a broad and diverse biological activity. In the USA and Canada, the most popular systems use emerging vegetation or free water surface system (Haberl, 1997; Cole, 1998). The free-water surface systems are highly diverse in shape and location, and are used for refining effluents from treatment systems in small or medium communities, although they have also been used for treating industrial wastewater.

Wetlands generally consist of reservoirs or ponds where plants are grown. They are built on a slant so that water may flow by gravity, and they are generally shallow to allow for better removal of contaminants. The plants typically used are:

- large plants with floating or aerial leaves;
- plants with well-developed and submerged roots, such as rushes, water hyacinth, reeds and water lilies; and
- very small floating plants with few roots or no roots at all, such as those of the *Lamenacea* family, *Lemna* or duckweed, *Spirodela*, *Wolffia*, *Wolffiela*, and *Salvinia* (Rico et al., 1992; Brix, 1993; Olguin et al., 1994; Metcalf & Eddy, 1996).

Free water surface systems are very efficient at removing nitrogen, phosphorus and heavy metals. For example, water lilies eliminate up to 350 kg of phosphorus and 200 kg of nitrogen per year (Brix, 1997). The main limitation of wetlands is the large area required, and the generation of mosquitoes and unpleasant odours when they are not operated correctly (Olguin & Hernández, 1998). The efficiency of coliform removal is very high in wetlands; however, there are wide variations depending on climate, season, wetland type and retention time. Thus, it is still difficult to control the stability of the process, despite the fact that its applicability to, and convenience for, developing countries are recognized. The problem of control of stability limits the use of water surface systems in water reuse schemes. Several wetlands have been installed in different countries, but few microbiological studies have been carried out because of the high cost involved. Some data about pathogen removal in wetlands are shown in Table 3.15; variations in percentages are due to the type of plant used and the different climates.

**Table 3.15 Pathogen removal in wetlands**

| **Microorganism** | **Removal** | **Conditions** | **Reference** |
|---|---|---|---|
| Thermotolerant coliform | 98 | Retention time of 4 days in surface flow wetland. Highest efficiencies observed when using duckweed. | Karpiscak et al. (1996) |
| | 92–99% | Removal rates from January to October using reeds. | Rivera et al. (1995, 1997) |
| | 90–98% | Using reeds | Haberl & Perfler (1991); Hiley (1991) |
| MS2 coliphages | 67–84% | Retention time of 4 days in surface flow wetland. Highest efficiencies observed when using duckweed. | Gersburgb, Gerhart & Ives (1989) |
| *Cryptosporidium* | 53–87% | Retention time of 4 days in surface flow wetland. Highest efficiencies observed when using duckweed. | Karpiscak et al. (1996) |
| *Giardia* | 58–98% | Retention time of 4 days in surface flow wetland. Highest efficiencies observed when using duckweed | Karpiscak et al. (1996) |
| *Acanthamoeba astronyxis*, *A. polyphaga*, *A. rhysodes*[a] | 60–100% | Retention time of 5 days in surface flow wetland. Using reeds. | Ramirez et al. (1993) |
| *E. coli, E. histolytica, E. nana, Iodamoeba bütschlii* | 100% | Combined system with gravel and reeds. | Rivera et al. (1995) |
| *Ascaris lumbricoides* ova | 100% | Combined system with gravel and reeds. | Rivera et al. (1995) |

[a] Tested positive for pathogenesis in mice.

### 3.4.5 Tertiary processes

*Coagulation–flocculation*

Several studies have shown that coagulation–flocculation is a very good way to remove enteric viruses and phages. In fact, in terms of the number of viruses remaining in treated effluents, coagulation–flocculation is considered the best option after chlorination. Iron salts eliminate 99.5% of pathogens, lime 98.8% and aluminium salts 95%. However, whether the difference between coagulants is statistically significant has yet to be established. For viruses, the removal mechanism appears to involve a cationic interaction with the protein cover, which forms a complex with the coagulants and destabilizes the virus. Coagulation, especially with lime at 300–500 mg/l, promotes more effective elimination of pathogens, particularly viruses. To remove 98.9%, a pH of at least 11 must be reached and maintained for one hour (Leong, 1983). Removal is achieved by neutralization of the protein cover.

*Rapid filtration*

Filtration is one of the most useful treatments for the removal of protozoa and helminths, either when combined with a primary treatment (Landa, Capella & Jiménez, 1997), or as a tertiary step. Rapid filtration removes 90% of indicator bacteria, pathogenic bacteria (*Salmonella* and *Pseudomonas aeruginosa)*, protozoan cysts (*Giardia* and *Entamoeba*), and enterovirus. This removal can be increased to over 99% with the addition of coagulants (US EPA, 1992b; Jiménez et al., 2001). The median value in tertiary treatment is 73%, but the range is very broad (0–99%) depending on the design and operation criteria, such as filtration rate, size of media and type of chemical pretreatment. Of these, the filtration rate (< 2.4 m/h), and the addition of coagulants are the most important (Leong, 1983).

Because viruses are not substantially eliminated in the absence of chemicals, both coagulants and filters should be used in water reclaiming projects. Removal is carried out by destabilization and chemical adhesion. Coagulation–flocculation, sedimentation and filtration together achieve a virus removal of 2 log. A pulsator (a sedimentation unit with a sludge blanket) removes from 3 to 4 log using ferric chloride.

Table 3.16 shows pathogen removal using a physicochemical process on wastewater with high pathogen content. The level achieved is a result of the intermediate filtration stage. However, in the developing world it is not common to include filtration in treatment systems, despite its economic advantages (Table 3.17). Combining filtration with a biological (aerobic or anaerobic) or physicochemical process is very helpful in improving the microbiological quality of water and, consequently, the health level of a region.

**Table 3.16 Pathogen removal in physicochemical processes**

| Microorganism | Raw wastewater | APT effluent | APT and filtration | Chlorination |
|---|---|---|---|---|
| Helminth ova (ova/l) | 30–120 | 1–4 | 0.2 | 0.2 |
| Thermotolerant coliforms (MPN/100 ml) | $10^7$–$10^9$ | $10^6$–$10^8$ | $10^6$–$10^8$ | $10^3$ |
| *Salmonella* (MPN/100 ml) | $10^6$ | $10^5$ | $10^4$ | ND–$10^3$ |
| Protozoan cysts[a] (N/l) | 1007–1814 | 400–524 | 190–524 | 1–30 |
| *Pseudomona aeruginosa* | $10^4$–$10^6$ | $10^3$–$10^5$ | $10^3$–$10^4$ | ND–2 x $10^2$ |

APT = advanced primary treatment; ND = not detected
[a] *Giardia* and *Entamoeba coli*, *E. histolytica* cysts
Source: Adapted from Jiménez et al. (2001)

Mean removal is 95% when the process is used for virus removal, but efficiency decreases substantially if the process is used for controlling organic matter, flavour or odour (Gerba et al., 1975). It is not advisable to fit up-flow activated carbon filtration for tertiary virus removal because the presence of fine material in the effluent prevents proper disinfection.

*Activated carbon*

Contrary to expectations, activated carbon does not efficiently remove viruses because the binding sites are occupied by organic material preferentially.

*Membrane processes*

Membrane processes, such as reverse osmosis and ultrafiltration, are considered useful in removing pathogens with an efficiency of 3–5 log. Leong (1983) pointed out that membrane processes could not, however, replace disinfection, even when the membrane pores are smaller than the pathogens, because of leakage in the connections and joints, or because of the action of the surface properties of the organisms.

Sorber, Malina & Sagik (1972) describe a faulty efficiency in virus removal. These authors were only able to achieve a decrease of 5–6 log under good operating conditions. Advances in this field are needed to achieve effluents of uniform high quality.

**Table 3.17 Comparative treatment costs for a 35 $m^3$/s wastewater treatment plant in Mexico[a]**

| Treatment process | Helminth ova in raw wastewater | TSS removal (%) | COD removal (%) | Cost (US$/$m^3$) |
|---|---|---|---|---|
| Wastewater | 9–380 | -- | -- | -- |
| Primary treatment | 20 | 25 | 10 | 0.03 |
| APT with filtration and disinfection | 1–5 | > 80 | 65 | 0.05 |
| Activated sludge with filtration and disinfection | 1–8 | 85 | 85 | 0.15 |

COD = carbon oxygen demand ; TSS = total suspended solids
[a] Including sludge stabilisation.
Source: Adapted from CNA (National Water Commission) (1995) and Jiménez and Chávez (1997)

Van Houtte (2001) studied an aquifer recharge system with an effluent treated with membranes, using microfiltration or ultrafiltration (in hollow or tubular fibres) as a pretreatment to reverse osmosis. In both cases, reverse osmosis effluents were free of total and faecal coliforms, as well as streptococcus. Also, heterotrophic plate count (HPC) and a natural hormone with endocrine disrupting potential were reduced by 2 log.

*Disinfection*

Disinfection is the main process used to control microorganisms. The challenge for producing recycled water is to determine, from one or several indicators, the efficiency of removal of pathogens (particularly viruses), given that each microorganism behaves differently during disinfection. Efficiency depends on the disinfecting agent, the type and variety of microorganism, the dosage and the exposure time. The water composition has an influence, which increases as the concentration and complexity increase. The main disinfectants used are chlorine, ozone and UV light, each of which is discussed below.

**Clorine**

Chlorine is a chemical agent that combines with many substances dissolved or suspended in water, such as organic material, hydrogen sulfide, manganese, iron, nitrites and ammonia. As a group, these substances are known as reductor compounds and their concentration varies from one body of water to another, depending on the content at the source and the extent of contamination of the water during use. During wastewater treatment many of these compounds remain in the water and are consequently found in the recycled water. Thus, when chlorine is added to water for reuse, some of the disinfectant is consumed by the reductor compounds, being converted into chlorides, organochlorinated compounds or chloramines, depending on whether the reductor compounds are inorganic, organic or contain ammonium derivatives respectively. The reductor compounds not only reduce the efficiency of chlorine disinfection, but also produce by-products that interfere with the later reuse of water for human consumption. Epidemiological studies suggest that chlorination by-products increase the risk of cancer (Batterman et al., 2002; Gibbons & Laha, 1999; Goldman & Murr, 2002; Korn, Andrews & Escobar, 2002; Monarca et al., 1998).

Chlorination by-products may be quantified as individual species such as chloroform; as groups of species such as trihalomethanes (THMs), haloacetic acids (HAAs) and aldehydes; or as total halogenated organics (TOX) (Goldman & Murr, 2002; KIWA, 1986; Korn et al., 2002). The occurrence of the different by-products is estimated at:

- 50% for THMs (trichloromethane or chloroform $CHC1_3$, dibromochloromethane $CHC1Br_2$, bromodichloromethane $CHC1_2Br$ and tribromomethane $CHBr_3$);
- 25% for HAAs (monochloroacetic, dichloroacetic, trichloroacetic, monobromoacetic and dibromoacetic acids);
- 7% for aldehydes.

Thus, the presence of THMs is indicative of the presence of other oxidized halogenated groups that are potentially hazardous, and that have not yet been characterized. For this reason, the US EPA has included two reduction limits: 80 µg/l in a first stage and 40 µg/l in a second stage (Gibbons and Laha, 1999; Golfinopoulos et al., 1998). There are several paths of exposure to THM for adults, such as ingestion of contaminated foods or drinking-water; and inhalation of vapours from rain, washing, swimming and personal hygiene (Batterman et al., 2002). Nevertheless, the microbiological risks of not chlorinating water have to be balanced against the long-term risks from consumption of by-products.

As regards virus removal by chlorine, a contact time of 1 hour and residual chlorine in a tertiary effluent eliminates 5–7 log units of virus, leaving less than one infectious unit per 1000 litres (Leong, 1983). Disinfection of viruses occurs through damage to the nucleic acid or protein capsid of the virus. However, as it has been shown that the viral nucleic acid must be damaged to achieve disinfection, chlorination is not always efficient unless operating conditions for carrying out both processes are assured.

**Ozone**

Ozone is very effective in virus control. It inactivates 3–4 log units in just a few minutes when oxidizing agents in the recycled water are at low levels. Where oxidation activity is present, ozone can still be effective, provided that the appropriate dose and contact time are used. The doses for several different ozone applications are shown in Table 3.18.

**Table 3.18 Ozone doses for various microorganisms present in water**

| Application | Theoretical ozone doses (mg/l) | Reference |
|---|---|---|
| Bacteriophages f2 | 0.033 | Garay & Cohn (1992) |
| Thermotolerant coliforms | 3–5 | Rakness et al (1993) |
| *Escherichia coli* | 0.53 | |
| *E. coli* | 0.239 | Garay & Cohn (1992) |
| Coxsackie virus | 0.51 | |
| Polio virus | 0.015 | |
| Porcine Pigina virus | 0.024 | |

By-products generated during ozone disinfection include diverse aldehydes, ketones and acids (Hoigné & Bader, 1977, 1978; Langlais, Reckhow & Brink, 1991). Many of these compounds are potentially toxic, but their concentrations are so low that the effect is minimal. The by-product is polyvinyl chloride, which can limit the reuse of water for human consumption in spite of being present at low concentrations. This compound is found in the ozonation of industrial effluents, so industrial discharges must be minimized in wastewater that will be reused for human consumption.

Indirectly, the main by-product of ozonation is $OH^{\bullet}$, which is a very reactive entity because it has an unpaired electron. Conditions that promote $OH^{\bullet}$ formation are pH greater than 9 and the presence of metallic ions forming the redox-pair ($Fe^{2+} \leftrightarrow Fe^{3+}$ and $Mn^{2+} \leftrightarrow Mn^{3+}$).

**Ultraviolet**

UV light disinfection is catching up with chlorination because it does not generate by-products, and because the strict regulations applied to organochlorinated agents have raised issues about chlorination and made the process expensive. Thus, over the last 20 years, the use of UV light has increased even in wastewater treatment plants (Droste, 1997). Unlike chlorination, UV light disinfects wastewater with no need for storage or handling of hazardous chemicals. Also, the short contact time periods (seconds or minutes) required for ozonation reduces the size of the treatment tanks and, therefore, the cost (Rajeshwar & Ibañez, 1997). Currently, many UV light disinfection systems have been planned or built and have proven to be inexpensive and competitive in comparison with chlorination. Furthermore, UV light has a wide field of application, as its simplicity of operation makes it suitable to rural and isolated communities.

Comparative values of disinfection doses required to kill different microorganisms are shown in Table 3.19. The highest values correspond to bacteriophages and protozoa. When assuming an initial virus concentration of $10^4$ vu/l, the virus removal shown in Table 3.20 for the overall treatment system is obtained.

**Table 3.19 Comparative value of UV disinfection dose for different microorganisms**

| Microorganism | Applied dose (mWs/cm$^2$) | Inactivation (log) | Conditions | Reference |
|---|---|---|---|---|
| ***Bacteria*** | | | | |
| Thermotolerant coliforms | 30–45 | 3–5 | Secondary and tertiary effluents | Lazarova et al. (1999) |
| Faecal streptococcus | 30–45 | 3–5 | | |
| Total coliforms | 35 | 3 | ns | Lazarova et al. (1998) |
| Thermotolerant coliforms | 35 | 3 | | |
| Faecal Streptococcus | 35 | 3 | | |
| *E. coli* | 10 | > 5 | Reference strain | Sommer et al. (1998) |
| *Bacillus subtilis* | 60 | 3 | Saline solution tests | |
| Thermotolerant coliforms | 15 | 3 | Secondary effluents | Jiménez & Beltrán (2002); Maya Beltrán & Jiménez (2002) |
| Faecal streptococcus | 15 | 2 | | |
| Salmonella typhi | 32 | 3 | | |
| *E. coli* | 1.3–3.0 | 1 | Reference strain<br>Filtered water | Abbaszadegan et al. (1997) |
| *E. coli* | 3–7 | 3 | ns | US EPA (1999) |
| ***Viruses*** | | | | |
| Bacteriophage MS2 | 17–200 | 2–5 | Secondary and tertiary effluents | Lazarova et al. (1999) |
| Bacteriophage MS2 | 100 | <5 | ns | Lazarova et al. (1998) |
| Bacteriophage φx174 | 30 | 6 | | |
| Bacteriophage MS2 | 93 | 4 | ns | US EPA (1999) |
| Bacteriophage MS2 | 30 | 2 | Monochromatic light. Test in saline solution and phosphate buffer at room temperature | Shin, Linden & Sobsey (2000) |
| Hepatitis A | 6–15 | 4 | ns | US EPA (1999) |
| ***Protozoa*** | | | | |
| *Acanthamoeba* | 60 | 2 | Secondary effluent | Maya Beltrán & Jiménez (2002) |
| *Giardia lamblia* | 63 | < 1 | ns | US EPA (1999) |
| *Giardia muris* | 82<br>121 | 1<br>2 | ns | US EPA (1999) |
| *Giardia muris* | 5–83 | 2–3 | Medium pressure lamp. Potable filtered water at ambient conditions. Noninfectious | Craik et al. (2000) |

| Microorganism | Applied dose (mWs/cm$^2$) | Inactivation (log) | Conditions | Reference |
|---|---|---|---|---|
| | | | tests used for measuring inactivation. | |
| *Cryptosporidium parvum* | 8800 | 2–3 | ns | US EPA (1999) |
| *C. parvum* | 10–25 | 2–3 | Potable filtered water at ambient conditions. Temperature changes. Use of noninfectious test for measuring inactivation. | Craik et al. (2001) |
| *C. parvum* | 19 | 4 | Medium pressure lamp. Use of noninfectious test for measuring inactivation. | Bukhari et al. (1999) |
| *C. parvum* | 3<br>3 | 3.4<br>3 | Medium pressure lamp. Use of noninfectious test for measuring inactivation. | Clancy et al. (2000) |
| *C. parvum* | 3 | 3 | Monochromatic light. Test in saline solution and phosphate buffer at room temperature | Shin, Linden & Sobsey (2000) |
| *C. parvum* | 2 | 1.8 | Use of wavelength between 250 and 275 nm | Linden, Shin & Sobsey (2000) |

ns = not specified

**Table 3.20 Virus removal in different treatment stages**

| Treatment | Removal |
|---|---|
| Primary | << 1 log |
| Secondary | > 2 log |
| Tertiary (coagulation–flocculation, sedimentation and filtration) | 3 log |
| With chlorination | additional 3 log |
| Tertiary with ozonation | 1 log more than chlorination |
| If membranes are used | 2 log more than chlorination, resulting in 8–9 log |

*Multibarrier concept*

It is difficult to ensure the quality of recycled water, because of the range and diversity of compounds that might be present. Therefore, a multiple barrier approach is the most effective way to provide the desired safety. This approach involves combining different processes and mechanisms that remove or inactivate the same kind of contaminants so that, if one barrier fails, the others complete the task. This principle is mainly used to control pathogens, although it is also used for organic materials and metals. A guide to the applicability of processes is shown in Table 3.21, in which the processes are at least 50% effective for the pollutant factor specified.

Potable reuse of water via an aquifer includes passage through the soil, mixing, dilution and retention in the aquifer, before transfer of the recycled water to a potable-water production plant. The multiple steps involved create multiple barriers at a low cost. That is, the planned indirect recycling approach is based on ensuring that no additional treatment will be carried out before recharging water. In the developed world, the entire (advanced) treatment for wastewater is carried out before introducing this water into the aquifer. This conservative approach has been applied with the objective of gaining more public acceptance. However, in countries where nonintentional recharge of wastewater is already happening in aquifers that are used for potable purposes, it is sufficient to provide at least some treatment to the wastewater before its infiltration, to improve the quality of the groundwater.

**Table 3.21 Advanced processes in multiple barrier programs**

| Category of contaminant | Biological treatment | Biological treatment with nutrient removal | Biological removal of nutrients | High lime with recarbonation | Coagulation–Flocculation | Chemical oxidation and disinfection |
|---|---|---|---|---|---|---|
| Suspended solids | X | X | X | X | X | |
| Dissolved solids | | | | | | |
| BOD | X | X | X | X | X | |
| TOC | X | X | | | X | |
| Volatile organics | X | X | X | X | | X |
| Heavy metals | X | X | | X | X | |
| Nutrients | | X | X | | X | |
| Radionuclides | X | X | X | | X | |
| Microbes | X | X | | X | X | X |

| Category of contaminant | Filtration | Activated carbon | Demineralisation with membranes | Air desorption | Particle removal with membranes |
|---|---|---|---|---|---|
| SS | X | | X | | X |
| DS | | | X | | |
| BOD | | X | X | | X |
| TOC | | X | X | | |
| Volatile organics | | X | X | X | |
| Heavy metals | | | X | | |
| Nutrients | | | X | X | X |
| Radionuclides | | | X | | |
| Microbes | X | | X | | X |

BOD = biochemical oxygen demand; DS = dissolved solids; SS = suspended solids; TOC = total organic carbon
Adapted from: McEwen (1998)

## 3.5 Aquifer recharge

### 3.5.1 A useful resource

Many cities and agricultural areas rely on the combined use of surface water and groundwater. However, for some cities that depend solely on surface water, there comes a time when groundwater is the most economic additional source of supply, taking into account the environmental cost of increased diversions or new dams. Where the groundwater quality is unsuitable for supply, improving the quality by using recharge in times of excess surface water can produce new groundwater supplies of suitable quality.

Water banking is a very simple concept. An aquifer can be used to store surface water during times of excess, to be recovered in times of shortage. An unexploited aquifer underlying or near a city represents a latent water resource that has the capacity to store, treat, and distribute water. Thus, an aquifer can be considered as a dam, a treatment plant and a reticulation network, for which the capital cost is a comparatively trivial access and restoration charge. The operating costs involve pre and post-treatment, pumping and monitoring (Dillon et al., 2001; Dillon, Toze & Pavelic, 2001).

One aim of water banking should be to improve the environmental values of the groundwater resource. Drinking-water has a higher resale value than lower classes of water such as that used for irrigation, and it has been suggested that the utility of an aquifer would be improved most by insisting that water injected into the aquifer must be of potable standard. However, in reality, treatment of recycled waters to potable standard is rarely economically viable. Also, this is not precluded as a future option as long as the principle of improving the environmental value of the resource is adopted. The existing environmental values of the groundwater system should be taken into account including existing beneficial extractive uses of the groundwater and ecosystem support values. That is, the potential for negative effects of aquifer storage recovery on groundwater quality and pressures should not be overlooked (Dillon et al., 2000).

### 3.5.2 Aquifer storage recovery system

An aquifer storage recovery system involves holding water in an appropriate underground formation, where it remains available in such a way that it can be recycled by extraction when needed. In such a system, wells can be used both for injection and for recovering water. The recovery systems can hold different qualities of water for later use, even in the case of shallow or saline water, and can have several objectives (Table 3.22).

The main advantage of underground storage is that there are no evaporation losses from the groundwater. Evaporation losses from continuously operated infiltration basins may range from 0.5 metres per year for temperate humid climates to 2.5 metres per year for hot dry climates. Thus, for year round operation of the system, evaporation losses may be on the order of 1% of the water going into the system. Also, groundwater recharge systems are sustainable, economical and have less ecoenvironmental problems than dams (Bouwer, 1997).

**Table 3.22 Objectives of the aquifer storage recovery system**

| Objectives | |
|---|---|
| Temporary storage for certain seasons | Nutrient lixiviation control |
| Long-term storage | Increase of well production |
| Emergency or strategic reserve storage | Defer expansion of water supply systems |
| Day storage | Storage of renovated water to be reused. |
| Decrease of disinfection by-products | Soil treatment |
| Reestablishment of underground water levels | Refine water quality. |
| Maintain pressure in the distribution network | Aggressive water stabilization |
| Maintain flow in the distribution network | Hydraulic control of contaminant plumes |
| Maintain water quality | Maintain water temperature for fish production |
| Prevent saline intrusion | Reduce environmental effects due to discharges |
| Agricultural supply | Compensation of soil salinity leaching |

Adapted from: Bouwer (1989); Pyne (1994); Oron (2001)

The water types that can be used for aquifer storage recovery include stormwater, municipal treated effluents and irrigation return flow. These are discussed below.

*Stormwater*

The quantity and quality of stormwater are determined by rainfall, catchment processes and human activities, which cause its flow and composition to vary in space and time. Compared to secondary treated sewage effluent, stormwater has higher levels of suspended solids, heavy metals and bacteria, but less dissolved solids, nutrients and oxygen demand (Dillon et al., 2001; Dillon, Toze & Pavelic, 2001).

Stormwater is often deliberately recharged by means of grasslined swales, French drains, porous pavements, drainage trenches, infiltration wells, percolating sewerage and percolating basins. The purpose of such practices is to dispose of urban runoff and decrease the risk of flooding, and to recharge groundwater. This has important implications for pollution of groundwater, because the varied, and often poor, quality of urban runoff may mean that some of these waters must undergo some degree of treatment before infiltration.

*Municipal treated effluents*

Municipal treated effluent is much more consistent in flow and composition than stormwater. Its quality is determined by source water, the nature of industries connected to sewer, and their proportion of sewer discharge, and the effluent treatment processes. Nutrient and microbiological content of recycled water may affect clogging (Rinck-Pfeiffer et al., 1998), transport and viability of pathogens (Pavelic et al., 1996, 1998), biogeochemical reactions within the aquifers, and water pressure and quality in wells of other groundwater users in the area.

*Irrigation return flow*

In most cities, watering of parks and gardens is common. Application is often carried out by unskilled workers, who may over-irrigation (Lerner, 1990). In arid climates, irrigation return flow may represent a substantial source of recharge to the groundwater, as shown in Table 3.23.

**Table 3.23 Example of water balances in urbanized aquifers (units: $10^3$ $m^3$/day)**

| City | Lima | Doha | Bermuda |
|---|---|---|---|
| Area ($km^2$) | 400 | 294 | 6.3 |
| Recharge from: | | | |
| Precipitation | 0 | 11.5 | 4.83 |
| Rivers | 280 | 0 | 0 |
| Agricultural irrigation | | 0 | 0 |
| Park irrigation | 390 | 37.6 | 0 |
| Leaking mains | 340 | 25.3 | 0 |
| Sewers and septic tanks | 0? | 17.6 | 0 |
| Soakways | 0 | - | 3.13 |

Source: Lerner (1990)

### 3.5.3 Management and operation of aquifer recharge

Complete technology exists for the design, construction and operation of aquifer recharge. The efficiency of the process, as regards the improvement in water quality, depends on the combination of soil filtration properties and the water deposit procedure (Bower, 1989). In turn, the infiltration capacity depends on the size of the grains of soil, their distribution, moisture content, amount of organic material and their chemical and electric adsorption properties. It also depends on how the recycled water or an effluent is applied (i.e. in wells or via recharging beds), and on the recharge regime. This is expressed by means of two parameters (Oron, 2001): hydraulic charge (flow rate per area unit, l/h m$^2$) and mass charge (BOD/day m$^2$ or TSS/day m$^2$, depending on the soil properties and on how the soil interacts with the effluent).

Contaminant migration and removal can be controlled through a combination of hydraulic and mass charges (McCaulou, Bales & Arnold, 1995). Hence, these charges are the critical factor in determining the manner of aquifer recharge, and how much treatment is needed before reinjection. Countries with better technology can hold more water volume in a smaller area, via infiltration, and even via direct injection. Several examples of intentional and unintentional recharge are described below.

*Examples of intentional recharge*

A review of international experience in aquifer recharge (Pavelic & Dillon, 1997) identified 45 cases studies, including 70 known sites in 12 countries, with published information on;

- site characteristics;
- recharge techniques;
- operational problems such as clogging; and the methods used to resolve them;
- monitoring of impacts on groundwater quality or the recovered water.

Of the 45 case studies, 71% used "natural" source waters (rivers, lakes and groundwater), with the remainder using treated sewage effluent (20%) or urban stormwater runoff (9%). Highly treated sewage effluent is commonly used in the USA. It yields a very consistent, high-quality (but expensive) water at a relatively uniform flow rate, making this attractive as a source of water for aquifer recharge (National Research Council, 1994). Retention in an aquifer provides the necessary contact with the natural environment to make recovery for potable reuse acceptable to consumers. The sustained injection of urban stormwater was found in only four documented cases, three of which were in South Australia.

The first aquifer recharge system used in the USA was put into operation in 1968. By 1994, there were already 20 aquifers used for this purpose in the USA (Table 3.24), and another 40 were in the process of being implemented or planned (Pyne, 1994). Total recharge at that time was 7.2 m$^3$/s, half of which was stored in fresh water aquifers. The other half was in saline aquifers or low-quality water aquifers, which had problems, such as natural contaminants like nitrates, barium (or boron), hydrogen sulfide, iron and manganese (Pyne, 1994).

The main motivation for the use of aquifer recharge in the USA has been economic. Typically, recharge increases supply capacity at a lower cost (half or less) than traditional alternatives. Recharge can involve recycled water or high-quality water, using different formations such as sand, clayey sand, sandstone, limestone, dolomite, basaltic and glacial drift aquifers. Usually confined or semiconfined aquifers are used; only rarely are nonconfined aquifers used.

*Examples of unintentional recharge*

Garcia-Fresca (2001) suggests that although urban-induced recharge is a known phenomenon, its magnitude is not known. Garcia-Fresca & Foster (1996) found this phenolmenon in Hat Yai, Thailand; Santa Cruz, Bolivia; Mérida, México; Lima, Peru; Caracas, Venezuela; Perth, Australia; Dresden, German; Wolverhampton, United Kingdom; Aguascalientes, Mexico and Los Angeles, USA: In some cases the increase over natural recharge was 100 fold. The findings of a review of the hydrogeological operation are presented in Figure 3.4. The circles and triangles represent the natural recharge and total recharge of several cities around the world, respectively. The difference between the two amounts reflects the magnitude of urban-induced recharge.

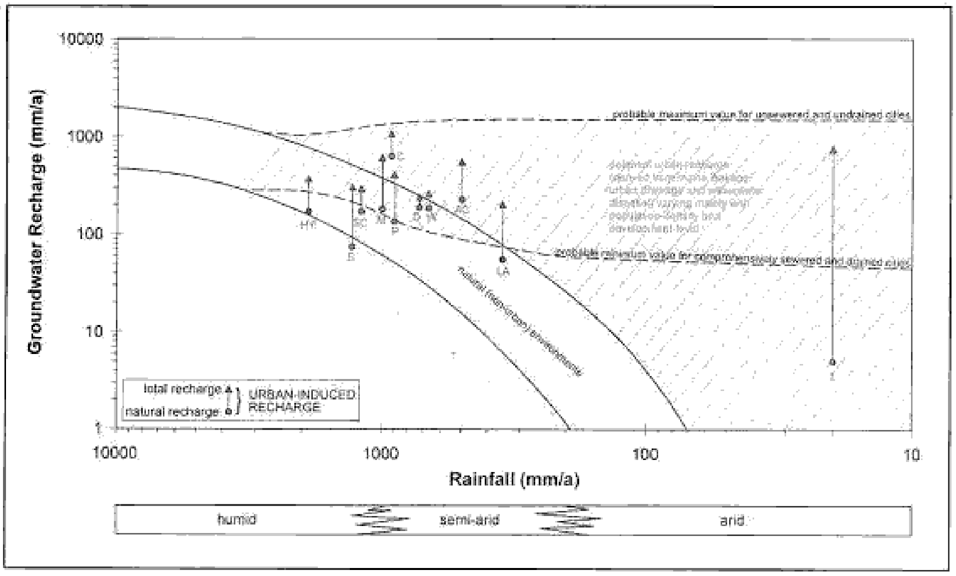


**Figure 3.4: Increase recharge to groundwater from urbanization**

Source: adapted from Foster (1996).
Legend symbols are: HY: Hat Yai, Thailand; SC: Santa Cruz, Bolivia; Mérida, México; L: Lima, Perú (Foster, 1996); S: Seoul, Korea (Kim *et al.*, 2001); C: Caracas, Venezuela (Séller ans Alvarado Rivas, 1999); P: Perth, Australia (Appleyard *et al.*, 1999); D: Dresden Germany (Grischek *et al.*, 1996); W: Wolverhamton, UK (Hooker *et al.*, 1999); Ac: Aguascalientes, México (Fernando and Gerardo, 1999); LA: Los Angeles, USA (Sharp *et al.*, 2000, Geomatrix estimated recharge from "delivered water" relative to recharge from precipitation under consitions).

**Table 3.24 Aquifer recharge systems operating in the USA up to May 1994**

| Place | Start of operation | Type of soil | Reclaiming capacity (Ml/day) |
|---|---|---|---|
| Wilwood, NJ | 1968 | Sand | 13 |
| Gordons Corner, NJ | 1971 | Clayey sand | 9 |
| Goleta, CA | 1978 | Silty, clayey sand | 23 |
| Manatee, FL | 1983 | Limestone | 15 |
| Peace River, FL | 1985 | Limestone | 18 |
| Cocoa, FL | 1987 | Limestone | 30 |
| Las Vegas, NV | 1988 | Sandstone | 385 |
| Port Malabar, FL | 1989 | Limestone | 4 |
| Oxnard, CA | 1989 | Sand | 33 |
| Chesapeake, VA | 1990 | Sand | 11 |
| Kerrville, TX | 1991 | Sandstone | 4 |
| N. Las Vegas, NV | 1991 | Sandstone | 6 |
| Seattle, WA | 1992 | Glacial drift | 26 |
| Calleguas, CA | 1992 | Sand | 6 |
| Pasadena, CA | 1992 | Sand | 19 |
| Centennial W.D., CO | 1993 | Sand | 3 |
| Boston Beach, FL | 1993 | Limestone | 6 |
| Marathon, FL | 1993 | Sand | 2 |
| Swimming River, NJ | 1993 | Clayey sand | 5 |
| Murray Avenue. NJ | 1994 | Clayey sand | 4 |
| **Total** | | **622 Ml/day or 7.2 $m^3$/s** | |

Source: Pyne (1994)

Many countries over-irrigate, leading to unintentional aquifer recharge. This practice should be recognized as a method of recharge method, even if it is unintentional, in order to be aware of the dangers and to maximize the advantages. A summary of some cases for which information on unintentional irrigation was obtained follows; however, the denial that this practice exists makes it difficult to obtain information. This section also presents other examples of controlled systems involving soil aquifer treatment.

**Mexico City**

In south Mexico City there is a reuse project that involves treating water to a secondary biological level and chlorinating it, before using it to feed a recreational lake, which in turn recharges the aquifer (Cifuentes et al., 2002). The lake provides feed flow to the Mexico City aquifer and, as a result, the surrounding wells also have "recycled" water. In addition, Capella (2000) estimated that wastewater infiltrations to the Mexico City aquifer from leaks of sewage are about 1 $m^3$/s, take place at a basaltic area in the south of the city, and account for the presence of certain pathogens in the supply water, because the same aquifer is used for supply.

**Mezquital valley**

Mexico is a country with apparent water sufficiency at the national level; however, two-thirds of the country suffers from lack of water, and domestic wastewater is often used for irrigation. In 1995, a total of 102 $m^3$/s of wastewater was used to irrigate 256,827 hectares in Mexico (Jiménez, Chávez & Capella, 1997). An example of a location where this practice occurs is the metropolitan zone of the Valley of Mexico, where rain and wastewater drain from the south to the Mezquital valley in the north. Untreated wastewater, at a rate of 52 $m^3$/s, has been used for irrigation of diverse crops and has contributed to the economic development of the region for more than a century. This is the largest and oldest scheme for agricultural irrigation using urban wastewater in the world.

As a result of this practice, the quality of the water table of the aquifer underlying the irrigation zone has improved. The unintentional artificial recharge is about 25 $m^3/s$, and springs with 100–600 l/s have been appearing for 35 years. This water is being used to supply 175 000 inhabitants with water for human consumption, with the only treatment being chlorination. Several studies indicate that this water meets the norms of potability, as well as 288 additional parameters from WHO human consumption guidelines, including toxicological tests (Jiménez et al., 2001).

### *Identification of recharge phenomenon*

Numerous environmental tracers are available to identify the source of recharge, but most of the tracers can enter groundwater from multiple sources (Lerner, 2002). Table 3.25 lists chemical marker species that can potentially be used to identify the source of recharge.

**Table 3.25 Source of possible marker species**

| Group of marker species | Potential sources of solutes | | | | | |
|---|---|---|---|---|---|---|
| | **Atmosphere** | **Geological materials** | **Agriculture** | **Mains water** | **Sewage** | **Industrial and commercial sites** |
| Major cations and anions | √ | √ | √ | √ | √ | √ |
| N species ($NO_3$, $NH_4$) | √ | | √ | √ | √ | √ |
| B and P | | √ | | | √ | √ |
| Other minor ions | √ | √ | | √ | √ | √ |
| Heavy metals | | √ | | | √ | √ |
| CFCs | √ | | | √ | √ | √ |
| THMs | | | | √ | √ | √ |
| Faecal organic compounds | | | | | √ | |
| Organics in detergents | | | | | √ | √ |
| Industrial organic chemicals | | | | | √ | √ |
| Microbiological species | | | | | √ | |
| Colloidal particles | | | | | √ | √ |

CFCs = chlorofluorocarbons; THMs = trihalomethanes
Source: Lerner (2002)

### *Issues concerning aquifer recharge*

Some issues which Oron (2001) considers critical for aquifer recharge are the following:

- the amount of treated wastewater is increasing and therefore demands more recharging area, which may become a problem in certain areas, depending on intensity of soil use and population density;

- the dilemma of whether or not to disinfect the effluent before recharging, knowing that to do so kills the soil microorganisms responsible for water treatment;
- the risk of introducing contaminants into the soil depuration (i.e. purifying) mechanisms, if there is no pretreatment or segregation of nonresidential discharges into the sewerage;
- leaching of contaminants in agricultural lands or dumps and landfills, or a greater usage of hazardous compounds in homes.

In addition to the issues mentioned above, aquifer characteristics such as depth, proximity to the coast, hydraulic conductivity and porosity should also be considered. From an engineering aspect, for each combination of water-soil factors, the operating conditions that imply a certain cost must be defined, as must specific environmental and health risks. This provides some idea of the usefulness of regionalizing this type of project.

### 3.5.4 Soil aquifer treatment

According to Cotruvo (2001), the main aspects to be analysed for recharging an aquifer for the purpose of human consumption are:

- the role of engineering pretreatment versus the efficiency and achievements of soil aquifer treatment
- a comparison of filtration with direct injection
- the reliability of the processes
- storage capacity and residence time
- mixture with native waters.

At present, this information is scarce, despite the existence of innumerable examples of unintentional recharge around the world (Foster, 2001).

High-rate soil treatment is a low-cost treatment via the soil, that can be used to produce irrigation water and water for other uses (Bouwer et al., 1980). Most of these kinds of systems are found in the USA, and they generally use secondary effluents, due to existing regulations. However, costs could be reduced by using primary effluents, whose main difference compared to secondary effluents is a greater content of organic material and solids (Rice & Bower, 1984). Laboratory studies have shown that the quality of water obtained with soil aquifer treatment is as good or better when applying a primary effluent as when a secondary effluent is used (Lance, Rice & Gilbert, 1980). It is important to point out that soil aquifer treatment improves water quality, even when the type of soil creates conditions considered unfavourable (Oron, 2001). However, if the content of suspended solids surpasses 80 mg/l, problems arise, the effects of which are more evident in the laboratory than in the field, where the soil can get fouled. Also, if the suspended solids are organic in nature, they will break down rapidly and fouling will be prevented; this does not occur with inorganic solids.

Using soil aquifer treatment, the following results are obtained: suspended solids less than 1 mg/l, organic carbon 3 mg/l, and total nitrogen 6 mg/l, with a phosphorus removal of almost 50%. Furthermore, a soil aquifer treatment system also effectively removes bacteria and viruses (Bouwer, 1984). Rapid infiltration with a constant minimal pretreatment produces a high-quality effluent as regards chemical oxygen demand (COD), N and P (Carlson et al., 1982). In fact, the infiltration rate and final quality of water is better when a primary effluent is used than when a secondary effluent is used (Figs. 3.5, 3.6 and 3.7). Furthermore, Lance & Whisler (1976) demonstrated, by adding dextrose, that the presence of carbon in an effluent promotes the ability of soil aquifer treatment to denitrify the soil. Passage of raw wastewater or an effluent through the porous zone (3 m minimum) provides an additional treatment that removes pathogens and residual organics (Fujita, Ding & Reinhard, 1996; Quanrud et al., 1996, Wilson et al., 1995).

The rate for fast infiltration soil aquifer treatmentis from 29 to 111 m/year (EPA, 1981). Because of this high rate, most of the water goes to the aquifer, whereas in low rate infiltration, most of the water is used for agriculture. These rates and the infiltration require appropriate handling, and they must be seen as recharge methods and ways to increase availability of underground water (Lance, Rice & Gilbert, 1980).

Soil aquifer treatment can also be used to recover rainwater and runoffs. Urban runoff water obtained during the rainy season is usually led to rivers or, in the best of cases, to treatment plants. Many perennial rivers carry water only during the rainy season and frequently this water is mixed with untreated or treated wastewater. The arbitrary nature of these flows, both in their generation and their collection, make it difficult to use them efficiently, despite the fact that said flows could constitute an additional source of water in arid zones, when combined with aquifer storage. There is much to be done in this regard, as little information is available as to quality and treatment needs. It is important to keep this possibility in mind, in order to develop it as part of an efficient overall water plan with other urban aspects of a city, such as street design and garbage collection (Oron, 2001).

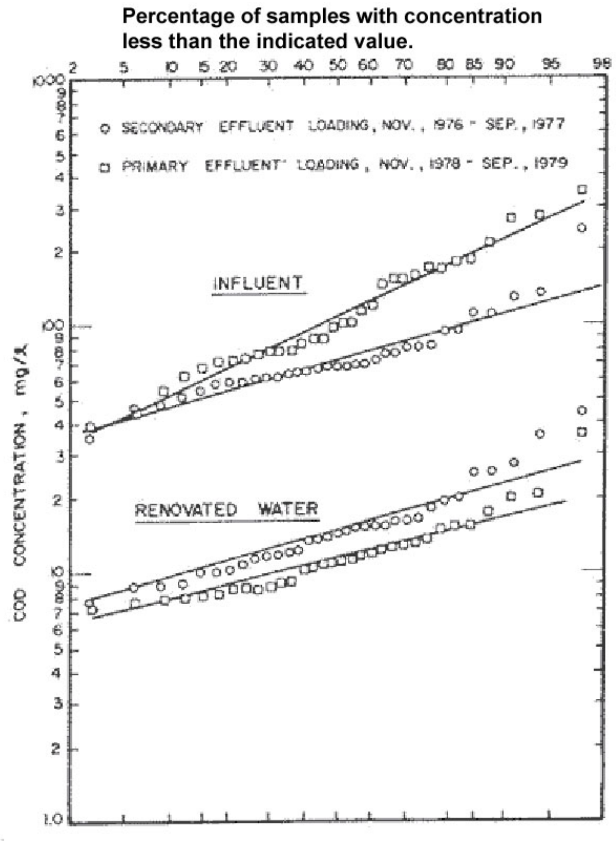


**Figure 3.5. COD concentration variations from Carlson *et al.*, 1982**

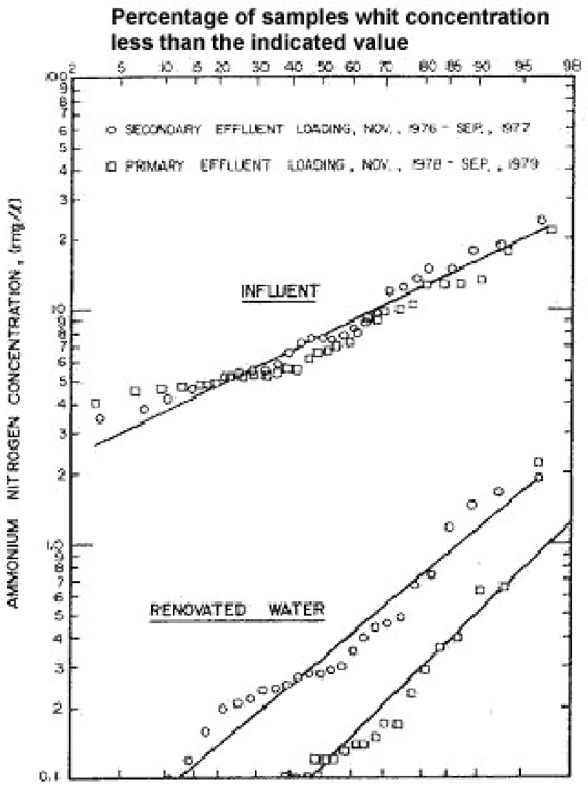


**Figure 3.6. COD concentration variations (Basin 1) from Carlson *et al.*, 1982**

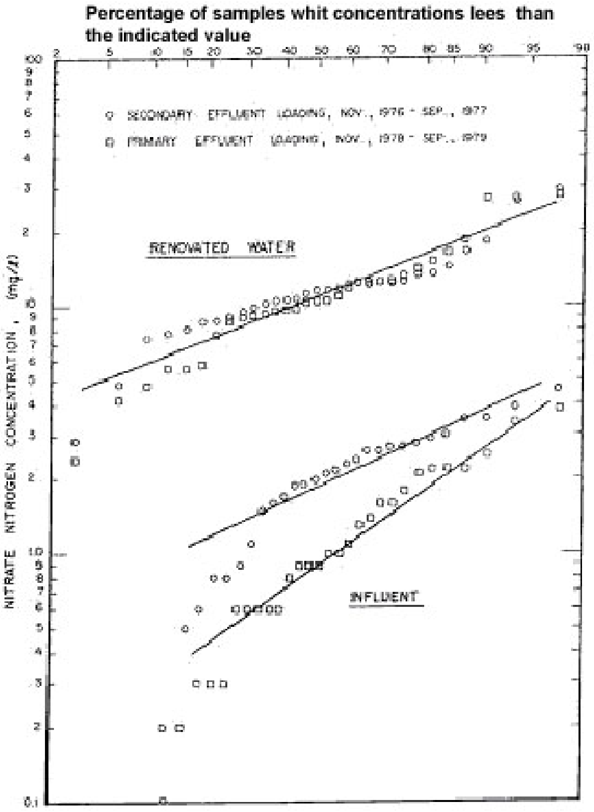


**Figure 3.7. COD concentration variations (Basin 1) from Carlson *et al.*, 1982**

*Factors influencing survival and transportation of viruses*

Due to their size, it is very difficult to separate viruses using primary and secondary processes. Studies by Lewis et al. (1986) comparing four processes (an oxidation pond, two biological filters and sedimentation with chlorination) demonstrate that none of the processes significantly removes enterovirus. On the other hand, soil filtration is efficient at absorbing viruses, which bind to the cations present in the soil particles.

Yates, Gerba & Kelly (1985) describe several factors that affect the survival and migration of viruses in the subsoil, including climate (temperature and rainfall), hydraulic conditions (water application rates and duration of wet and dry cycles), solar radiation, pH, organic matter, antagonistic microflora, soil type (texture and humidity) and the type of virus involved. Viruses may migrate long distances; studies show migrations of up to 67 m deep and 480 m horizontally. There is much information on virus survival in soil, but little on what happens to viruses in underground water (Table 3.26). The main factors influencing the survival and transportations of viruses are discussed below.

**Virus adsorption**

It is generally accepted that virus adsorption onto soil particles increases their survival and delays their transportation. However, this is not a permanent phenomenon because changes in the ion force, pH, humidity saturating the pores or saline concentration may dislodge viruses and make them move in the soil.

**Virus aggregation**

The formation of virus clusters in water makes them more resistant to disinfection. This may apply to virus clusters found in the soil, has although it has not yet been demonstrated.

**Temperature**

Perhaps the most important factor in the inactivation of viruses is temperature. It accounts for 78% of the process (Yates, Gerba & Kelly, 1985), although how it acts remains unknown.

**Microbiological activity**

It is believed that the presence of other microorganisms and their interaction with viruses is a removal mechanism, although it is not clear how this mechanism works, as the scarce literature on this subject is contradictory (Yates, Gerba & Kelly, 1985).

**Humidity**

It is commonly accepted that the absence of humidity has a negative effect on the persistence of viruses in soils, because saturation prevents viruses from being absorbed and thus allows them to reach greater depths. The water content (i.e. the humidity), can control virus transportation from a few centimeters in dry soil to several metres in saturated soil. Clean dry sand has low virus removal capacity (Berg, 1973), whereas wet sand has better retention capacity (Nestor & Costin, 1971).

**pH**

The effect of pH on virus survival has not been well studied. Sobsey (1983), indicates that an acid pH tends to deactivate viruses; however, the values and intensity of responses are different for each type of organism. According to Gerba and Bitton (1984), pH can affect adherence, and consequently the possibility of survival. These authors suggest that pH affects virus mobility because it affects the charge on the protein coat. At pH 7, most viruses carry a net negative charge and are not absorbed by most types of soil. At a higher pH, the negative charge on both the virus and the soil decreases but because the effect on soils is less intense, they can attract virus particles. This attraction can be modified by the presence in the soil of cations and humic and fulvic acids.

**Table 3.26 Survival of microroganisms in aquifers**

| Microorganism | Remarks | Reference |
|---|---|---|
| *Salmonella typhimurium*<br>*Streptococcus faecalis*<br>*E. coli*<br>Polio virus 1 and 2 | Poliovirus 1 and S. faecali have the lowest decay rates, followed by *S. typhimurium* and *E. coli*. Poliovirus 2 is the most rapidly inactivated. | Bitton et al. (1983) |
| Coxsackie virus B3<br>Polio virus1<br>Rotavirus SA11<br>Phages f2<br>*E. coli*<br>Faecal streptococcus | Enterovirus, poliovirus 1, and coxsackie virus B3 have lower decay rates than bacteria. Rotavirus SA11 and other phages f2 are inactivated more rapidly than bacteria. | Keswick et al. (1982) |
| Polio virus 1<br>Echovirus 1<br>MS-2coliphages | Similar decay rates. The use of MS2 coliphage as a model to predict animal origin virus survival is proposed. Of all the factors studied the one that affects survival the most was temperature. | Yates, Gerba & Kelly (1985) |

**Dissolved salts**

Aluminium and iron salts tend to inactivate viruses. Also, virus transportation is delayed in proportion to the salt content and cation balance, due to alteration of the adhesion properties (as discussed above in the section on pH).

**Organic material**

The influence of organic material on viruses is not yet well understood; some studies appear to show a protective action, while others show no effect at all. Humic and fulvic acids reduce infectivity, but this can be recovered by changing the ambient conditions. Organic material also prevents adsorption, a property that explains the use of plastic rather than glass material for obtaining samples for virus analysis (Bixby & O'Brien, 1979).

Powelson, Gerba & Yahya (1993) found that the kind of effluent (primary, secondary or tertiary) used has no effect on transportation of poliovirus 1, M52 and PRD1 in alluvial gross sand packed columns.

**Hydraulic conditions**

The rate at which water is applied to the soil has a definitive effect on the amount of viruses removed as well as on their adsorption. Removal efficiency increases as the rate decreases. Lance & Gerba (1984), demonstrated that increasing the rate from 0.6 to 1.2 m/day increased the number of viruses in the effluent of a 250 cm soil column (3% clay, 8% silt and 89% sand), but changing from 1.2 to 12 m/day gave no further deterioration in quality. These authors also observed that, when using nonsaturated soil, the depth to which viruses penetrated was much less. For infiltration rates below 5 m/day in loamy sand and sand soils, virus removal can be correlated to infiltration removal with up to 99% precision, at depths of up to 3 m. When combining wastewater treatment with infiltration in unsaturated soil, the distance between the discharge point and the extraction site (50 and 100 m, respectively) and water retention time of 6–12 months, the total virus removal rate is estimated between 13 and 17 log (Asano & Mujeriego, 1998; State of California, 1989).

**Types of virus**

Different types of viruses behave in different ways under the same conditions. This is true not only for different genera (e.g. adenovirus, enterovirus, rotavirus) but even for different strains, which complicates matters further (Goyal & Gerba, 1979).

**Soil properties**

Soil has a marked influence on virus survival and transportation. In general, viruses are more mobile in soils that have a gross texture than in finely textured karstic systems. Drewry & Eliassen (1968) reported that soils with clay and silt have better virus retention efficiency. Clays have proven to be excellent adsorbent materials due to their large surface area (Bitton, 1975). The composition of the soil affects virus survival; for example, aluminium-rich soils decrease survival and those high in phosphorus increase it (Hurst, Gerba & Cech, 1980).

### *Examples of soil aquifer treatment schemes*

The following section describes several examples of soil aquifer treatment systems.

**Vall-Llobrega, Catalonia, Spain**

A circular drained dune sand infiltration percolation filter, 1.5 m sand deep, with a surface of 565 $m^2$, was constructed in Vall-Llobrega, Catalonia, Spain. The filter was fed with activated sludge effluent using a pivot irrigation system equipped with low-pressure bubbles. The plant worked for two years, with the hydraulic load ranging from 0.165 to 0.35 m/day of infiltration surface (Salgot, Brissaud & Campos, 1996).

COD removal was 50%, with faecal coliforms reduced from more than 1000 to values of less than 137 CFU/100 ml for the different hydraulic rates. The system thus appears to be capable of reliably removing bacteria. Also, the use of a pivot irrigation system appears to be a major technological improvement over the infiltration percolation technique, because it allows uniform distribution of the infiltration with fractionation of the feeding as necessary.

**El Dan, Tel Aviv region, Israel**

The El Dan region in Tel-Aviv, Israel infiltrates $130 \times 10^6$ $m^3$/year of a biological secondary effluent ,with nitrogen elimination through a series of open ponds. The cycle lasts four days and the recharge rate is 70–210 m/year. In this way, 90% of BOD, TSS, ammonia nitrogen, and phosphorus are removed (Table 3.27). However, there is an increase in salinity due to the proximity to the coast, and in boron content due to the type of soil (Oron, 2001).

**Table 3.27 Characteristics of soil aquifer treatment at El Dan**

| Parameter (mg/l) | Before treatment | After treatment | Removal (%) |
|---|---|---|---|
| BOD | 6 | < 0.5 | > 92 |
| COD | 46 | 7 | 85 |
| TSS | 7 | 0 | 100 |
| DOC | 11 | 2.8 | 75 |
| Detergents | 0.241 | < 0.1 | > 55 |
| Mineral oils | 0.4 | 0.3 | 25 |
| Phenols | 4 | 1 | 75 |
| Ammonia as N | 8.2 | <0.02 | >99 |
| Total N | 12 | 5.4 | 55 |
| Phosphorus | 2.7 | 0.05 | 98 |

BOD = biochemical oxygen demand; COD = carbon oxygen demand; DOC = dissolved organic carbon; TSS = total suspended solids
Source: Oron (2001)

**Belgium**

Van Houtte (2001) studied soil aquifer treatment system in sand followed by membrane filtration of a biological secondary effluent or ultrafiltration with reverse osmosis. The system achieved good bacteriological control and eliminated nutrients and organic material, provided that passage time was sufficient. Disadvantages include an increase in the iron content of water that can lead to membrane fouling and must be controlled by an acid wash, as well as an increase in alkalinity. Advantages include the reduction in biofouling observed in subsequent membranes, treatment costs, recharge and storing of water in the aquifer. Further, in this case, chlorination can be performed before passing through the membranes, thereby preventing biofouling and controlling the presence of by-products. This would make a direct reclaim for human consumption possible. The water quality data for the different treatment stages are shown in Table 3.28.

**Table 3.28 Soil aquifer treatment coupled with membranes**

| Parameter | Effluent | SAT | MF filtered | RO filtered |
|---|---|---|---|---|
| Conductivity (µS/cm) | 1252 | 2100 | 2100 | 38 |
| BOD ($mgO_2$/l) | 3 | 7 | 2.3 | <2 |
| COD ($mgO_2$/l) | 30 | 16 | 17.6 | <5 |
| Sodium (mg/l) | 113 | 225 | | 10 |
| Chlorides (mg/l) | 171 | 413 | | 8 |
| Sulfates (mg/l) | 54 | 73 | | <10 |
| Phosphorus (mg/l) | 0.6 | 1.12 | 1.08 | 0.3 |
| Nitrates (mg/l) | 40 | 1.1 | 0.4 | <0.1 |
| TOC (mg/l) | 20 | 14.5 | 12.6 | <5 |
| UV absorption ($cm^{-1}$) | 0.39 | 0.26 | 0.24 | 0< |
| SS (mg/l) | 25 | 11 | <1 | <0.1 |
| HPC at 22 ºC | 31111 | 677 | 123 | |
| HPC at 37 ºC | 1833 | 122 | 47 | |

BOD = biochemical oxygen demand; COD = carbon oxygen demand; HPC =heterotrophic plate count; MF = microfiltration; RO = reverse osmosis; SAT = soil aquifer treatment; SS = suspended solids; TOC = total organic carbon; UV = ultraviolet
Adapted from Van Houtte (2001)

**USA**

According to Crook (1998), in most of the infiltration projects in the USA, water is treated to a secondary level. When the water is to be subsequently used for human consumption, treatment is upgraded to a tertiary level (i.e. secondary level with filtration and disinfection or an advanced treatment). However, this is not congruent with what other authors specialized in aquifer storage recovery say about the USA (Bouwer et al., 1980).

### 3.5.5 Examples of direct injection programs

Several of the direct injection projects reported in USA are described below.

**Pilot plant in Mission Valley, San Diego, California**

This consists of a pilot level treatment with water lilies for the secondary level; a solid contact clarifier using ferric chloride, a gravity filter and dual packing for the tertiary stage; and UV light disinfection, cartridge filtration, reverse osmosis, air desorption and activated carbon, as an advanced treatment. The secondary level had a capacity of 6.8 l/s and the tertiary level, 2.2 l/s (Olivieri, Eisenberg & Cooper, 1998).

The water recycled in this manner was compared with the inlet water from a conventional potabilization plant that supplies potable water to the San Diego area. Table 3.29 compares the microbiological quality of these three components: inlet water to the potabilization plant, raw residual water and effluent from the nonconventional advanced treatment plant. The results indicate the high quality of the recycled water

**Table 3.29 Comparison of supply water to San Diego and water recycled in a nonconventional process**

| Parameter | Influent from conventional potabilization plant | | Wastewater | | Recycled water | |
|---|---|---|---|---|---|---|
| | Min | Max | Min | Max | Min | Max |
| Total coliforms in 100 ml | 176 | 425 | $10^6$ | $10^8$ | 7 | 413 |
| Thermotolerant coliforms in 100ml | 133 | 425 | $10^6$ | $10^8$ | 2 | 413 |
| Faecal Streptococcus | 152 | 425 | $10^5$ | $10^8$ | 2 | 411 |
| Phages (F+ RNA), PFU/l | < 0.007 | 0.72 | $10^5$ | $10^7$ | < 0.01 | < 0.07 |
| Enteric virus | < 0.0005 | < 0.004 | < 1 | 1067 | < 0.001 | < 0.009 |
| Salmonella (MPN/l)[a] | <0.22 | 22 | < 22 | 92 | < 0.22 | < 1.6 |
| *Giardia lamblia*, cysts/l | < 0.001 | < 0.006 | < 0.004 | 3200 | < 0.001 | < 0.01 |
| *Cryptosporidium* oocysts/l | < 0.001 | < 0.006 | < 0.004 | 25 | < 0.001 | 0.02 |

MPN = most probable number; PFU = plaque forming units
Adapted from Olivier and Eisenberg et al. (1998)

**Fred Hervey water reclamation and groundwater recharge project**

The Bolson del Hueco aquifer is recharged by direct injection with residual water treated at a potable level. The influent and effluent quality, as well as the quality of the aquifer that is recharged is shown in Table 3.30. The recharge permit requires the monitoring of 23 parameters and a record of the average for 30 days. It also requires that the result of 8-hour batch analysis be less than 10 mg nitrates N/l and less than 5 nephelometric turbidity units (NTU) turbidity. The treatment consists of a biological process in two stages, with powdered activated carbon, treatment with lime, recarbonation, sand filtration, ozonation, adsorption in granular activated carbon and storage. Water is carried to 11 injection wells and the residence time is at least two years before extraction. Again, the reclaimed water is of high quality.

**Table 3.30 Water quality data from the Fred Hervey recharge project**

| Parameter | Influent | Effluent | Bolson del hueco | Recharge permit limits |
|---|---|---|---|---|
| Alkalinity (mg/l) | 250 | 150 | 129 | |
| Ammonia N (mg/l) | 25 | <0.1 | | |
| Boron (mg/l) | <1 | <0.8 | | |
| Calcium (mg/l) | 35 | 50 | 45 | |
| Chlorides (mg/l) | 122 | 140 | 64.7 | 300 |
| COD (mg/l) | 400 | <10 | | |
| *E. coli* (MPN/100mL) | | 0 | | |
| Electrical conductivity (dS/m) | 1.20 | 1.00 | 0.51 | |
| Fluorides (mg/l) | 0.9 | 0.9 | 1 | |
| Magnesium (mg/l) | 7.9 | 3 | 10.6 | |
| pH | 7.1 | 7.5–8 | 8.1 | |
| P-$PO_4$ (mg/l) | 20 | 0.1 | <0.5 | |
| Sodium (mg/l) | 192 | 145 | 96 | |
| Sulfates (mg/l) | 125 | 125 | 53.7 | 300 |
| Silica (mg/l) | 38 | 38 | 38 | |
| Total hardness (mg/l) | 120 | 140 | 102 | |
| Total nitrogen (mg/l) | 36 | 2 | | |
| N-nitrates (mg/l) | 0 | 1.5 | <2.1 | 10 |
| TOC (mg/l) | | < 2 | | |
| Turbidity (mg/l) | 97 | <0.5 | 0.14 | 1 |
| TSS (mg/l) | 150 | <1 | | |
| Cyanides (mg/l) | | 0.02 | | |
| Arsenic (mg/l) | 0.008 | 0.05 | | 0.05 |
| Barium (mg/l) | 0.46 | 0.1 | | 1 |
| Cadmium (mg/l) | <0.01 | <0.01 | <0.01 | 0.01 |
| Chromium (mg/l) | 0.01 | 0.01 | <0.05 | 0.05 |
| Copper (mg/l) | < 0.05 | <0.05 | <0.1 | 1 |
| Iron (mg/l) | <0.31 | 0.05 | <0.1 | 0.3 |
| Lead (mg/l) | <0.05 | <0.05 | 0.05 | 0.05 |
| Manganese (mg/l) | <0.05 | <0.05 | 0.01 | 0.05 |
| Mercury (mg/l) | 0.002 | 0.0014 | | 0.002 |
| Selenium (mg/l) | <0.0035 | 0.01 | | 0.01 |
| Silver (mg/l) | <0.01 | <0.01 | | 0.05 |
| Zinc (mg/l) | <0.1 | <1 | 0.1 | 5 |
| Aluminium (mg/l) | 0.28 | <0.15 | | |
| Colour (mg/l) | 52 | <10 | | |
| MBAS (mg/l) | 3.5 | | | |
| Corrosion | 0.2 | | | |
| Hydrogen sulfide (mg/l) | | Non corrosive | | |
| Odour, TON | 11 | 0.01 | | |
| TDS (mg/l) | 770 | 1 | 391 | 1000 |
| Dissolved oxygen mg/l | | 1 | | |

| Parameter | Influent | Effluent | Bolson del hueco | Recharge permit limits |
|---|---|---|---|---|
| Phenol (mg/l) (mg/l) | 0.004 | 0.001 | 0.001 | |
| Residual Chlorine (mg/l) | | 0.25 free | | |
| Virus | | 0 | | |
| Endrin (mg/l) | | <0.002 | | 0.0002 |
| Lindane (mg/l) | | <0.004 | | 0.004 |
| Metoxychlorine (mg/l) | | <0.1 | | 0.1 |
| Toxaphene (mg/l) | | <0.005 | | 0.005 |
| 2-4 D (mg/l) | | <0.1 | | 0.1 |
| 2,4,5-TP (mg/l) | | <0.01 | | 0.005 |
| THMs (mg/l) | | <0.1 | | |
| Radio 226 and 228 pCi/l | 32 | <5 | | |
| Alpha pCi/l | | <15 | | |
| Beta mrem/year | | <4 | | |

2-4 D = 2,4-dichlorophenoxyacetic acid; 2,4,5-TP = 2,4,5-trichlorophenoxypropionic acid COD = carbon oxygen demand; MBAS = methylene blue active substances; TDS = total dissolved solids; THMs = trihalomethanes; TOC = total organic carbon; TON = threshold odour number; TSS = total suspended solids; UV = ultraviolet
Data correspond to the effluent and influent design data
Adapted from Bureau of Reclamation (1993)

**Denver's Potable Water Reuse Demonstration Project**

Denver's Potable Water Reuse Demonstration Project was designed to examine the feasibility of converting secondary treated wastewater to potable water quality. The goal was to investigate the correlation between product security, economic and technical feasibility, and public acceptance. To this end, a 44 l/s demonstration plant was built at a total cost of US$ 30 million. The process provides multiple barriers of protection against the primary contaminant groups (Table 3.31). The water reuse treatment process was composed of coagulation–flocculation with lime, recarbonation, filtration, selective ion exchange, carbon adsorption first step, ozonation, carbon adsorption second step, reverse osmosis and disinfection with chlorine dioxide. The level of protection may appear excessive, but it is certainly adequate to achieve the stated objectives (Lauer, 1991).

A summary of water quality from the two types of resulting effluents (with reverse osmosis, and ultrafiltration) (Lauer & Rogers, 1994) is shown in Table 3.32. The reliability of recycled water was demonstrated, and the fact that it had a higher quality than potable water.

In the case of the Denver plant, even though the results of trials on living entities were all negative, the National Academy of Science's Panel on Quality Criteria for Water Reuse (NAS, 1982) considered them insufficient and even inadequate. Borzolleca and Carchman (1982) researched the effects of certain multichlorinated organic compounds (2,4 nitrotoluene, 1,2,3,4 tetrabromobutane, hexachlorobenzene, dibromochloromethane and chloroform), and found that high concentrations thereof produce severe effects. Although the experiments used a single chemical compound and very high concentrations, they are considered sufficiently valid to affirm that the water carries risk.

**Table 3.31 Multiple barriers against primary contaminant groups in the Denver project**

| Organism | Number of barriers | Description of barriers |
|---|---|---|
| Viruses and bacteria | 5 | High pH lime clarification<br>Ultraviolet irradiation<br>Reverse osmosis<br>Ozonation<br>Chloramination |
| Protozoa | 4 | High pH lime clarification<br>Filtration<br>Reverse osmosis<br>Ozonation |
| Metals and inorganic compounds | 3 | High pH lime clarification<br>Activated carbon adsorption<br>Reverse osmosis |
| Organic compounds | 4 | High pH lime clarification<br>Activated carbon adsorption<br>Reverse osmosis<br>Air stripping |

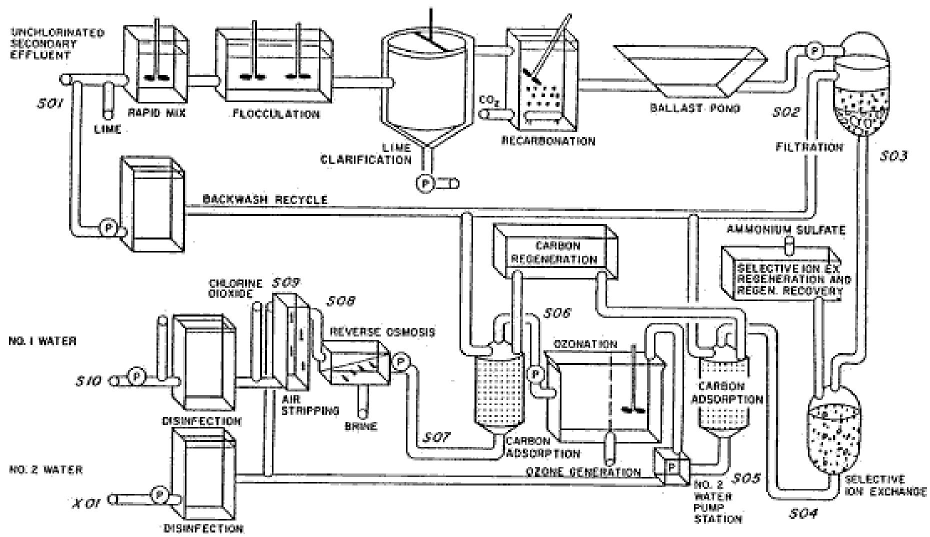


**Figure 3.8. Water reuse treatment process. Health effects with ultrafiltration sidestream (Source: Lauer *et al.*, 1990).**

**Table 3.32 Quality of water recycled by reverse osmosis and ultrafiltration, and comparison with potable water from the City of Denver.**

| Parameter | Influent to reclaiming plant | Product | | Denver potable water |
|---|---|---|---|---|
| | | RO | UF | |
| Total alkalinity — $CaCO_3$ | 247 | 3 | 166 | 60 |
| Total hardness —$CaCO_3$ | 203 | 6 | 108 | 107 |
| Total suspended solids | 14.2 | a | A | A |
| Total dissolved solids | 583 | 18 | 352 | 174 |

| **Parameter** | **Influent to reclaiming plant** | **Product** | | **Denver potable water** |
|---|---|---|---|---|
| | | **RO** | **UF** | |
| Electrical conductivity (dS/m) | 0.907 | 0.67 | 0.648 | 0.263 |
| pH | 6.9 | 6.6 | 7.8 | 7.8 |
| Turbidity (NTU) | 9.2 | 0.06 | 0.2 | 0.3 |
| Particle size (No./50 ml) | | | | |
| >128µ | b | a | a | a |
| 64–128µ | – | a | a | 1 |
| 32–64µ | – | 1.2 | 18 | 18 |
| 16–32µ | – | 58 | 100 | 168 |
| 8–16µ | – | 147 | 448 | 930 |
| 4–8µ | – | 219 | 1290 | 3460 |
| Radiological (pCi/l) | | | | |
| Gross alpha | 2.9 | <0.1 | <0.1 | 1.3 |
| Gross beta | 10.0 | <0.4 | 5.6 | 2.3 |
| Radio 228 | <1 | <1 | <1 | <1 |
| Radio 226 | <0.3 | <0.3 | <0.3 | <0.3 |
| Tritium | <100 | <100 | <100 | <100 |
| Radon 222 | <20 | <20 | <20 | <20 |
| Total plutonium | <0.02 | <0.02 | <0.02 | <0.02 |
| Total uranium | 0.004 | <0.0006 | <0.0006 | <0.002 |
| Microbiology | | | | |
| m-HPC (No./ml) | $1.3x10^6$ | a | 350[c] | 3.3 |
| TC (MPN/100ml) | $7.7x10^5$ | a | a | a |
| Thermotolerant coliforms (MPN/100ml) | $6.3x10^4$ | a | a | a |
| Faecal streptococcus, (MPN/100ml) | $9.3x10^3$ | a | a | a |
| Coliphages B (No./100 ml) | $1.7x10^4$ | a | a | a |
| Coliphages C (No./100ml) | $4.8x10^4$ | a | a | a |
| *Giardia* (cysts/l) | 0.8 | a | a | a |
| *E. coli* (cysts/l) | 0.5 | a | a | a |
| Nematodes (No./l) | 3.8 | a | a | a |
| Enteric virus (No./l) | – | a | a | a |
| Entamoeba histolytica (cysts/l) | a | a | a | a |
| *Cryptosporidia* (oocysts/l) | 0.1 | a | a | a |
| Algae (No./ml) | 1.1 | a | a | 1.9 |
| Clostridium perfringens (No./100ml) | $8.5x10^3$ | <0.2 | <0.2 | <0.2 |
| *Shigella* | Present | Absent | Absent | Absent |
| *Salmonella* | Present | Absent | Absent | Absent |
| *Campylobacter* | Present | Absent | Absent | Absent |
| *Legionella* | Present | Absent | Absent | Absent |
| Inorganic elements[d] (mg/l) | | | | |
| Aluminium | 0.051 | a | a | 0.2 |
| Arsenic | 0.001 | a | a | a |
| Boron | 0.4 | 0.2 | 0.3 | 0.1 |

| Parameter | Influent to reclaiming plant | Product | | Denver potable water |
|---|---|---|---|---|
| | | RO | UF | |
| Bromine | a | a | a | a |
| Cadmium | a | a | a | a |
| Calcium | 77 | 1 | 38 | 27 |
| Chlorides | 98 | 19 | 96 | 22 |
| Chromium | 0.003 | a | a | a |
| Copper | 0.024 | 0.009 | 0.01 | 0.006 |
| Cyanides | a | a | a | a |
| Fluorides | 1.4 | a | 0.8 | 0.8 |
| Iron | 0.025 | 0.02 | 0.07 | 0.03 |
| Potassium | 13.5 | 0.07 | 9.1 | 2.0 |
| Magnesium | 12.6 | 0.1 | 1.8 | 7.2 |
| Manganese | 0.103 | a | a | 0.008 |
| Mercury | 0.0001 | a | a | a |
| Molybdenum | 0.019 | a | 0.004 | 0.02 |
| TKN | 29.5 | 5 | 19 | 0.9 |
| Ammonia-N | 26.0 | 5 | 19 | 0.6 |
| Nitrates-N | 0.2 | 0.1 | 0.3 | 0.1 |
| Nitrites-N | a | a | a | a |
| Nickel | 0.007 | a | a | a |
| Total phosphorus | 5.6 | 0.02 | 0.05 | 0.01 |
| Selenium | a | a | a | a |
| Silica | 15.0 | a | 8.8 | 6.4 |
| Strontium | 0.44 | 1 | 0.13 | 0.2 |
| Sulfate | 158 | a | 58 | 47 |
| Lead | 0.002 | a | a | a |
| Uranium | 0.003 | 0.006 | a | 0.001 |
| Zinc | 0.036 | 4.8 | 0.016 | 0.006 |
| Sodium | 119 | a | 78 | 18 |
| Lithium | 0.018 | a | 0.014 | 0.007 |
| Titanium | 0.107 | a | 0.035 | 0.005 |
| Barium | 0.034 | a | a | 0.04 |
| Silver | 0.001 | a | a | a |
| Rubidium | 0.004 | a | 0.003 | 0.001 |
| Vanadium | 0.002 | a | a | a |
| Iodide | a | a | a | a |
| Antimony | a | a | a | a |
| Organics (µg/l) | | | | |
| Total organic carbon (mg/l) | 16.5 | A | 0.7 | 2.0 |
| Total organic halogen (mg/l) | 109 | 8 | 23 | 45 |
| MBSA (mg/l) | 400 | a | a | a |
| Total trihalomethanes (µg/l) | 2.9 | a | a | 3.9 |
| Methylene chloride (µg/l) | 17.4 | a | a | a |

| Parameter | Influent to reclaiming plant | Product | | Denver potable water |
|---|---|---|---|---|
| | | RO | UF | |
| Tetrachloroethene (µg/l) | 9.6 | a | a | a |
| 1,1,1 trichloroethane (µg/l) | 2.7 | a | a | a |
| Trichloroethene (µg/l) | 0.7 | a | a | a |
| 1,4 dichlorobenzene (µg/l) | 2.1 | a | a | a |
| Formaldehyde (µg/l) | a | a | 12.4 | a |
| Acetaldehyde (µg/l) | 9.5 | a | 7.2 | a |
| Dichloroacetic acid (µg/l) | 1.0 | a | a | 3.9 |
| Trichloroacetic acid (µg/l) | 5.6 | a | a | a |

NTU = nephelometric turbidity units; RO = reverse osmosis; MBSA = methylene blue active substances; UF = ultrafiltration;
a: Below detection limits
b: (-) not analysed
c: Disinfection is not considered optimum at pilot scale
d Other parameters were determined, but their concentration was below the detectable limit, and the information available was insufficient. These elements were: hafnium, holmium, tellurium, beryllium, iridium, terbium, cobalt, lanthanum, lutetium, neodymium, thallium, bismuth, niobium, tin, cerium, osmium, tungsten, caesium, palladium, dysprosium, platinum, ytterbium, erbium, praseodymium, yttrium, europium, zirconium, gadolinium, rhodium, gallium, germanium, ruthenium, gold, samarium and scandium.
Adapted from: Lauer et al. (1988)

The Denver study was carried out for 13 years. The main contributions of this study were:

- a treatment scheme that produced a reliable product meeting potable water standards in the secondary effluent;
- studies on chronic toxicity and carcinogenesis carried out for two years, which demonstrated no effects in any of the samples of recycled water;
- comparative reproductive studies carried out using potable water as reference, which demonstrated no adverse effects;
- physical, chemical, and microbiological analyses, which demonstrated that recycled water was of higher quality than potable water;
- acceptance by the public as long as the need to use this kind of water was demonstrated;
- an indication that options based on natural barriers are more acceptable, because there is awareness that any kind of human process is subject to failure.

**Orange County, California**

Orange County has been practicing aquifer recharge, via both infiltration and direct injection, for over 20 years, to maximize the use of local water resources. In the early 1950s, the overexploitation of aquifers led to saline intrusion into fresh water aquifers, and recharge was implemented to control this. To increase water volume and water injection quality, a program called Water Factory 21 (WF-21) was created, as a water-reclaiming project for direct advance injection into the aquifer. Another project involves recharging via infiltration ponds using natural percolation. Together, the projects increase the availability of water from 6.2 $x10^7$ to 9.3 x $10^7$ m/year (Mills et al., 1998). WF-21 has a capacity of 660 l/s; since 1976 it has consistently produced water exceeding the standards for human consumption. The results of monitoring the effluent quality are shown in Table 3.33.

**Table 3.33 Water quality obtained in WF-21**

| Component | Selected parameters for permitted conditions[a] (mg/l) | Average concentration in 1994 for injection water[b] (mg/l) |
|---|---|---|
| Boron | 0.7 | 0.2 |
| Chloride | 120 | 90 |
| Fluoride | 1.0 | 0.5 |
| Total nitrogen | 10 | 3.7 |
| Sodium | 115 | 64 |
| Sulfate | 125 | 40 |
| pH | 6.5–8.5 | 7.4 |
| Total dissolved solid | 500 | 237 |
| Hardness | 180 | 33 |
| Total organic carbon[c] | 2.0 | 1.9 |
| Aluminium | 1.0 | 0.05 |
| Arsenic | 0.05 | 0.002 |
| Barium | 1.0 | 0.006 |
| Cadmium | 0.01 | 0.001 |
| Chromium | 0.05 | 0.001 |
| Cobalt | 0.2 | 0.002 |
| Copper | 1.0 | 0.008 |
| Cyanide | 0.2 | 0.04 |
| Iron | 0.3 | 0.054 |
| Lead | 0.05 | 0.006 |
| Manganese | 0.05 | 0.004 |
| Mercury | 0.002 | 0.0005 |
| MBAS | 0.5 | 0.06 |
| Selenium | 0.01 | 0.005 |
| Silver | 0.05 | 0.001 |
| Zinc | 5.0 | 0.057 |

[a] Regional Water Quality Control Board No. 91-121, adopted November 15, 1991.
[b] Recycled water blended with deep well water. Averages based on values above or at the detection limit.
[c] Total organic carbon concentration of 2 mg/l becomes effective only when the percentage of reclaimed water injected first exceeds 67% and is based on the quarterly average of daily samples.
Adapted from: Mills et al. (1998)

Another intensive monitoring process included the main groups of organics, potential trihalomethanes, halogenated organics, UV absorbance and specific nonvolatile organic compounds. The monitoring found very low concentrations of brominated forms of alkyl phenol polyethoxy carboxilates (APECs) in both filtrated effluents and chlorinated granular activated carbon effluents. By 1993, APECs were found again, and it is thought that they may serve as excellent tracers for recycled water and its movement within the aquifer. Other compounds that could serve this purpose are ethylene-diamine-tetracetic acid (EDTA) and nitrile acetic acid (NAA). In addition to this program, rainwater recharge to control saline intrusion is also practiced in Orange County.

**Summary of reclaiming projects in the USA**

A summary of unintentional water recycling projects is presented in Table 3.34.

**Table 3.34 Summary of international recycling projects**

| Year | Place | Description |
|---|---|---|
| 1956–1957 | Chanute, Kansas | Lack of water forced the use of direct potable reclaiming. Secondary level-treated water was introduced to a potabilization plant where it was super-chlorinated. No adverse effect was identified, however, after several cycles, water turned yellowish. |
| 1960 | USPS | Advanced Waste Treatment Research Program of the USA Health Service for the purpose of developing new water reclaiming technology. |
| 1962 | Whittier Narrows, California | Aquifer recharge in Los Angeles County by infiltration of a filtrated secondary effluent in dual media. Recharge was 16% of total supply. Depending on their location, local population is exposed to a range of 0–23% of recycled water. A state panel concluded that the water is safe for human consumption. |
| 1965 | Ruth Lake Tahoe | First advanced treatment plant for nutrient removal |
| 1969 | Windhoek, Namibia | Pilot plant operation begins, with launch in 1971 of the first plant in the world to reuse water for human consumption. Epidemiological studies showed no increase in disease or illness due to said reuse. |
| 1972 | Federal Water Pollution Control Act | This act controls discharges, thereby indirectly promoting the reuse and reclaiming of water. |
| 1976 | Orange County, California | Operates Water Factory 21. The plant has a capacity of 657 l/s, treating a nonchlorinated secondary effluent to recharge an aquifer, to control saline intrusion. Treatment includes lime clarification, air desorption, recarbonation, filtration, activated carbon, and treating part of the effluent by reverse osmosis and disinfection. With this, the water supply contains 5% of recycled water. There is no evidence of any risk due to this practice. |
| 1978 | UOSA Water Reclamation Plant | The Upper Occoqan Sewage Plant treats wastewater to feed the Occoqan basin that supplies one million people in North Virginia. The treatment includes primary sedimentation, secondary treatment, biological nitrification, lime clarification, recarbonation, filtration, activated carbon adsorption and disinfection. No negative effects have been observed from the indirect reclaiming in over 20 years. The amount of recycled water by extraction is usually from 10 to 15%, however during droughts it is 90%. |
| 1978 | Tahoe Sanitation Agent Water Reclamation Plant | Treatment plant for discharges into Truckeee River, supply source for Reno, Nevada. |
| 1981–1983 | Potomac | The Potomac estuary, the source of potable water supply, is fed with 50% recycled water. Treatment includes aeration, coagulation, clarification, predisinfection, activated carbon adsorption and post disinfection. |
| 1980 | US EPA evaluation of surface water quality | Study of unplanned reclaim, evaluating the quality of superficial water used as supply for a population of more than 25,000 inhabitants. Results show that 33% of the population drinks water containing from 5 to 100% of recycled water. |
| 1983 | Denver Demonstration Plant | Study that shows the feasibility of using a secondary effluent for direct supply on a pilot scale. After seven years, it was concluded that the process should include lime clarification, adsorption, reverse osmosis, air desorption, ozonation for a primary disinfection, secondary chloramine disinfection. The project cost 30 |

| Year | Place | Description |
|---|---|---|
| | | million USD by late 1993. |
| 1986 | Tampa Water Resource Recovery Project, Florida | The feasibility of reusing a denitrified secondary effluent, to be reused for human consumption after mixing with superficial water or aquifer water, was evaluated. |
| 1988 | Phoenix, Arizona Potable Reclaiming Feasibility Project | It was demonstrated that recycled water for human consumption is cost effective in comparison with other supply alternatives. |
| 1990 | Regulation from the California Department of Health Services | From the WF-21 results, a set of criteria was established, which was the first of its kind in the USA. The criteria apply both for infiltrated superficial water and for direct injection, and establish that:<br>▪ recycled water should not be more than 20% for direct injection and 50% for infiltration.<br>▪ water should remain at least 12 months for direct injection and 6 months for infiltration.<br>▪ TOC of recycled water should be < 2 mg/l for direct injection or < 3 mg/l for infiltration. |
| 1990–1995 | West Basin Water Recycling Program | Recharge project for controlling saline intrusion. |
| 1993 | City of Colorado Springs | Effluent is treated by microfiltration and reverse osmosis. There is a high degree of public acceptance for recycled water use. |
| 1994 | California Department of Health Services Conceptual Approval of the San Diego City Repurification Project | This project seeks to increase availability from reclaiming. Six criteria were established which must be met before reclaiming. |
| 1996 | The California Potable Reuse Committee proposes a framework for regulating the indirect potable reuse of advanced treated recycled water by surface augmentation | The committee was created in 1993 for the purpose of determining the feasibility and safety of reusing advanced treated recycled water for human consumption. After 2 years of examining relevant data it was concluded that it is possible to do so under certain conditions. |

Adapted from: McEwen (1998)

The California Committee for Recycled Water determined that aquifer recharge to obtain water for consumption should meet the following requirements:

- apply the best technology available for advanced treatment of wastewater and assure that plant operation complies with the design criteria;
- maintain appropriate retention times in natural storage, taking into account the reservoir's dynamic;
- achieve a reliable operation, so that the effluent consistently meets the microbiological, chemical and physical aspects of primary potabilization standard;
- comply with the established criteria for direct injection when trying to increase superficial water availability using recycled water via aquifer storage recovery;
- maintain the quality of water in reservoir;

- have an effective discharge control program.

**Belgium**

Van Houtte (2001) studied an aquifer recharge system with membrane treated water. An infiltration pond of 20 000 $m^2$ was used to recharge 2 500 000 $m^3$ of water per year (i.e. 0.01427 m/h or 342 $l/m^2$ per day). Recycled water moves inside the aquifer from 40 to 60 m to achieve a minimum residence time of 6 weeks, which in Belgium is considered hygienically safe. To treat infiltration water, membrane processes are used, which in a single step eliminate salts and nutrients. Microfiltration or ultrafiltration (in hollow or tubular fibres) was used as a pretreatment to osmosis. The objective was to produce water with the characteristics indicated in Table 3.35. Both the effluents from microfiltration and those from reverse osmosis were found to be free of total coliforms and faecal coliforms as well as streptococcus. Likewise, an important reduction of HPC of 2 log was obtained (Table 3.36).

**Table 3.35 Effluent quality from the Culpen treatment plant and comparison with infiltration standards**

| Parameter | Treated effluent | | | Infiltration standard |
|---|---|---|---|---|
| | **Mean** | **Min** | **Max** | |
| pH | 7.67 | 7.28 | 8.26 | > 6.5–<9.2 |
| Temperature (ºC) | 15 | 9.7 | 26.5 | 25 |
| Conductivity (dS/m) | 1.645 | 0.524 | 2.670 | 1.000 |
| Suspended solids (mg/l) | 5.9 | 0 | 19 | |
| Turbidity (NTU) | 2 | 0 | 11 | |
| Calcium (mg Ca/l) | 131 | 56 | 184 | TH < 40$^{o}$F |
| Potassium (mg K/l) | 34 | 2 | 100 | |
| Magnesium (mg mg/l) | 18 | 3 | 30 | 50 |
| Sodium (mg Na/l) | 210 | 60 | 379 | 150 |
| Total phosphorus (mg P/l) | 1.54 | 0.29 | 3.45 | 0.4 |
| Nitrate (mg $NO_3$/l) | 23 | 4 | 45 | 15 |
| Ammonia (mg $NH_4$/l) | 1.8 | 0 | 25 | 1.5 |
| Bicarbonate (mg $HCO_3$/l) | 367 | 110 | 543 | |
| Sulfate (mg $SO_4$/l) | 145 | 11 | 281 | 250 |
| Chloride (mg Cl/l) | 320 | 90 | 776 | 250 |
| UV-254 absorption ($cm^{-1}$) | 0.28 | 0.09 | 0.37 | |
| Total organic carbon (mg C/l) | 18 | 5 | 43 | |
| BOD (mg $O_2$/l) | 7 | 2 | 20 | |
| COD (mg $O_2$/l) | 50 | 10 | 144 | |
| Total coliforms (CFU/100 ml) | $10^5$ | | | |

BOD =biochemical oxygen demand; CFU = colony forming unit; COD = chemical oxygen demand; NTU = nephelometric turbidity unit; UV = ultraviolet

Adapted from: Van Houtte (2000)

**Table 3.36 Effluent water from Culpen plant and microfiltration quality**

| Parameter | Effluent | Filtered MF |
|---|---|---|
| pH | 7.6 | 7.7–7.8 |
| Total phosphorus (mg P/l) | 0.52 | 0.34–0.4 |
| UV 254 absorption ($cm^{-1}$) | 0.2686 | .2424–.2426 |
| TOC (mg C/l) | 24.2 | 18.4–18.5 |
| BOD (mg $O_2$/l) | 8 | 3 |
| COD (mg $O_2$/l) | 39 | 33 |
| Total coliforms (counts/ml) | $10^5$–$10^6$ | 0 |
| Faecal coliforms (counts/ml) | $10^5$ | 0 |
| Faecal Streptococcus (counts/ml) | $10^4$–$10^5$ | 0 |
| Log removal HPC 37ºC (48h) | $10^4$–$10^5$ | 2.8–3.2 |
| Log removal HPC 22ºC (72h) | $10^4$–$10^5$ | 2.5–3.2 |

BOD= biochemical oxygen demand; COD =carbon oxygen demand; HPC = heterotrophic plate counts at 22ºC and 37ºC; TOC = total organic carbon; UV = ultraviolet
Adapted from: Van Houtte (2001)

Estrogen activity was measured twice at the membrane outlet with the results shown in Table 3.37. At laboratory level, it was demonstrated that both nanofiltration and reverse osmosis membranes adsorb 17β-estradiol, the natural hormone with the greatest potential for endocrine disruption (Van Houtte, 2001).

**Table 3.37 Reduction of estrogen activity (expressed as µg/l 17 β estradiol) in a membrane processed secondary effluent**

| Sample | 28/8/1998 | 10/12/1998 |
|---|---|---|
| Effluent | 0.015 | 0.016 |
| MF-filtrate | 0.012 | <0.002 |
| RO-filtrate | 0.010 | <0.002 |

Adapted from: Van Houtte (2001)

**Windhoek, Namibia**

The Windhoek plant is the only case in the world of intentional reclamation of sewage for human consumption effected directly. It has a current capacity of 243 l/s. From the start of the project, society has been kept informed and, although it met with rejection at the beginning, today people accept the reuse. This project was developed on the basis of three elements considered vital for success:

- segregation of potentially toxic industrial water from the main discharge lines;
- secondary treatment of water to consistently produce an effluent with adequate quality; and
- advanced treatment to produce potable quality water.

The secondary treatment initially consisted of trickling filters followed by maturation ponds, with an initial retention time of 6 days that was later increased to 14 days. Subsequently, in 1979, 104 l/s were treated by activated sludges in order to diminish the concentration of ammonium. The effluent was still sent to maturation ponds and, in fact, at the present time, it is the only feeder to advanced treatment, because in 1993 its capacity was increased to 197 l/s. At this stage, the process was modified to include phosphorus removal by a biological process. The effluent has a carbon oxygen demand of 33–37 mg/l, and phosphorus is maintained between 6 and 8 mg/l.

The advanced treatment process has been continuously improved since its implementation in 1968 (Table 3.38). The main problem observed during the initial phase was a high content of ammonia nitrogen. Replacing hard detergents with soft detergents generated the first change in configuration, which eliminated foam fractionation. In the second phase, the use of lime precipitation in the ammonia desorption towers caused many operational problems due to fouling and scaling. It was then that activated sludge was put into use, and lime precipitation was replaced by aluminium flocculation (phase 3). In the fourth phase, there were sedimentation problems in the maturation ponds due to the presence of algae, and a dissolved air flotation system was implemented to deal with this problem.

In terms of monitoring, a wide range of samples was initially taken at different treatment points such as the storage tanks, distribution networks, and the users feeding points). The aim was to evaluate chemical quality and toxicity (lethality of daphnia, urease enzyme activity, bacteriological growth and inhibition), and to monitor viruses, somatic coliphages, bacteria, algae and mutagenesis (using the Ames tests). From 1968, this monitoring on water quality was complemented by performing studies on every patient visiting a hospital, clinic or physician’s office. Originally, the analyses were done by the City Water Department and the South African Medical Research Institute. Duplicate samples were analysed, but when it was shown that this was not necessary, they were analysed in just one laboratory. Monitoring represents 20% of the operating cost. Currently, water quality is monitored by three independent laboratories, in addition to continuous monitoring by the operator.

**Table 3.38 Configurations of the Windhoek (Namibia) reclaiming plants**

| Configuration # 1 (1969) | Configuration # 2 (1977) | Configuration # 3 (1980) | Configuration # 4 (1986) |
|---|---|---|---|
| Gammams Wastewater Treatment Plant | | | |
| Primary sedimentation | Primary sedimentation | Primary sedimentation | Primary sedimentation |
| Trickling filters | Trickling filters | Activated sludges$^{+}$ | Activated sludges$^{+}$ |
| Secondary sedimentation | Secondary sedimentation | Secondary sedimentation | Secondary sedimentation |
| Maturation ponds | Maturation ponds | Maturation ponds | Maturation ponds |
| Goreangab Reclaiming Plant | | | |
| Carbon dioxide$^{++}$ | Lime$^{+++}$ | Chlorine$^{+++}$ | Alumina |
| Alumina | Sedimentation | Alumina$^{+++}$ | Dissolved air flotation |
| Autoflotation | Ammonia desorption | Lime$^{+++}$ | Chlorination$^{*+++}$ |
| Foam fractionation | Primary carbon dioxide$^{++}$ | Sedimentation | Lime$^{+++}$ |
| Break point chlorination$^{++}$ | Chlorination$^{++}$ | Break point chlorination | Sedimentation |
| Sedimentation | Sedimentation | Chlorine contact | Sand filtration |
| Sand filtration | Secondary Carbon dioxide$^{+++}$ | Sedimentation | Break point chlorination$^{+++}$ |
| Carbon filtration | Sand filtration | Sand filtration | Chlorine contact |
| Chlorination$^{++}$ | Break point chlorination$^{+++}$ | Chlorination$^{++}$ | Activated carbon filtration |
| Mixing | Carbon filtration | Carbon filtration | Chlorination$^{++}$ |
| | Chlorination$^{++}$ | Chlorination$^{++}$ | Mixing |
| | Mixing | Mixing | |

Process modification; $^{++}$ chemical addition, $^{+++}$chlorine marked only by intermittent shock dosing. In 1995, aluminium was replaced with ferric chloride, activated carbon filtration relocated just after sand filtration, and take chlorination to break point before carbon treatment. Source: Haarhoff & Van der Merwe (1995)

To validate monitoring, a technical supervision committee was created, composed of experts from five independent professional associations, who met three times a year. Additionally, a group of 28 experts was meeting once a year, up until 1988, when it was no longer considered useful. At the present time, the committees meet whenever the need arises.

From 1976 to 1986, an epidemiological study was done with 75,000 to 100,000 people, which is not a sufficiently large sample size to have statistical significance. However, when the results were classified by race or population segment, they were consistent with other worldwide studies carried out by WHO, which were statistically significant. The lack of national statistics prevented validation of this information. It should be noted that Windhoek plant does not operate continuously, as it is only in operation when there is a water shortage. Dilution at the recycled water plant inlet, at storage and at distribution lines represents a ratio of 1:3.5.

## 3.6 Regulatory framework

Regarding regulations, there is a view that the criteria established for controlling potable water are not applicable to intentionally renovated water because potable water, by definition, is obtained from high-quality sources (meaning first-use sources, McGauhy, 1968), and therefore, should not be polluted. Thus, using reclaimed water for potable purposes is considered less desirable than using first-use water, because it may carry unknown contaminants. This is the case, even when reclaimed water complies with all physical, chemical, radiological, and microbiological parameters set in potable standards, and meets more stringent standards developed specifically for reclaimed water. This situation denotes a suspicion of these “new” sources.

In part, the reason for this is that reuse for drinkable purposes implies a contact between the population and the renovated water in a conscious way. There are both technical and psychological barriers against the use of recycled water, which make regulation of such water very difficult. This is the case even though it is well known that the demand for water is increasing day by day, that first use water is scarce, and that there are studies that show no negative effects in some of the actual cases of unintentional reuse of water, as well as in cases of intentional reuse.

The purpose of this section is to reflect on how to develop effective regulations to control reuse of water for human consumption, via either intentional or unintentional aquifer recharge. The intention is to clarify the main concepts as well as the differences in approach that must exist to deal with the specific issues of different regions or countries. The complexity of the hydrologic system has led scientists to study surface and underground systems separately. Combining both efforts is important to optimize the management of water in a watershed. Although the most important source of aquifer recharge is agriculture, cities should also be considered.

It is important to bear in mind that any possible definitive regulation for the direct or indirect, intentional or unintentional reuse of water for human consumption will take time. After all, the current regulatory framework for potable water took 80 years to be finalized (Sayre, 1988). Furthermore, we must be aware that today, because of the relevance of the topic and advances in analytical techniques, there is a tendency to include many parameters and set very strict limits. Perhaps as time goes by, and as recycling processes become more common, it will become evident that the previously imposed barriers were based on fear, excessive caution or simple ignorance, and there may be a deregulation in this field. In fact, because regulations for drinking-water are increasingly including more parameters and more restricted range limits, in the long run we may expect to see a convergence in the regulations for drinking-water and recycled water for human consumption. However, this scenario has not yet been considered.

### 3.6.1 First-use water and renovated water

The idea of setting standards for drinkable water based on the fact that the original (unused) water was good quality to begin with is becoming obsolete. The criteria and guidelines imposed on drinking-water are becoming more complex, even though the presence of renovated, but contaminated water, is not openly acknowledged. Surprisingly, the standards by which drinkable water is evaluated are not contested as much as the proposed standards for the regulation of recycled water. Apparently, the fact that renovated water is obtained from an engineering process for some reason diminishes its value when compared with naturally recycled water. It appears that, by definition, naturally recycled water is safer, even when both meet the same standards. The distinction goes even further, as it is assumed that the quality of renovated water can be improved or made acceptable by blending it with first-use water, regardless of the fact that the quality of the latter might be inferior (Sakaji & Funamizu, 1998). For example, a study compared recycled water from a pilot plant in Mission Valley, San Diego with the influent to a potabilization plant (Tables 3.39 and 3.40). The Health Advisory Committee for this project concluded that: “The health risk associated with the use of the Aqua II AWT water as raw water supply is less than or equal to that of the existing city raw water as represented by the water entering the Miramar water treatment plant”. A further conclusion was that, using currently available technology, it is possible to produce water that meets all the standard requirements for drinking-water.

There are other studies on renovated water showing that it meets all the standards applied to potable water (National Research Council, 1994). However, it is still feared that unidentified compounds may be present, and not measured. Nevertheless, this also occurs in normal supplies, as it has been estimated that 85% in weight of its organic components have not been identified and their effects are unknown (National Research Council, 1982; Eisenberg, Olivieri & Cooper, 1987; Cooper et al., 1992)

**Table 3.39 Comparison of metal content in influent water to Miramar plant serving San Diego, water produced by a water reclaiming plant, and comparison with mean values in aquifers and superficial waters in the USA**

| Element | MCL | Renovated water | Influent potabilization plant | Underground water | Superficial water |
|---|---|---|---|---|---|
| Metals, mg/l | | | | | |
| Arsenic | 0.005 | 0.0003 | 0.0017 | 0.001 to 0.100 | 0.001 to 0.02 |
| Cadmium | 0.010 | .00002 | .00004 | 0.001 to 0.058 | 0.001 to 0.44 |
| Chromium | 0.050 | 0.008 | 0.003 | 0.0008 to 0.050 | 0.0002 to 0.65 |
| Copper | na | 0.005 | 0.009 | 0.001 to 0.100 | 0.0004 to 0.200 |
| Lead | 0.050 | 0.0004 | 0.0005 | 0.002 to 0.100 | 0.0002 to 1.56 |
| Nickel | na | 0.008 | 0.003 | 0.001 to 0.130 | 0.0013 to 0.50 |
| Selenium | 0.01 | 0.0001 | 0.0007 | 0.001 to 0.061 | 0.0001 to 0.047 |
| Zinc | na | 0.025 | 0.017 | 0.0027 to 1.37 | 0.001 to 8.6 |
| Trihalomethanes, µg/l | | | | | |
| Bromoform | 100[a] | 0.004 | 0.972 | 0.100 to 43.3 | 0.133 to 27 |
| Bromodichloromethane | 100[a] | 0.008 | 3.79 | 0.1 to 11.50 | 0.1 to 57 |
| Chloroform | 100[a] | 0.130 | 2.78 | 0.267 to 19.98 | 0.267 to 198 |
| Dibromochloromethane | 100[a] | 0.017 | 3.15 | 0.1 to 19.73 | 0.1 to 92 |
| Volatile organics | | | | | |
| Benzene | 1.0 | 0.048 | 0.054 | 0.2 to 11.0 | 0.23 to 0.47 |
| 1,1 dichloroethane | 6.0 | 0.004 | 0.004 | 0.28 to 17.7 | 7.2 |
| Methylene Chloride | na | 11.62 | 9.55 | 0.1 to 72 | 0.367 to 1.067 |
| Tetrachloroethane | 5.0 | 0.006 | 0.009 | 0.1 to 106 | 0.1 to 2.03 |
| Trichlorofluoromethane | 150 | 0.857 | 0.400 | 0.3 to 4.2 | na |
| Toluene | na | 0.099 | 0.042 | 0.1 to 8.0 | 0.2 to 21 |
| Xylene | 1750 | 0.015 | 0.003 | 0.2 to 7.58 | 0.3 to 400 |
| Extractable organics, µg/l | | | | | |
| Benzo-a-pyrene | na | 0.023 | 0.037 | 0.001 to 0.01 | 0.001 to 0.01 |
| Butylbenzyl phthalate | na | 0.166 | 0.152 | 38 | 0.034 to 0.59 |
| Dibutyl phthalate | na | 0.994 | 0.912 | 470 | 0.79 to 0.09 |
| Di-n-octyl-phthalate | na | 1.395 | 0.705 | 2.4 | 0.024 to 0.477 |
| Bis (2ethylhexyl) phthalate | 4.0 | 8.05 | 4.65 | 0.4 to 12.36 | |

MCL = maximum contaminant level; na = not available
[a] Total limit for trihalomethanes
Adapted from: Olivieri, Eisenberg & Cooper (1998)

**Table 3.40 Concentration of compounds considered hazardous in the influent to a potabilization plant (Miramar, San Diego, California) and in renovated water obtained from wastewater**

| Compound | Renovated water (µg/l) | Influent to Miramar potabilization plant (µg/l) | MCL (µg/l) |
|---|---|---|---|
| Organics | | | |
| Benzyl butyl phthalate | < 0.300 | 0.153 | na |
| Benzoic acid | < 4.0111 | 0.114 | na |
| Bis (2-ethylhexyl)-phthalate | < 0.048 to 7.584 | < 0.045 | 4 |
| Bromodichloromethane | < 0.04 | 3.726 | 100 |
| Bromoform | < 0.06 | 0.957 | 100 |
| Chloroform | < 0.03 | 2.194 | 100 |
| Dibromochloromethane | < 0.05 | 3.153 | 100 |
| Toluene | 0.101 | 0.042 | 100 |
| Trichlorofluoromethane | < 0.07 | 0.389 | na |
| Metals | | | |
| Aluminium | 63 | 206 | 1000 |
| Arsenic | < 2 | 1.542 | 50 |
| Barium | < 24 | 77 | na |
| Boron | 195 | < 25 | na |
| Calcium | 5082 | 45740 | na |
| Iron | 13 | 43 | na |
| Magnesium | 1749 | 19010 | na |
| Manganese | 9 | 9 | na |
| Phosphorus | 738 | 1293 | na |
| Potassium | < 1255 | 2878 | na |
| Sodium | 23540 | 69040 | na |
| Silicon | 909 | 4386 | na |
| Strontium | 46 | 67.4 | na |

MCL = maximum contaminant level; na = not available.
Adapted from: Olivieri et al. (1998)

Based on the available information, there are no indications that the health risk from using renovated water that has been treated intensively for human consumption is higher than from water originating in normal supply sources, or that the concentration of regulated compounds and organisms in renovated water exceeds the standards of drinkable water. Nonetheless, the reuse of water for human consumption is questioned and establishing criteria for this use is difficult.

### 3.6.2 Some aspects to be considered in developing standards for water for human consumption

As a basic premise for the establishment of at least local or national criteria and standards, on a country or regional level, we should bear in mind that the selected parameters and controls should not imply greater risk than that caused by existing usage. It must be understood that, in regions where water is scarce or poor quality (polluted or "renovated" water, whichever the case may be), criteria may be established that might not be accepted in other regions. An overall objective should be for indirect potable reuse water projects to provide the same degree of safety as current supplies (or at least improve the present situation). In the case of renovated water, safety is defined in terms of the risk acceptable to the population who will use it, and who are willing to pay, in every sense (see "Stockholm framework" discussion in Appendix A). Risk is established by governmental officials who are responsible for:

- keeping the population informed about renovated and nonrenovated water conditions, particularly as regards quality;
- protecting health;
- imposing feasible and viable legislation in a social, economic and political context (not just a technical context).

The regulatory framework for consumption of recycled water refers not only to defining a set of parameters and values, but also to:

- water processing;
- reliability of systems to produce water for human consumption;
- recharge conditions;
- size and characteristics of the aquifer relative to recharge;
- extraction and potabilization conditions;
- type of routine and surveillance monitoring;
- the points shown in Table 3.41.

### 3.6.3 Regulatory options

In countries where water reuse is not practiced, recommendations or compulsory standards for the intentional or unintentional use of recycled water are established to promote future reuse. However, in countries where use of recycled water is already established, the objective is to regulate such use. In both cases, one or more of the following aspects may be defined:

- level of existing or admissible risk (see "Stockholm framework" discussion in Appendix A);
- renovated water quality;
- monitoring;
- type and reliability of treatment;
- area of application.

**Table 3.41 Aspects to consider when setting a regulation of recycled water for human consumption via aquifer recharge**

| Aspects | Advantages | Disadvantages |
|---|---|---|
| Include treatment criteria. | Less need for monitoring and surveillance<br>Project implementation is easier. | Limits technological development and may encourage bias in regulator who will be responsible for both selecting the controlling method and meeting objectives. May lead to nonviable schemes from an economic point of view. |
| Select and use the "best" indicator organisms, or a set of them. | Introduces the "ideal" in the definition of a good effluent. | Reduces monitoring and control costs, provided redundant microorganisms are not used (for instance total and faecal).<br>May give a false impression of safety. |
| Select monitoring parameters and establish limits for each one. | Facilitates supervision. | Cannot be universal or static in time.<br>Increase supervision costs. |
| Define sampling and monitoring frequency for compliance data points | Makes administrative supervision easier. | Maybe complicated if not doing properly from a practical point of view. |
| Take into account epidemiological research. | Recognizes information obtained in different parts of the world, in humans.<br>Makes legislation unnecessarily stringent | Not accepted in developed countries. |
| Use of toxicological tests | Helps to establish cause–effect relationship. | Does not emulate real conditions of use of renovated water.<br>Makes legislation too stringent. |
| Use of risk evaluation models to determine possible health hazards and water quality requirements. | Helps government make rational decisions. | Hard to explain to the population. |

Establishing water quality or determining water processes to be applied may be independent or combined. The problem with establishment of quality is that it is impossible to define appropriate indicators for all kinds of contaminants; therefore, there is a risk that certain pathogens or compounds may not be detected with this approach. On the other hand, determination of the treatment indirectly establishes a cost and limits technological development. There are two approaches for establishing standards: consider that water must be completely treated to meet the requirements for human consumption, before it is sent to the aquifer; and consider that during infiltration and storage, water quality improves, and recycled water may be subjected to additional treatment, in order to comply with potabilization criteria.

### 3.6.4 Toxicology versus epidemiology: establishing acceptable risk and existing risk

*Usefulness of toxicology*

To evaluate the health effects of renovated water, it is believed that the nonvolatile organic material should be concentrated up to 1000 times, and the effect should be established of exposing a 70 kg human consuming 2 l/day from a single source, to the following experiments (McEwen, 1998):

- mutagenicity — using Ames *Salmonella* reverse mutation assay;
- genotoxicity — sister chromatid exchange test and micronuclei essay in mouse splenocytes;
- subchronic toxicity — 90-day subchronic toxicity assay in mice and rats;
- carcinogenicity — SENCAR mouse skin initiation–promotion essay and strain A mouse lung adenoma assay;
- reproductive effects — two-generation reproductive toxicity assay in mice;
- teratogencity: developmental toxicity in rats (for microbiological tests, an infective dose (see Section 3.2) is determined, also assuming a consumption of 2 l/day).

Despite the meticulousness of this protocol, as discussed in Section 3.3.3, these tests do not reflect the situation in everydaylife. While toxicological studies are useful for establishing cause and effect and predicting effects, they do not identify the real risk level.

*Usefulness of epidemiology*

Epidemiology is a science that studies exposure to certain factors and their relation with the occurrence of illness in a human population. Environmental epidemiology, in particular, is concerned with establishing a relationship between a risk factor and a health effect; however, the fact that the populations studied are exposed to many environmental aspects makes it impossible to eliminate interference from factors that also affect public health, such as food, beverages, personal care products and air pollution.

There are several kinds of epidemiological studies. Ecological studies are the easiest to do, they are very common, and have generated a great deal of information, but they lack precision due to the difficulty of obtaining a useful sample size. Other types of study from which to choose are cross-section, case control and cohort studies; because they use a smaller sample size, they are useful in the study of renovated water. The choice is based on the approach of the study, that is to say, whether we want to know the potential cause of a given illness (retrospective) or, on the contrary, we want to determine future effects (prospective) from sampling the environment (Sakaji & Funamizu, 1998).

The information obtained from retrospective studies may be limited by the number of people exposed to the causal agent, or by the possibility of defining a control group that is not influenced by said agent. In prospective studies, it is easier to obtain the required information because the study population is established from the beginning, provided sufficient people are identified who are able to participate throughout the lengthy period of the research. This can be a problem due to immigration into and emigration from the study area. With these types of studies, although the morbidity and mortality rate are insufficient to establish cause–effect relation, they do provide information useful to establish the health and death status of a population (current risk). Sensitivity of the studies can be improved if the segment of population analysed is at greater risk.

The public health implications of using recycled water for potable purposes have been evaluated in numerous reclaiming projects in the USA, and no effects have been observed in the exposed populations. However, based on the size and types of studies (mainly ecological), the US National

Research Council (1994) considers that they are lacking in precision. For this reason, the council affirms that "in the absence of definitive epidemiological information, it is necessary to define the effects by means of risk studies" (Olivieri, Eisenberg & Cooper, 1998).

Historically speaking, acceptance of water quality from a microbiological point of view was based on epidemiological evidence demonstrating that treatment has a significant impact on the transmission of diseases such as cholera and typhoid fever. Because the risk of waterborne infections is now so low in developed countries, epidemiology are no longer considered applicable, and risk studies must be used. However, this is not in the case in developing countries, where epidemiological studies are still useful, even in cases of nonintentional reuse.

### 3.6.5 Consideration for potable reuse

Some of the factors to be considered when recharging an aquifer for the purpose of human consumption are discussed below.

*Discharge pretreatment*

When recharging aquifers for human consumption it is important to develop efficient pretreatment programs for industrial discharges into the sewerage, so that effluents have relatively "controlled" characteristics. Although this is not part of recharge legislation, it is definitely an essential component. The presence of industrial discharges into the sewer system is a concern, because they carry compounds that are hard to determine and remove, and that have unpredictable and even unknown effects, so they must be segregated from the water before infiltration. Because there is reuse of treated wastewater for human consumption, regardless of whether it is intentional or unintentional, the discharge of toxic compounds must be regulated so that only domestic water is used.

*Pre-infiltration treatment*

The effects of toxic elements and pathogens that may be present in recycled water have not been thoroughly studied or characterized; however, there is an increasing certainty about the technological capacity to remove them. Costs much be balanced against the need for recharge. In the event that a treatment before infiltration is required, it is advisable to allow for the introduction of new processes, so as not to limit technological development. As regards treatment itself, it is advisable to establish critical operating requirements and combine them with certain parameters of treated water quality, to limit monitoring.

*Soil as a treatment method*

Infiltration treats water at the unsaturated zone. This process is described in detail in Chapter X.

*Multiple barrier and redundancy approach*

Wastewater and treated water contain a large amount of inorganic and microbiological compounds, most of which can be detected, identified and quantified. Available technology can eliminate most of these contaminants to produce water that is cleaner than clean water (i.e. "first use water"). However, the fear of unknown organic components, the difficulty in assessing the risks from long-term exposure and the flaws in human systems leads to a preference for treatment systems based on diverse barriers, which ensure the reliability of the renovated water (i.e. a multiple-barrier approach). Among the requirements related to process reliability, the most important are the implementation of continuous controls for certain parameters, the installation of alarms and automation, the availability of replacement parts, redundancy of processes and equipment, the existence of automated standby equipment in case of failure, the existence of a stock of reactants (especially disinfectants) and the installation of self-powered equipment or duplicate power supplies (Mujeriego, 1998).

*Additional barriers*

In addition to dilution, the retention time in the aquifer is a natural treatment for water. Frequently, in planned indirect reuse, a natural treatment is seen as a redundant, additional system that complements the efficiency of engineered treating systems. Therefore, regulations may be set for mixing and dilution, retention time, and distance to the extraction site. Each of these factors is described below.

**Mixing and dilution**

The mixing ratio is undoubtedly a topic that must be defined in each case. For example, in California, it is stipulated that it should not exceed 50%. When reuse is incidental, the extraction permit process in some way constitutes a limit on reuse, established by the assimilation capacity of the receiving body.

**Retention time**

Underground water moves very slowly and depends on the transmissiveness of the aquifer materials as well as the hydraulic gradient. This movement translates into a level of dilution that varies for each case. The time that water remains in the aquifer represents the time required for a natural treatment to refine water quality. During this time, monitoring can make it possible to react in the event that a corrective measure should be needed. This separation also provides the psychological comfort of knowing that there is a natural barrier. Literature indicates retention times of up to 6 months when using infiltration and up to 12 months for direct injection (in California), but there are other examples such as the Fred Hervey Plant in El Paso, Texas (see Section X), where the retention time in the Bolson del Hueco aquifers is of 2 years.

**Distance to the extraction site**

From a practical point of view, retention time implies a geographical separation between the point where water is introduced into the aquifer and the site where it is extracted (which is easier to measure and observe). Therefore, it is preferable to set up a point a minimal distance from the extraction site. As regards the site of compliance, standards for renovated water should satisfy quality criteria at the point of extraction, based on the risk of microorganisms regrowing during transportation and storage, or in the site of recharge. However, if standards are properly complied with, there is little possibility of microbial growth, and viruses (the main microbiological risk) do not reproduce in the absence of a host. Furthermore, any modification would be similar to that which occurs during the transportation of potable water.

*Other parameters*

Other parameters that must be considered when recharging an aquifer for the purpose of human consumption are microbiological factors, toxicity and salts.

**Microbiological**

Once the relevant pathogens are defined, and indicators and appropriate analytical techniques have been set, there is still the problem of establishing the acceptable level for reuse. Ideally, these standards should be based on awareness of the relationship between levels of indicators and pathogens, as well as infectious doses. Unfortunately, it is not feasible to establish absolute universal relationships or values. Thus, in practice, most of the regulations applied to reuse water are based on faecal coliforms, even when their limitations are evident (Feachem et al., 1983; Cooper & Olivieri, 1998).

**Toxicity**

Foster et al. (2002) state that parameters that may serve to regulate treated wastewater recharge are soluble organic carbon (as a measure of potential toxic components), nitrogen and faecal coliforms. These three elements are an inexpensive way to control recharge.

**Salts**

Increase in salinity is a long-term problem with water reuse, and it should always be kept in mind, because it may lead to the establishment of stricter treatment criteria on a local level.

*Monitoring water quality*

Determining how to monitor reused water for human consumption is a very important aspect of the drafting of criteria and standards. It requires definition of parameters of water quality, numerical limits, monitoring frequency and compliance site. It is impossible to monitor each and every toxic chemical compound and pathogen. Therefore, it is necessary to select parameters Monitoring programs for recycled projects should constantly verify the efficiency of the treatment processes and detect potentially harmful contaminants; for example, by biomonitoring. A conservative approach should be taken to intensive monitoring of water quality, and contingency plans to respond to possible failures. Qualitative parameters such as turbidity or solids, and viruses, are of interest, and are described below.

**Turbidity or solids**

It is a common practice to establish turbidity or solid content as a control parameter, because the presence of particles prevents effective disinfection. Some particles may contain organic material and, therefore, promote the generation of organochlorinated compounds. A value of less than 2 nephelometric turbidity units or 5 mg/l of total suspended solids is considered appropriate in recycled water, as this permits effective disinfection. As regards the frequency of measurement, turbidity is usually monitored continuously because it is easy to do so, while solids are monitored by composite samples for the day.

**Viruses**

Although there is no agreement on a standard for viruses, the tendency is to consider the need to set it as a value within the rules for reuse water for human consumption. However, it has not yet been possible to make any rule or proposal on the matter.

### 3.6.6 Public information

An additional problem for risk administrators, regarding water recycling and reuse for human consumption, is how to communicate their decisions to the public. It is undoubtedly difficult to tell the public that there is no zero-risk policy, and that whatever has been decided is reasonable, prudent and feasible given the economic and technical conditions of the country or region. Additionally, public policy evolves more slowly than knowledge and generation of technical information; thus, politicians frequently confront specialists who enjoy greater social credibility. For Therefore, it is important to promote problem-solving research and public debate.

Finally, public confidence in the use of recycled water must be gained through education. Confidence must be built in practice, as well as politically; and those in charge must have the necessary ability to communicate effectively (to both talk and listen). All of this must take place within a framework where communication flows freely, thus making it certain that the public and the specialists have a more complete outlook on the management of water resources. This also serves to make them actively participate in programs, such as the control of hazardous waste discharges into the sewer system (Mujeriego, 1998).

### 3.6.7 Examples of legislation

For historical and social reasons more than technical reasons, it is very difficult to establish a single framework of criteria on an international level. This is also true in the case of a single country, such as the USA.

The reuse of water in the USA for all kinds of purposes was estimated as 5.7 X$10^6$ $m^3$/day (66 $m^3$/s) in 1990 (Crook, 1998), and of this amount, 7.2 $m^3$/s are obtained by aquifer recharge (Pyne, 1994).

The first reuse standards in the USA were established in 1918, in California, a state with little water and many economic resources. These standards were directed toward the control of wastewater in agriculture ("sewage farms"), which has been practiced with low-quality water in certain areas of the country since the late 1800s. In time, water reuse increased and applications diversified, while treatment processes and analytical techniques became more sophisticated, and reuse criteria evolved in parallel. In the beginning, the decision to reuse water was motivated by opportunity, convenience and economic criteria, with greater emphasis on the economic aspects. Thus, the few existing regulations from the 1960s and 1970s were written with a conservative focus, regulating only nonpotable reuse, and reflecting the state of the art at that time. Little attention was given to regulation of storage in basins for later use.

At least 22 of the 50 states of the USA have standards and criteria for diverse reuse practices (Cotruvo, 2001). California has proposed criteria for aquifer recharge, while Florida has developed recharge criteria to increase water availability. Some states have used criteria developed by the US EPA as a basis for the recharge of aquifers, but others have simply prohibited reuse for potable consumption. In some states, treatment via the soil has been controlled, with a view toward purifying the water more than reusing it, despite the fact that the objective of soil aquifer treatment was frequently recharge. Thus, there are no federal regulations on the reuse of water for potable reuse via aquifer recharge.

With the population growth in the second half of the 20th century, and increasing demand, first-use water became very expensive. The 1960s were the start of an era with many demonstration projects, initiated to rationalize regulations or encompass different types of reuse, including human consumption. Different states established their own standards and their own demonstraton projects, duplicating information, and huge data banks were created. More specific examples from USA are described below.

*California*

Since issuing the first regulation for reuse of water in 1918, regulations in California have been constantly modified. During the 1990s, reuse criteria were proposed (including human consumption), reflecting a serious concern for both short and long-term health effects (State of California, 1992). These standards are based on a combination of controls to achieve maximum microbiological and chemical safety. The proposal included the control of sources, type of wastewater treatment, recycled quality standards, recharge methods, recharge area, volume of recharge water, soil characteristics, distance between recharge and extraction site, residence time in the aquifer and requirements for installation of monitoring wells.

The California criteria were based on demonstration studies of recharge by infiltration and direct reinjection for potable purposes. They were established to ensure that aquifer water complied with the criteria for human consumption, and did not require additional treatment before distribution. A summary of these criteria is shown in Table 3.42.

The criterion for indirect drinkable reuse refers to aquifer recharge with domestic water via superficial infiltration. It is established that water must be treated before infiltration, and should always have the quality needed to completely protect health. Recycled water must comply with the recommendations of the Department of Health Services (DHS), which has the authority and responsibility for establishing water quality standards.

**Table 3.42 Proposed criteria for aquifer recharge with recycled water (State of California, 1993)**

| Treat and recharge site requirements | Project category | | | |
|---|---|---|---|---|
| | Surface spreading | | | Direct injection |
| | I | II | II | IV |
| Level of wastewater treatment | | | | |
| Primary/secondary | X | X | X | X |
| Filtration | X | X | X | X |
| Organics removal | X | X | | X |
| Disinfection | X | | | X |
| Maximal recycled wastewater in extracted well water (%) | 50 | 20 | 20 | 50 |
| Depth to groundwater at initial percolation rate, m | | | | |
| 50 mm/min | 1 | 2 | 2 | na |
| 80 mm/min | 2 | 2 | 5 | na |
| Retention time underground (months) | 6 | 6 | 12 | 12 |
| Horizontal separation[a] (m) | 50 | 50 | 100 | 100 |

na = not applicable; X = a treatment process is required.
[a] From the edge of the groundwater operation to nearest potable water supply well.

There are strict criteria for microbiological and chemical constituents. Concentrations of minerals, inorganic matter (with the exception of nitrogen compounds) and organic matter in renovated water before recharge must not exceed the maximum permissible limits in potable water regulations. Total content of nitrogen must be less than 10 mg/l unless it is demonstrated that a different value does not alter water at the point where it reaches the aquifer level. Regulation considers a value for total organic carbon (TOC) of 1 mg/l, as a way of regulating organics not contemplated within the standard. This value represents the treatment level required so that the extraction well does not exceed 1 mg TOC/l as a measure of organic contaminants that cannot be regulated (Asano et al., 1992). Values in the water effluent to be infiltrated correspond to those shown in Table 3.43.

Based on the information generated in California it is possible to expect a chlorinated tertiary effluent to produce a virus-free effluent 99% of the time (i.e. the limit could be exceeded three or four times a year). Although there is no maximum virus value, the California criteria are designed to ensure that for any kind of recharge method, wastewater type and retention time, there is a virus removal or inactivation of 20 log. The maximum published virus concentration in wastewater is $10^6$ (Buras, 1976). This is the result of combining the treatment method, the passage of water through the unsaturated zone and underground storage, thereby obtaining a “negligible risk” (Asano et al., 1992). In spite of this, it is believed that more investigation is required to establish the reliability of the reuse programs (Asano & Mjueriego, 1998).

**Table 3.43 Maximum permissible concentration of organics in effluents to be infiltrated, as per the Criteria Proposal for Recharge in California, USA**

| Renovated water percentage in the aquifer | Maximum concentration of organic carbon in treated water (mg TOC/l) | |
|---|---|---|
| | Infiltration | Direct injection |
| 0–20 | 20 | 5 |
| 21–25 | 16 | 4 |
| 26–30 | 12 | 3 |
| 31–35 | 10 | 3 |
| 36–45 | 8 | 2 |
| 46–50 | 6 | 2 |

TOC = total organic carbon

For infiltration projects, maximum percolation values and the distance to the aquifer are established, to ensure adequate removal of organic components and pathogens. The distance to the aquifer is defined in order to maintain an aerobic zone allowing biological degradation. The required depth of the unsaturated zone is 3–15 m, depending on the site and water pretreatment. Furthermore, the percolation rate should be less than 8 cm/min in order to obtain the soil benefits. If the initial rate is less than 5 cm/min, then lower distances to the aquifer are allowed. Maximum percolation rate is determined in the field, at the beginning of the study, before starting the recharge project. A maturation or equilibrium rate is not used due to the difficulty in assessing compliance therewith.

The DHS has established that renovated water at the extraction site should not exceed 20–50%, with the exact value depending on each project. The specified dilution should be met at all extraction sites. The total amount of injected water is determined annually, and the proportion is calculated at the potable water well that has the least dilution.

The criteria proposed by the DHS also define standard treatments and efficiencies for aquifer recharge projects (Table 3.44). The object of this is to provide sufficient guidelines for all types of recharge projects and thus reduce case-by-case assessment. Site requirements in the draft of Title 22, Section 5.1 establish that no more than 50% of water extracted from a well may come from renovated water. Furthermore, a minimum retention time of 6 months is proposed, in combination with a horizontal separation of 152 m between the infiltration area and the extraction well. The minimum distance to the aquifer is also specified for an initial fixed percolation rate; for example, for 5 mm/min or more, with a minimum depth of 6 m for a 50% recharge of renovated water.

**Table 3.44 Recharge requirements proposed by the DHS, Title 22, Section 5.1[a]**

| **Project category** | | **Infiltration** | | | **Direct injection** |
|---|---|---|---|---|---|
| | | **I** | **II** | **III** | **IV** |
| Maximum content of renovated water in a potable water well | | 50% | 20% | 20% | 50% |
| Minimum depth (m) to aquifer at a specific initial infiltration rate of: | | | | | |
| | < 0.5 cm/min | 3 | 3 | 3 | na |
| | > 0.8 cm/min | 6 | 6 | 15 | na |
| Retention time in the aquifer in months | | 6 | 6 | 12 | 12 |
| Horizontal[b] distance (m) | | 152 | 152 | 305 | 305 |
| Required treatment: | | | | | |
| | Secondary | X | X | | X |
| | Filtration | X | X | X | X |
| | Disinfection | X | X | X | X |
| | Organics removal | X | | | X |

a Criteria proposed for aquifer discharge in March 1993, California DHS, California Code Regulations, Title 22.
b Minimum distance between recharging infrastructure and extraction site.
na: Not applicable
Source: Crook (1998)

The DHS concern for possible long-term effects caused by the presence of organics in the recycled water led to the establishment of a limit for TOC of 1mg/l in the well closest to the infiltration site. This value limits the maximum amount of renovated water that may be infiltrated in relation to first-use water (native water), and also sets a maximum value for TOC in recharge water. A reduction in TOC has been observed even in saturated soils, in addition to the decrease occurring in unsaturated soil (Mills et al., 1998)

To obtain a permit, a study must be presented with all supporting data. This includes a description of the project and its impact, the initial percolation ratio, the maximum amount of injected water, the manner in which it will be done, and the annual minimal retention time. In the case of reuse for consumption via the increase in superficial water there are no criteria and projects are analysed on a case-by-case basis. In this instance, two state permits are required.

**Orange County, California, USA**

The Orange County project has demonstrated that aquifer recharge for purposes of human consumption and control of saline intrusion can be carried out safely. In this region it is foreseen that criteria for aquifer recharge will be a combination of state, DHS and Regional Board criteria, along with specific criteria established for that region. A comparison of these criteria is shown in Table 3.44. DHS criteria were developed with the objective of protecting public health, while the Regional Board criteria were designed to prevent degradation of the watershed and protect the beneficial use of superficial water and aquifers. For approval of each water reuse project, these criteria are applied based on a case-by-case analysis. DHS regulations define treatment needs, as shown in Table 3.45.

**Table 3.45 Legal requirements for aquifer recharge in California**

| **Constituents** | **DHS Title 22 reclamation criteria** | **Regional board groundwater basin objectives**[a] | **Regional board groundwater basin objectives plus mineral increment**[b] |
|---|---|---|---|
| Turbidity (NTU) | 2 | na | na |
| Microbiology, TC (MPN/100 ml) | < 2.2 | na | na |
| Sodium (mg/l) | na | 60 | 130 |
| Chlorides (mg/l) | 250[c] | 65 | 130 |
| Sulfates (mg/l) | 250[c] | 120 | 160 |
| TDS (mg/l) | 500[c] | 600 | 600 |
| TOC (mg/l) | 2 to 20[d] | na | na |
| Hardness | na | 290 | 320 |
| Nitrate-nitrogen (mg/l) | 10 | 3 | 3 |

na = not applicable; NTU = nephelometric turbidity units; TC = total coliforms; TDS = total dissolved solids; TOC = total organic carbon.
[a] Groundwater quality objectives set based on the Water Quality Control Plan, Santa Ana River Basin in 1994, by the Regional Water Control Board, to *"ensure the reasonable protection of beneficial uses and the prevention of nuisance within a specific area"*
[b] Water Quality Control Plan
[c] Potable water standards of the California Department of Health Services (DHS), maximum recommendable level.
[d] Maximum allowable TOC concentration based on percent contribution of recycled water.
Adapted from: Mills et al. (1998)

**Florida**

In the late 1970s the main motive behind the implementation of reuse projects in Florida was the disposal of effluents and this led to regulations controlling the application of wastewater into the soil (Table 3.46).

The regulations for indirect potable reuse contain sections that refer to rapid infiltration tanks, as well as absorption fields, which in both cases may result in aquifer recharge. Since almost all the underground water in Florida is classified as G-II (i.e. water with 10 000 mg/l or less of TDS), and is a source for human consumption, any land treatment system located above the aquifer is an unintentional water recovery system for human consumption. In absorption fields, a limit of 10 mg/l of TSS has been established in order to prevent fouling. If more than 50% of infiltrated water is collected, the system is considered a disposal system, and not for beneficial reuse. In these systems, the charge is limited to 23 cm/day, and dry and wet cycles must be used. For systems with higher charges, or those that are more directly connected to an aquifer than normal, renovated water must receive secondary treatment, filtration and high-level disinfection, as well as comply with the primary and secondary standards for potable water. The requirements are similar to those in California as regards the reuse of water via infiltration.

**Table 3.46 Reuse criteria in Florida applied to aquifer recharge**

| Type of use | Water quality limits | Recommended treatment |
|---|---|---|
| Rapid infiltration basins, absorption fields | 200 faecal coliforms/100 ml<br>20 mg/l of TSS<br>20 mg/l of BOD<br>12 mg/l of $NO_3$ (as N) | Secondary and disinfection |
| Rapid infiltration basins in unfavorable geohydrologic conditions | Non detectable faecal coliforms/100 ml<br>5 mg/l of TSS<br>Primary and secondary drinking-water standards. | Secondary, filtration and disinfection |
| Injection to groundwater | Non detectable faecal coliforms/100 ml<br>5 mg/l of TSS<br>Primary and secondary drinking-water standards | Secondary, filtration and disinfection |
| Injection to form of Florida or Biscayne aquifers having TSD than 500 mg/l | Non detectable faecal coliforms/100 ml<br>5 mg/l of TSS<br>5mg/l of TOC<br>0.2 mg/l of TOX<br>Primary and secondary drinking-water standards | Secondary, filtration, disinfection, and activated carbon absorption |

BOD = biochemical oxygen demand; TDS = total dissolved solids; TOC = total organic carbon; TOX = total halogenated organic compounds; TSS = total suspended solids.
Adapted from: Crook (1998)

The Florida regulations also include criteria, which address indirect potable reuse via injection into aquifers, in order to increase availability. This regulation takes into account aquifers classified as G-I, G-II and F-I, all of which are for human consumption. In this case, secondary treatment, filtration and high disinfection levels are required, as is class I reliability. Renovated water must comply with G-II standards before injection. Standards for G-II aquifers are mostly those established as primary and secondary for potable water.

When water is injected in formations in Florida or Biscayne, where dissolved solids are less than 500 mg/l, the rules are stricter and establish that recycled water must comply with potabilization standards and so forth, in order to be subject to an additional treatment with activated carbon, in order to remove TOC to 5 mg/l and TOX to 0.2 mg/l. Before this, an infiltration test program must be done with a minimum two-year duration.

Chapter 62-600 of the Florida Administrative Code, Reuse of Reclaimed Water and Land Application (FDEP, 1996), specifically establishes that renovated water may be used to increase the availability of class I surface waters, which are those used for human consumption. Renovated water must meet primary and secondary standards and have no more than 10 mg/l of nitrogen. Water treatment should have a class I reliability level and high levels of disinfection. The injection site of treated water should not be found less than 150 m from the intake site. Since 1997, regulations have been in the process of being drafted for indirect potable reuse of water, storage in aquifers and recovery of saline aquifers via barrier formation.

**Arizona, USA**

Indirect reuse for human consumption is regulated by an agency that grants permits for each recharge project. In each case, it must be shown that the process does not degrade the aquifer quality at compliance sites established for that purpose. In the event that the aquifer is already degraded, it must be shown that recharging does not further damage the quality. All Arizona aquifers are classified for human consumption and the state has adopted the National Primary Standards of maximum contamination levels (MCLs). These standards are applicable to all aquifers producing more than 20 l/day. Consequently, every recharge project must comply with the potabilization criteria at the injection site.

An aquifer recharge project using treated water must also have a storage permit. To obtain this permit the following must be demonstrated:

- the project contractor has the technical and financial capacity to build and operate a recharging project;
- the maximum treated water volume that will be stored at any given time;
- that storage will not cause irrational damage to the land or to the users of water in the zone impacted by the project, at any time while the permit is in force;
- the person responsible for the project requested and obtained a floodplain permit from the corresponding county agency;
- the person responsible has a permit for recharging the aquifer.

Storage permits establish the design and operation capacity of the recharging project, the annual maximum volume of renovated water that can be stored and the monitoring requirements. They also define conditions for monitoring the project and assessing its impact on the aquifer, soil and users of underground water in the project's area of influence. Before recovering recharged water, a permit must be obtained. The agency responsible for granting permits must define whether or not said exploitation irrationally alters the quality of the water or soil, or affects other users. These permits specify maximum usable amount.

#### US EPA's proposal for reclaimation criteria

These criteria sought to promote reuse in areas of the USA where it was not yet being done, and to eliminate inconsistencies among different state regulations (Table 3.47). These criteria must be combined with flexible state regulations that take into account the specific characteristics of each region. The criteria were developed for all types of reuse, and include recommendations on treatment, quality limits, monitoring frequency, distances for water recovery, and other controls.

**Table 3.47 US EPA proposed criteria for reusing municipal water (aquifer recharge as advisable)**

| Type of reuse | Treatment | Recycled water quality | Recycled water monitoring | Setback distances | Comments |
|---|---|---|---|---|---|
| Groundwater recharge<br>by spreading or injection into nonpotable aquifers. | Site-specific and use dependent.<br>Primary minimun for spreading.<br>Secondary[a] minimun for injection. | Site-specific and use dependent. | Depends on treatment and use. | Site-specific. | Facility should be designed to ensure that no recycled water reaches potable water supply aquifers.<br>For injection projects, filtration and disinfection may be needed to prevent clogging.<br>Provide treatment reliability. |
| Indirect potable reuse<br>Groundwater recharge by spreading into potable aquifers | Site-specific<br>Secondary[a] and disinfection[c] minimum<br>May also need filtration and /or advanced wastewater treatment[d] | Depending on site<br>Meet drinking-water standards before infiltration into unsaturated zone. | Including, but not limited to<br>pH, daily.<br>Coliforms, daily.<br>Residual chlorine, continuously.<br>Potable water standards, quarterly.<br>Others, depending on original water constituents. | 600 m to extraction wells. May vary depending on treatment provided and site –specific conditions | The depth to groundwater (i.e. thickness of the vadose zone) should be at least 2 m at the maximum groundwater mounding point.<br>The recycled water should be retained underground for at least one year prior to withdrawal.<br>Recommended treatment is site-specific and depends on factors such as type of soil, percolation rate, thickness of vadose zone, native groundwater quality, and dilution.<br>Monitoring wells are necessary to detect the influence of the recharge operation on the groundwater.<br>The recycled water should not contain measurable levels of pathogens after percolation through the vadose zone[k].<br>Provide treatment reliability. |
| Indirect potable reuse<br>Groundwater recharge by injection into potable aquifers. | Secondary[a]<br>Filtration[b]<br>Disinfection[c]<br>Advanced wastewater treatment[d] | Including, but not limited to:<br>pH between 6.5 and 8.5.<br>$\leq$ 2 NTU<br>Not detectable | Includes, but not limited to:<br>pH, daily.<br>Turbidity — continuous<br>Coliforms, daily. | 600 m to the extraction wells. May vary depending on site-specific conditions. | The recycled should be retained underground for at least year one prior to withdrawal.<br>Monitoring wells are necessary to detect the influence of the recharge operation on the groundwater.<br>Recommended quality limits should be met at the point of injection |

| **Type of reuse** | **Treatment** | **Recycled water quality** | **Recycled water monitoring** | **Setback distances** | **Comments** |
|---|---|---|---|---|---|
| | | faecal coliforms /100 ml [f,g]<br>≥1 mg/l residual $Cl_2$ [h,i]<br>Meet drinking-water standards | $Cl_2$ residual continuous.<br>Drinking-water standards, quarterly.<br>Others, depends on original constituent. | | The recycled water should not contain measurable levels of pathogens at the point of injection.<br>A higher chlorine residual and/or a longer contact time may be necessary to assure virus inactivation.<br>Provide treatment reliability. |
| Augmentation of surface supplies. | Secondary [a]<br>Filtration [b]<br>Disinfection [c]<br>Advanced treatment [d] | Includes, but not limited to, the following:<br>pH between 6.5 and 8.5.<br>≤2 NTU [e]<br>Not detectable faecal coliforms /100 ml [f,g]<br>≥1 mg/l residual $Cl_2$ [h,i]<br>Meet drinking-water standards. | Includes, but not limited to, the following:<br>PH — daily<br>Turbidity continuous.<br>Coliforms — daily.<br>$Cl_2$ residual continuous. Drinking-water standards, quarterly.<br>Others, depending on constituent. | Site-specific | Recommended level of treatment is site-specific and depends on factors such as receiving water quality, time and distance to point of withdrawal, dilution and subsequent treatment prior to distribution for potable uses.<br>The recycled water should not contain measurable levels of pathogens.<br>A higher chorine residual and /or a longer contact time may be necessary to assure virus inactivation.<br>Provide treatment reliability. |

[a] Secondary treatment processes include activated sludge processes, trickling filters, percolating filters, rotating biological contactors, and many stabilization ponds systems. Secondary treatment should produce effluents in which both the BOD and SS do not exceed 30 mg/l.
[b] Filtration means the passing of wastewater through natural undisturbed soil or filter media such as sand and/or anthracite.
[c] Disinfection means the destruction, inactivation or removal of pathogen microorganisms by physical, chemical or biological means. Disinfection may be accomplished by chlorination, ozonation, other chemical disinfectants, UV radiation, membrane processes, or other processes.
[d] Advanced wastewater treatment processes include chemical clarification, carbon adsorption, reverse osmosis and other membrane processes, air stripping, ultrafiltration, and ion exchange.
[e] Turbidity level set must be met before disinfection. The value to be used is the average for 24 hours, during which turbidity must never exceed 5 NTU; and when solids are used as a measure, they must be 5 mg/l.
[f] Unless otherwise specified, coliform limits correspond to the median for 7-day experiments.
[g] The number of faecal coliforms must not exceed 14/100 ml in any sample.
[h] Chlorine value corresponds to residual chlorine after a contact time of 30 min.
[i] It is recommended that the microbiological quality of water be characterized before implementing a reuse program.

The establishment of limits as well as treatment methods was considered because:

- it is a limited list, the use of parameters does not fully cover or reach a complete description of water;
- the combination of a set of parameters with defined treatments allows for a reduction in monitoring needs;
- expensive pathogen monitoring is eliminated;
- the reliability of the treatment system is increased.

As regards the microbiological aspect, on the recommendation of outstanding microbiologists, only faecal coliforms were used due to their significance, and both multiple tube and membrane techniques were used to measure them. Coliform and turbidity (2 NTU) levels must be met. No value limits are contemplated for parasites or virus. In the case of parasites, they are not a concern in reuse water in the USA, because of the original water quality and treatment level. No limits are included for virus due to:

- lack of information on the presence of active viruses at levels sufficient to cause damage, when an appropriate treatment is applied;
- difficulty in determining virus type and concentration in residual water with precision, due to the low recovery rates;
- the lack of laboratories and trained personnel to carry out the analysis;
- experiments that take 4 weeks;
- lack of agreement among experts on the seriousness of health risk implied by low levels of virus in renovated water;
- the fact that no case of virus disease has been detected in connection with renovated water.

However, there is pressure to include some of them, such as *Giardia lamblia* and *Cryptosporidium parvum*. To meet these criteria it is recommended that a filtration step always be applied.

*Australia*

As regards aquifer recharge for human consumption, Australia produced draft criteria in 1995 (Dillon et al., 2001), which establish that wastewater must undergo a secondary treatment, filtration, pathogen reduction and advanced treatment, at specific sites. Renovated water must not present deleterious effects in the aquifer, faecal coliforms must not exceed 100 plaque forming units (PFU)/100 $m.l^{-1}$, and water must remain in the aquifer at least 12 months before extraction. Recovered water must comply with the criteria for potable water.

Australian regulations are more practical than those of the USA in two ways (Dillon et al., 2001):

- recharge conditions are established according to the aquifer use (e.g. for human consumption or agricultural irrigation), which allows for the use of different quality water in infiltration;
- a sustainable capacity of aquifer attenuation is accepted and is used to establish value limits, because it is known attenuation may control some contaminants, particularly microbiological ones.

The Australian standards regulating aquifer recharge give attention to the use and purposes of the recharge, and even environmental values. They promote removal of persistent compounds before injection as much as practicable, they use the aquifer treatment capacity in a sustainable manner, and demand demonstration of the attenuation capacity through monitoring and research, when applicable.

The standards have been incorporated in a multibarrier approach designed to protect groundwater quality (Dillon et al., 2000) (Fig. 3.9). Barriers include knowledge of the pollutant sources in catchment, source selection, aquifer recharge selection, detention time in the aquifer, level of pretreatment of injected water, injection shutdown system, maintenance and contingency plans, and quality of recovered water after treatment. In particular, natural attenuation must be demonstrated for each case because there is little knowledge about the natural capacity of assimilation of disinfection by-products, endocrine disrupters and toxic organic materials.

In 1996, further criteria were drafted to address the topic of recharge with rainwater and treated effluents (Dillon et al., 2001), with three objectives:

- to prevent irreversible fouling of injection wells and infiltration ponds;
- to protect and, in some cases, even improve underground water quality
- to ensure that recycled water is appropriate for its designated use.

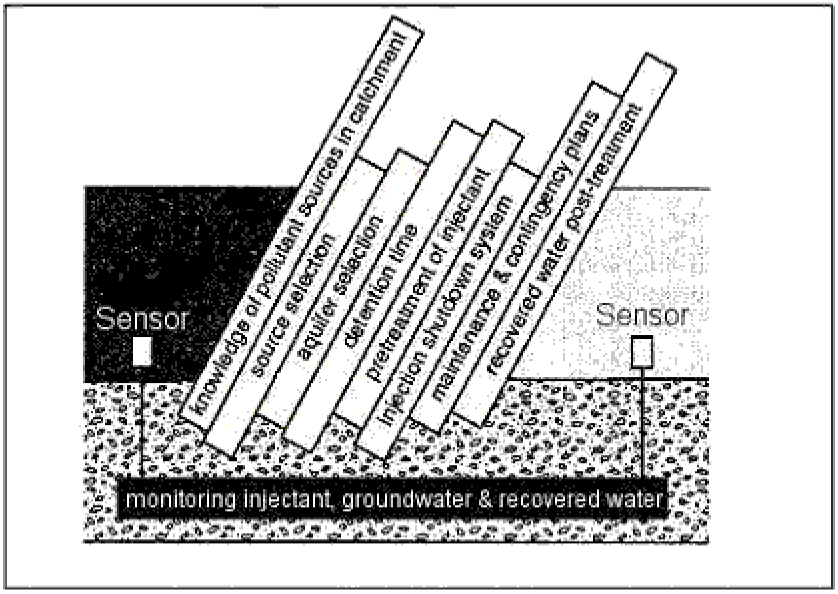


**Figure 3.9 A series of barriers has been developed to protect the quality of groundwater and recovered water at ASR sites (Dillon *et al.*, 2000)**

These criteria particularly take into account that water passage through soil leads to an improvement in quality, which should be taken advantage of in a sustainable manner. They use the precautionary principle[5] in view of the lack of reliable information on attenuation mechanisms. As a result, the criteria are considered, even in Australia, as being conservative. A better awareness and understanding of the phenomena that occur in aquifer recharge should allow the criteria to be improved in the future.

The criteria also contain sections on how to obtain licenses, how environmental management should be planned, pretreatment, monitoring, maximum concentration of contaminants and minimum residence times. Pretreatment is noted as especially important for those contaminants that are refractory to the attenuation mechanisms of the soil and groundwater. Shorter residence times

[5] 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.

may be used, depending on the water treatment capacity of the subsoil. Reinjection of storm water is allowed.

*Comments on regulations*

The history of public health and its relationship with water refers to the efforts made to provide a safe supply and dispose of wastewater adequately. This led to the control and treatment of discharges in developed countries, up to secondary levels, including disinfection. Reuse standards evolved in parallel with such actions, and are considered a "good practice", because many epidemics were controlled and other environmental improvements were made. Consequently, reuse standards reflect the quality of water obtained when it is submitted to an additional treatment in well-designed and well-operated secondary plants. In effect, the criteria originated with a pre-established quality and, because of the desire to promote reuse, they abandoned the use of lower quality water, regardless of the effect of the soil during recharge. The regulations thus not only reveal a concern for the consumption of renovated water, but also an institutional maturity and complexity difficult to achieve by less developed countries.

Throughout the world, it appears there is a dichotomy in the application of standards for the indirect reuse of water for drinkable purposes. At the present time there are physical and biological standards for the control of potable water, even when the sources from which this water comes have a quality similar to or worse than that of the recycled water. However, the fact that the former are considered "natural" makes them seem more reliable. On the other hand, when water has been processed in any way, it incites an almost natural distrust. However, until specific standards for reuse are developed, which permit assessment of human consumption of reused water, reuse projects are and will be judged against potabilization standards, in addition to whatever is established specifically for recycled water.

In addition, the management of water-related diseases requires a comprehensive approach that addresses multiple exposure pathways at the same time similar to that discussed in the "Stockholm framework" (see Appendix A).

### 3.6.8 A vision for the future–present

The establishment of regulations to promote reuse for human consumption is problematic, as is finding a way to improve the situation of those who engage in unintentional reuse. We find ourselves in a world divided into those concerned with problems that may arise in the future, and those who live in the present with no regard for the future situation of others, who will perhaps face worse conditions. For this reason, criteria must be sought that, at the present time, can look after the problems of both the present and the future. A solution could be the type of criteria proposed by Cotruvo in Table 3.48 with some modifications.

**Table 3.48 Guidelines for criteria applied to indirect potable reuse**

| |
|---|
| Recharge by spreading into potable aquifers<br>▪ **Primary treatment and disinfection, plus soil aquifer treatment, handling of dry and wet cycles as well as hydraulic and mass charges to avoid soil colmatation (clogging), in the event that suspended solids are mostly mineral.**<br>▪ **Primary advanced treatment and disinfection, plus handling of dry and wet cycles as well as hydraulic and mass charges so as to avoid soil colmatation, in the event that suspended solids are mostly mineral.**<br>▪ **Secondary treatment and disinfection with a well operated soil aquifer treatment.**<br>▪ **Possibly advanced treatment under site-specific considerations.**<br>▪ Meet drinking-water standards after percolation.<br>▪ Monitoring for coliforms. pH, chlorine residual, drinking-water standards plus site-specific others.<br>▪ Distance to point of extraction (600 m) or dependent on site-specific factors |
| Recharge by injection into potable aquifers.<br>▪ Secondary treatment, filtration, disinfection, advanced wastewater treatment.<br>▪ Meet drinking-water standards no detectable faecal coliforms in 100 ml, turbidity limits, 1 mg/l chlorine residual, pH between 6.5 and 8.5, others.<br>▪ Monitoring for turbidity, coliforms, chlorine residual, pH, drinking-water standards, others.<br>▪ Distance to point of extraction (600 m) or dependent on site-specific factors. |

Adapted from: Cotruvo (2001) (Wording in bold indicates modifications proposed to Cotruvo guidelines).

### 3.6.9 Research needs

Developing appropriate reuse criteria will require time and research. Some topics for this research are:

- how to more precisely establish the microbiological risks for developed and developing countries;
- presence and concentration of pathogen and toxic substances by region, with real-time and online monitoring;
- health significance of toxic and pathogenic concentrations present in renovated water;
- types of pathogen bacteria and viruses, and their behaviour in wastewater, treated water, renovated water and potable water;
- behaviour of each type of virus during the treatment processes;
- sustainable attenuation rate of specific pathogens and organic materials in the soil and the aquifer;
- development of models using the above data, to establish residence times or extraction distances;
- chlorination alternatives;

- careful evaluation of environmental and health risks, based on sound understanding of subsurface processes;
- determination of soil and aquifer attenuation for diverse locally relevant contaminants.

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# 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 aqufiers, 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 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 assessed, 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 17β-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$10^{-4}$ µg/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 17β-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 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 CCL[6] 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 $ID_{50}$ 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 $ID_{50}$, 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.

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[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.

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[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 water[8] 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 pharmaceuticals[9] 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 (TEQ[10]) 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 $ID_{50}$ 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. 17β-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 | | | Log10 removal | | |
|---|---|---|---|---|---|---|
| | Raw wastewater (maximum) | Allowed in 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 | n 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 |
| *Cryptosporidium parvum* | $6 \times 10^4$ oocysts/l | $2.6 \times 10^{-5}$ oocysts/l[h] | $5.4 \times 10^4$ oocysts/l | 9.35 | 1.1 | 8.3 |
| *Giardia intestinalis* | $1.5 \times 10^4$ cysts/l | $5.5 \times 10^{-6}$ cysts/l[g] | $0.6 \times 10^4$ cysts/l | 9.4 | 0.4 | 9.0 |
| Enteroviruses | $3 \times 10^5$ PFU/l | $1.8 \times 10^{-7}$ PFU/l[g] | $1.56 \times 10^5$ PFU/l | 12.2 | 0.3 | 11.9 |
| *Salmonella typhi* | $8 \times 10^4$ CFU/l | 0.19 CFU/l[g] | $8 \times 10^4$ 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

- electrodialysis.

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 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.

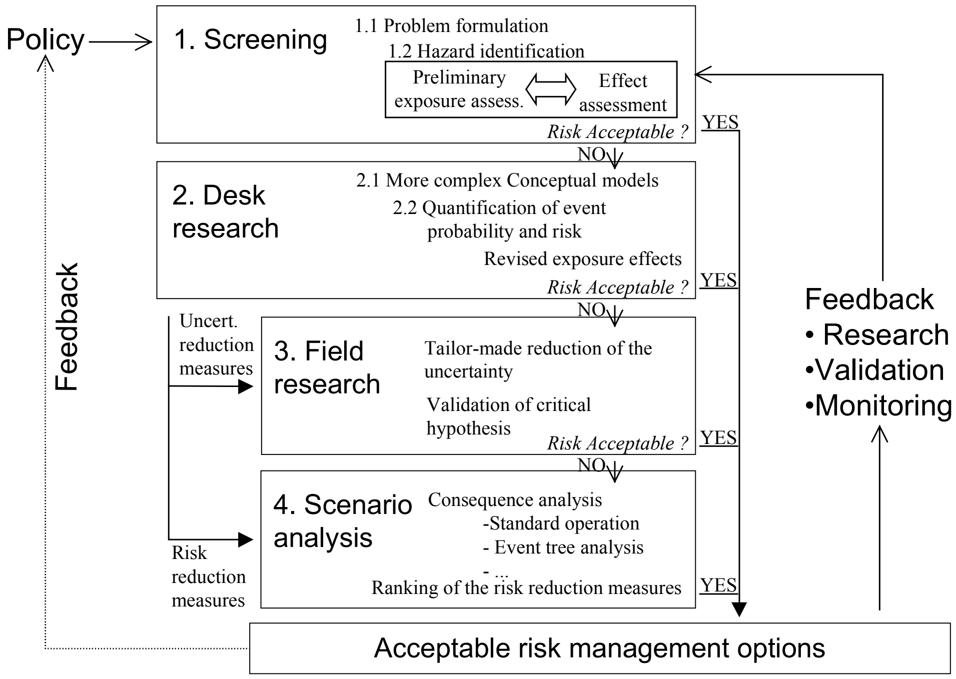


**Figure 4.1 A phased approach to model-based risk characterization**

*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 principle[11], 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.

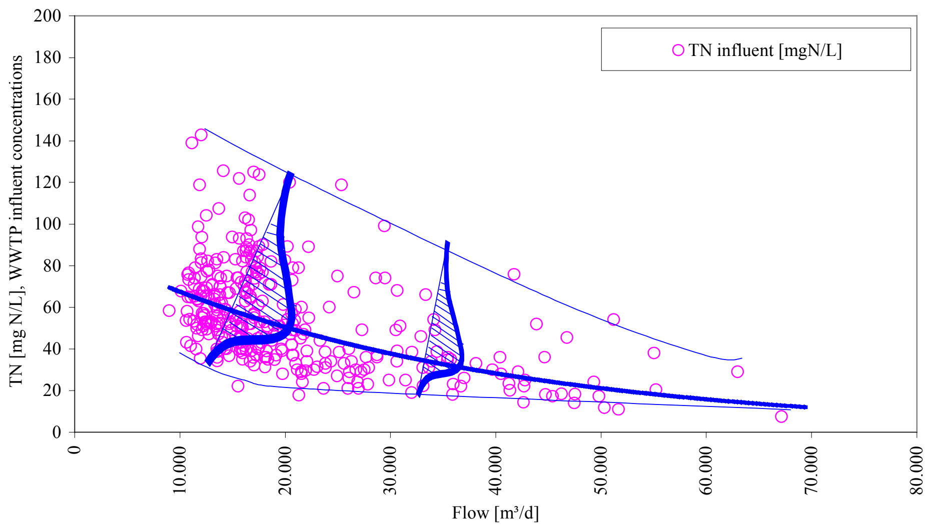


**Figure 4.2 Influent total nitrogen concentrations versus daily flow at a wastewater treatment plant inlet (400 data points)**

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.

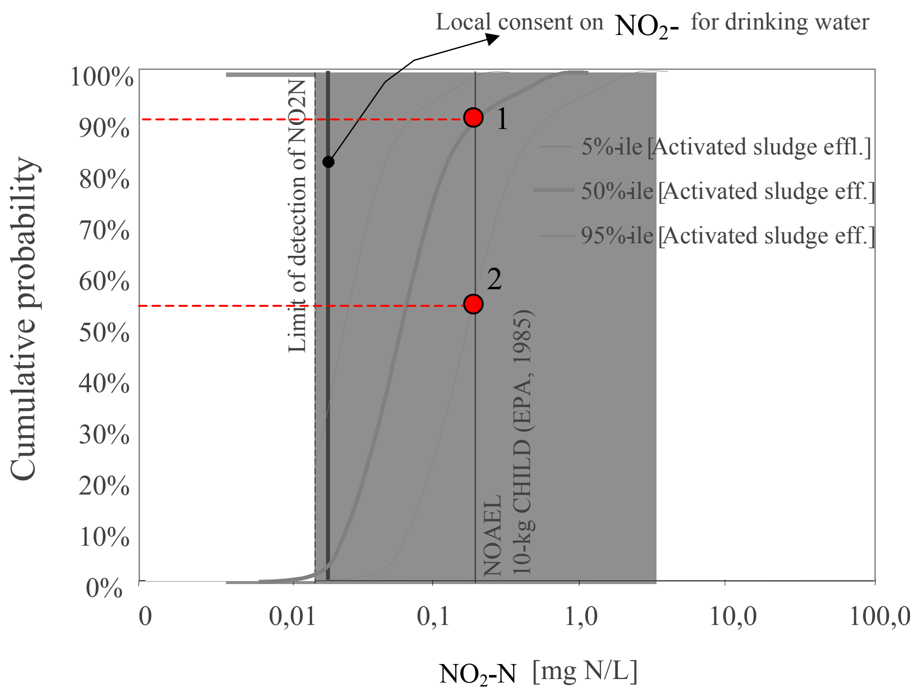


**Figure 4.3 Simulation results for nitrites for artificial aquifer recharge at the reuse installation: secondary effluent**

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).

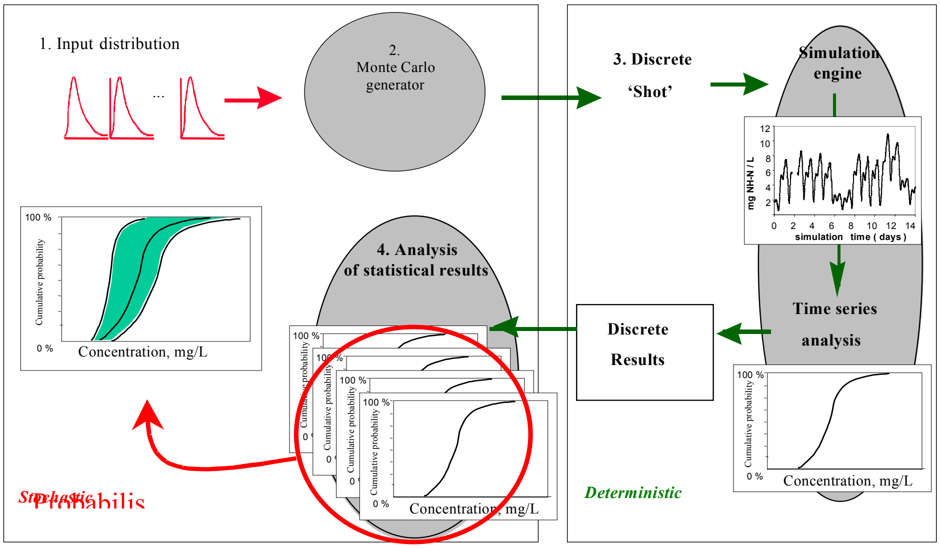


**Figure 4.4 Layout of the probabilistic methodology**

**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 & Aalderink (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 (Laîné 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 (Laîné 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:

- 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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# 5 Impact assessment of aquifer recharge

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## 5.1 Introduction

### 5.1.1 A changing context

Environmental development plans, policies and programmes can result in outcomes that may affect the health of present and future generations. There is a growing consensus that a systematic assessment of health effects is needed in such cases. This consensus has been expressed in numerous international laws and legally binding agreements, which now include provisions for the integration of health impact assessments in policy development (e.g. the Amsterdam Treaty of the European Union, Art.152; Declaration of the Third Ministerial Conference on Environment and Health).

The World Health Organization (WHO) stated that:

> *"Environmental health comprises those aspects of human health, including quality of life, that are determined by physical, chemical, biological, social and psychosocial factors in the environment. It also refers to the theory and practice of assessing, correcting, controlling, and preventing those factors in the environment that can potentially affect adversely the health of present and future generations".*

This definition clearly illustrates the need to integrate the results of health environmental impact assessments, because this is the only way to incorporate health protection and promotion in development policy. Incorporation of health promotion strategies in national and local development policy is now widely supported by health professionals involved in the formulation of national health policies.

The Ottawa Charter for Health Promotion (1998) states that:

> *"... Health promotion is the process of enabling people to increase control over, and to improve, their health. To reach a state of complete physical, mental and social, well-being, an individual or group must be able to identify and to realise aspirations, to satisfy needs, and to change or cope with the environment. Health is, therefore, seen as a resource for everyday life, not the objective of living. Health is a positive concept emphasizing social and personal resources, as well as physical capacities. Therefore, health promotion is not just the responsibility of the health sector, but goes beyond healthy life-styles to well-being...".*

The Director of the Pan American Health Organization stated that:

> *"...It is not enough to look at the health outcomes. One must look at those social conditions that determine health outcomes – the determinants of health ... . we must look at the disparities of these determinants of health and determine to what extent they are distributed so unequally as to produce health disparities. It is of fundamental importance that in discussions on equity, we understand the difference between disparities in health status and disparities in the determinants of health that cause these health inequalities or inequities" (Alleyne, 2000).*

There is evidently general international agreement on the concept of integrating health impact assessments in development planning. However, institutional mechanisms are lacking, especially at the local level, to include such assessments in development efforts outside the health sector proper, let alone take the result of such studies into account in the final policy formulation.

The omission of health impact assessment and the adoption of a nonintegrated, sectoral approach is likely to result in a serious underestimate of the overall cost of an adopted policy, particularly in the cost–benefit assessment, if this is based on a narrow preliminary impact assessment. This concern is particularly relevant in the area of water resource management, as will be demonstrated in this chapter.

### 5.1.2 Role of the health community

Health impact assessment can be described as the estimation of the effects of any specific action (plans, policies or programmes) in any given environment, on the health of a defined population. As such, a health impact assessment is an integral component of:

- strategic environment assessment for large-scale planning procedures;
- environmental impact assessment for specific projects;
- environmental assessment.

These three procedures share several key stages and follow-up measures. Implementing an appropriate assessment procedure during a limited preliminary assessment of policy options can achieve many goals at low costs and great benefit to the community. Yet, even in their basic form, assessment procedures can be dauntingly complex.

The use of groundwater for the production of drinking-water and the cultivation of a variety of food products, some of which are consumed raw, make it imperative to assess the health risks associated with any recharge option. An integrated approach to groundwater management in general, and a health impact assessment of management options in particular, is strongly recommended, especially where territorial limitations for water use and safe water are interconnected with social, economic and environmental issues.

The chemical and microbial quality of groundwater is inextricably linked to events occurring above the aquifer. Examples of some of the many factors that can impact on the quality of the groundwater are air deposition of small particles, contaminated rainfall, untreated stormwater, polluted agricultural runoff, untreated or partially treated wastewater discharged from municipal and industrial sources, accidental spills and illegal waste dumping. Chemical pollution can originate at a site far removed from the aquifer, but may still contaminate groundwater. Furthermore, some types of chemical pollution are irreversible, precluding the intended use of the aquifer (e.g. irrigation or production of drinking-water) for many decades. Intrusion of one specific ‘contaminant’ — seawater — is a particular concern in some areas, because salination compromises even industrial uses of the aquifer and is considered to be virtually irreversible. Finally, there is a history of institutional inattention to groundwater management compared to surface water management. This has hampered the collection of adequate environmental data, and the compilation of information on health impact assessment experiences, especially for new technological options. The lack of consolidated experience has caused further uncertainties in guidance on policy formulation and implementation.

The complexity of carrying out impact assessments makes it necessary for health professionals to extend their skills into new areas. For example, assessing the potential health effects of planned or accidental contamination of groundwater resources requires health professionals to acquire new knowledge on toxicology, and to become familiar with modern laboratory and data interpretation techniques.

This chapter addresses the challenges posed by health impact assessment related to aquifer recharge by means of recycled water. In particular, it:

- reviews the social, economic and environmental issues that act as health determinants;

- explores the role of preliminary environmental health assessment in defining policy options;
- describes the basic principles of environmental health impact assessment as applied to aquifer recharge.

## 5.2 Social, economic and environmental issues as determinants in groundwater management

### 5.2.1 Importance of a groundwater policy

Groundwater serves a variety of needs:

- *Agriculture* is by far the most intensive user of groundwater resources, especially through irrigation. It is estimated that 90% of all groundwater abstractions serve an agricultural need.
- *Industry* is the second largest consumer of groundwater, often using drinking-water for free, or at a heavily subsidized rate.
- *Household needs* for drinking, hygiene and food preparation are a third need to be met by groundwater resources.
- *Natural environment* relys on groundwater as the main source for the base flow of shallow aquifers and wetlands, representing an effective buffer against droughts.

Groundwater may also serve a variety of community water needs, such as watering of common areas and supply of water to community services (e.g. public buildings and fire brigades). Such applications are at present not considered as a strict health policy issue, but this view may well be revised in future.

Water resource management is an area undergoing rapid change. Policy development will need to take into account future evolution in the sector, which may include a number of issues that are discussed below.

*Effect of global forces*

The availability of water resources, their use and management will be shaped by a range of global forces. Such forces include rapidly changing demographic conditions, wider geographical distribution of human settlements, increased local demand for water and wider availability of technologically advanced options, such as desalination and water reuse.

*Private sector involvement*

An increased interest by the private sector in the management of groundwater resources can be expected as a result of advances in demand management. This, in turn, will affect water pricing. As the price of water increases, the demand for exploiting groundwater will grow, as will research into new technologies for abstraction and aquifer recharge.

*Deteriorating water quality*

Aquifer water quality can be expected to deteriorate as a result of several factors. However, this problem is currently receiving inadequate attention, because facilities and expertise for aquifer water quality assessment are lacking, as are databases dedicated to the gathering of relevant information. Inadequate investment hampers assessment of the impact of diffuse sources of pollution, and prevents a better understanding of the interaction between sources of pollution and groundwater quality. Pollution affecting groundwater can originate from the air or the surface, as discussed in Section 5.1.2. Excessive use of groundwater in many key crop-producing areas is pushing the watertable downward, while changes in groundwater flow carry pollutants into noncontaminated areas. Furthermore, the increasing use of bottled water (much of which is spring water, and thus comes from aquifers) is a trend that is extending into poorer countries or regions, setting a new exposure scenario that requires appropriate environmental health assessment and surveillance.

*Financial aspects*

Aquifer recharge with recycled water may require major investments to pretreat wastewater before is used for recharge purposes. This might, in turn, require the construction of new wastewater treatment plants, modernization of old plants, maintenance to more exacting standards or other substantial investments. The total financial requirements might make other forms of water resource management, such as desalination, financially competitive in certain areas.

Financial aspects such as access to funding and investment will be key to addressing the water crisis. The current trend of privatisation of water services may lead to an access to water determined by market-driven economical strategies, rather than by the recognition of any universal right of access to water. It is therefore crucial to implement a regulatory framework on water resource management and water use.

*Legislation*

Legislation concerning groundwater has often been neglected in comparison to the attention given to surface water. Regulatory frameworks are variable, and are often split into self-contained sets of regulations dealing with water, industry and agriculture — a situation that is not particularly helpful to local policy-makers.

*New technologies*

Artificial recharge will require the application of advanced technologies to avoid adverse health effects, especially where the end goal is the production of drinking-water. Preliminary assessment of all technical options for aquifer recharge needs to be part of any water management plan; for example, enhanced natural recharge could be an alternative option achievable at significantly lower cost than recharge with wastewater, but would require carefully planned land use.

### 5.2.2 Health aspects of a groundwater policy

The health impacts caused by scarcity or impaired quality of groundwater can be organized into three main categories:

- environment-related physical health:
  - communicable (microbiological contamination) and noncommunicable (chemical contamination) diseases;
  - poor hygiene caused by water scarcity;
  - unsafe drinking-water;
  - unsafe food crops due to contaminated irrigation water;
  - contamination-related diseases with effects on future generations (e.g. endocrine disrupting chemicals causing sterility and impaired juvenile development);
- health care services:
  - incremental health care needs,
  - displacement of traditional care services, surveillance, laboratory and expertise costs;

- social well-being:
  - effects on income, socioeconomic status and employment (industrial and agribusiness based on groundwater resource);
  - effects on migration and resettlement (migration related to water scarcity, increased water demand in highly urbanized areas);
  - continued investment needed for groundwater management (development, implementation and maintenance of aquifer recharge and water treatment plants), which can be a significant burden for local policy-makers.

A wise investment in groundwater management is therefore an investment in public health. Table 5.1 summarizes the benefits of investing in groundwater management, and the potential problems that arise from the lack of such investment.

**Table 5.1 Effects of investing or failing to invest in groundwater management**

| **Potential benefits of investing in groundwater management** |
|---|
| ▪ increased prosperity because a healthy population is a major contributor to a vibrant economy;<br>▪ reduced expenditures on health and social issues;<br>▪ overall social stability and well-being. |
| **Potential consequences of failing to invest in groundwater management** |
| ▪ lack of health impact assessment of policies, programmes and projects;<br>▪ greater-than-necessary adverse health impact of development policies.;<br>▪ the tendency of vertical disease control programmes to ignore environment and development links;<br>▪ lack of funding for research into health impacts |

It is important to remember that the concept of health has evolved to include not simply the absence of disease, but also the promotion of good health. The implementation of this expanded concept requires an assessment of the impact of any development on all aspects of human well-being. Decision-makers therefore need to be aware of the following basic principles when defining a policy for water resources management:

- health determinants can be defined as the range of personal, social, economic and environmental factors that determine the health status of an individual or a defined population;
- coordination of actions to improve health across various sectors implies multidisciplinary cooperation as well as in-depth knowledge of the individual sectors and their specific rules and regulations;
- the protection of human health reduces the socioeconomic burden of ill-health;
- policy and planning efforts need to include the potential reduction of costs in the health-care sector if policies with negative health consequences are abandoned;
- improved health results in improved productivity throughout life;
- WHO recommends that greater equity in health be achieved;
- health and well-being are key components of sustainable development;
- public awareness of health impact of environmental activities is increasing;

- health policy planning requires the experience of a multidisciplinary team based on the principle that “no one is essential but everyone is needed”;
- environmental health impact assessment is an essential element of any cost–benefit investigation;
- changes in environmental and social determinants of health will provide an incentive for the health sector to review the delivery of its services and improve performance and efficiency.

## 5.3 Preliminary environmental health impact assessement

In the case of aquifer recharge by means of recycled water, a cost–benefit analysis is particularly important because of the high costs involved in implementing recharge schemes, maintaining equipment and monitoring treatment facilities. Performing these operations without a proper environmental health impact assessment could create a considerable financial burden, without any guarantee of ultimate success.

A preliminary environmental health impact assessment is recommended as the first step during the initial evaluation of a new water option for water resource management. The goal is to assess all realistic policy options (e.g. natural recharge, artificial recharge, recharge by means of recycled water and desalination). The assessment must consider the potential health benefits as well as the health risks, to maximize benefits and reduce costs (especially those related to surveillance). Historical data can be particularly useful in this endeavour; examples of such data are incidence of water-related diseases in areas that use untreated water, or prejudice suffered in agricultural areas in times of water scarcity. An environmental health impact assessment should be approached from both a technical and an economical viewpoint, with a health impact assessment integrated as just one component in a broader appraisal.

The preliminary assessment should follow a staged approach, in line with the standard procedures for cost–benefit analysis (Pearce, 1983; Layard & Glaister, 1994; Layard & Misham, 1988) as shown in Figure 5.1.

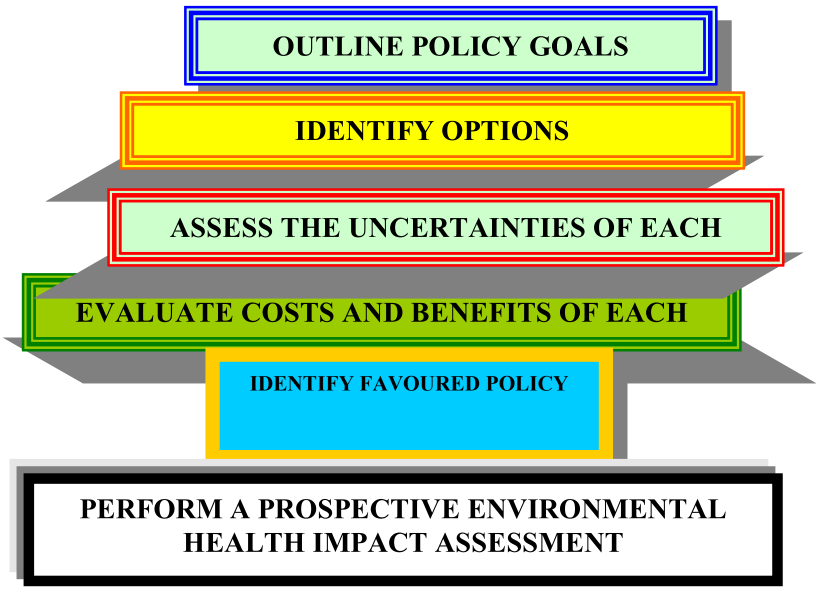


**Figure 5.1 Schematic outline of the preliminary assessment approach.**

Only the first two stages of the approach outlined in Figure 5.1 are discussed in detail here, because the aim of this chapter is limited to outlining basic principles of environmental health assessment, rather than detailed discussion of particular policy options.

### 5.3.1 Outline policy goals

National and international regulations related to groundwater management have been formulated in a variety of forums, and have generally been well received by the international community. Often, however, the true front-line managers dealing with water resource management are municipalities, communities, farmers and manufacturers, in short, the hands-on users. The adequacy of groundwater management policies and procedures thus depends on these stakeholders, who are often also responsible for land-use planning and wastewater management.

Actual or future problems related to water resource management need to be defined when policy goals are outlined, taking into account scarcity or deterioration of quality of water destined for drinking, agriculture, industrial use or community purposes. Demand management is a common component of any policy dealing with water resources, but in setting policy, it is important to identify different options for management (eg artificial or natural recharge) before deciding on the final selection.

Groundwater resources supporting drinking-water production will be of primary concern in the definition of any water resource policy. This is particularly so if a potential threat of contamination by persistent pollutants exists, because such contamination could be effectively irreversible. In this case, a more complex environmental health assessment is required, and a higher financial risk must be taken.

Groundwater resources used to meet needs other than drinking-water production (such as industrial, irrigation or community uses) may have contamination levels that are deemed "acceptable" for the intended purpose. Quality standards for such uses have to be set by local authorities based on international guidelines. Monitoring of sites where pollution originates and of quality of the applied groundwater, should be instituted and maintained.

Potential causes of water stress or water quality deterioration are summarized in Table 5.2.

**Table 5.2 Potential causes of water stress and water quality deterioration**

| Type of problem | Causes | Concerns |
|---|---|---|
| Anthropogenic pollution | Inadequate protection of vulnerable aquifers against human-made discharges and leachates from urban and/industrial activities and intensification of agricultural cultivation | Pathogens, $NO_3$, $NH_4$, Cl, $SO_4$, B, heavy metals, DOC aromatic and halogenated hydrocarbons |
| Excessive abstraction | Saline and/or polluted groundwater induced to flow into freshwater aquifer | Mainly Na and Cl but can also include persistent anthropogenic contaminants |
| Wellhead contamination | Inadequate well design and construction allowing direct ingress of polluted surface water or shallow groundwater | Mainly pathogens |
| Naturally occurring contamination | Relate to pH–Eh evolution of groundwater and dissolution of minerals (aggravated by anthropogenic and /or excessive abstraction) | Mainly Fe and F, but sometimes As, Mn, Al, Mg, $SO_4^{2-}$, Se and even $NO_3^-$ from palaeorecharge |

Even if water scarcity is not be a current problem, future demand, both in quantity and in quality, needs to be forecaste at this stage. Inclusion of data on population health is highly recommended, particularly data on mortality and morbidity suspected to be related to water scarcity, incidence of communicable disease caused by poor hygiene and contamination by pathogens. Information on health determinants should also be included (e.g. industrial growth and evolution of markets for agricultural produce).

The most desirable data could conceivably result from a retrospective health impact assessment related to historical water scarcity and its impact on population health. Environment and health data collected at this stage will also be important in the next stages, particularly in cost–benefit evaluation. Basic environmental studies, including monitoring programmes, are necessary at this stage to assess the real water scarcity. Such studies should give special attention to the determination of the potential storage capacity and flow of aquifers.

More complex environmental studies may be required at a later stage to identify options, particularly regarding contamination of the resource. In extreme cases, contamination of an aquifer, especially by persistent pollutants, would preclude any recharge.

### 5.3.2 Identification of options

Site-specific environmental assessment of aquifers (including flow directions relevant to exposure assessment) is a prerequisite to evaluating options for recharge operations. In the case of extended aquifers, the environmental assessment should be regional. The assessment should preferably follow a basin approach as this will give a better understanding of the potential for sustainable implementation. Important aquifer characteristics to assess at this stage include:

- nature of aquifer storage;
- groundwater recharge processes and rates of recharge;
- vulnerability or impaired effectiveness of the subsoil in pollutant attenuation.

These characteristics are discussed in detail below.

*Nature of aquifer storage*

Characteristics of the aquifer to be considered in assessing the nature of aquifer storage include hydraulic properties such as permeability and storability, and reservoir volume properties such as effective thickness and geographical extent. This assessment will provide information about the aquifer's capacity for autopurification or self-limitation of pollution. It will also help to verify whether aquifer recharge could be a useful part of groundwater demand management.

*Groundwater recharge processes and rates of recharge.*

Processes to be considered include direct recharge from land infiltration and indirect recharge from the beds of watercourses in the recharge area. This type of data is quite helpful for exploring links between recharge rates, land use and final water quality. Impaired recharge resulting from land use changes, anthropogenic contamination sources and other potential problems can be identified at this stage. Flow direction data are useful to identify pollution sources, and can be used in future exposure studies. Flow rates may form the basis for assessing the need for health surveillance on populations using water for agricultural irrigation or general community purposes. Information on groundwater recharge processes is also important for evaluating the possible advisability of actions aimed at improving natural recharge. Improving natural recharge can cost significantly less than implementing artificial recharge. Artificial recharge by injection will require extra energy for pumping, and the construction of new wells (unless the same well is used for recharge and abstraction). If the source of the recharge is recycled water and the final purpose is production of drinking-water, the cost of building, monitoring and managing facilities also needs to be taken into account.

*Vulnerability or impaired effectiveness of subsoil in pollutant attenuation*

Assessing the vulnerability or impaired effectiveness of subsoil will require the assessment of subsoil profiles and the determination of hydrogeological characteristics. Subsoil attenuation for certain pollutants could help to reverse water contamination, thus allowing the aquifer to meet the requirements of the most prevalent water use. Aquifer recharge can only be attempted if natural defences are intact and appropriate land use zoning is implemented.

### 5.3.3 Integrating the results of the assessment

The environmental investigation outlined above is highly relevant for safeguarding population health. Toxicological studies undertaken at the start of the preliminary assessment will define the primary exposure pathway (e.g. drinking-water, agricultural produce or irrigation water). This information is particularly important in the case of chemical contamination, because it will allow a better definition and management of hazards to population health.

The integrated information resulting from this stage is also important in cases where recharge is considered in response to water scarcity. Preferred options should also be identified through this study. For example, if water scarcity is mainly due to climatic changes, improvement of natural aquifer recharge may be preferred over the construction of artificial basins in order to reduce loss by evapotranspiration at comparatively minimal cost. Alternatively, knowing that natural aquifer recharge can be quantitatively impaired by human land use such as construction of housing, deforestation and changes in river courses, environmental (including climatic) assessment will identify data and trends that are helpful for predicting potential benefits from new land use policies.

The study will also revisit current land-use policy. Implementation of appropriate zoning laws can result in the protection of the quality of the natural recharge, particularly against contamination by pathogens originating in, for example, cattle herding in the recharge zone.

## 5.4 Environmental health impact assessment

Environmental health impact assessment can be performed as a:

- prospective assessment of the proposed new policy to identify its likely impact;
- retrospective assessment of effects following policy implementation;
- concurrent assessment, where the policy is assessed at the same time as it is implemented, to identify the true nature of the impact in circumstances where the impacts have been anticipated but not characterized.

While a preliminary assessment needs to consider all determinants of health, once a project has been chosen, the focus will change to physical health, in particular:

- the potential health impact on the surrounding population — this will require hazard and exposure assessments;
- an assessment of the need, and a definition of the methodology, of epidemiological studies, to determine whether a link exists between health outcomes and water contamination;
- an assessment of the necessity to monitor environmental and health data (e.g. health surveillance to investigate any significant deviation from the baseline status leading to a risk factor identification).

Health professionals will contribute to the implementation of risk communication to stakeholders, and to the determination of (potential or actual) health costs that are supposed to be an integral part of the overall cost–benefit evaluation.

### 5.4.1 Epidemiological studies

Environmental health impact assessments require also epidemiological studies. These are mostly observational studies, carried out along any of the following approaches:

*Cohort studies*

Cohort studies respond to the question: "What are the health effects of a given exposure?" The cohort study is an observational approach, which most closely resembles an experimental study. Exposed and unexposed populations or identified groups (e.g. vulnerable groups such as children or elderly persons) identified at one point in time are then followed to assess differences in health outcomes among them. In this study, the investigator controls neither the exposure conditions nor the attribution of exposures to the study object. As a result, risk factors to the health outcome are more likely to be unevenly distributed between the exposed and the unexposed groups, leading to differences in baseline risk. To moderate this comparability problem (characteristic of all observational studies), the investigator can only control unexposed groups. The technique may be used retrospectively when subjects have been identified and followed-up in the past, or may be used prospectively, in which case, a long-term and hence costly follow-up may be required.

The measure of the effect is described by:

- the risk ratio or relative risk (the proportion of exposed cohort developing the disease of interest, relative to the unexposed group); and
- incidence of mortality rate ratio (incidence rate of the outcome in the exposed group relative to the unexposed group).

*Case–control studies*

Case–control studies are used when there is a need to assess the contribution of environmental causes to a given disease. Also known as case–referent studies, they are the most commonly used environmental health investigation. Case–control studies differ substantially from cohort studies: investigators identify and select cases (i.e. subjects affected by the disease of interest), and controls (i.e subjects without the disease of interest). These groups are then followed backward to assess whether their respective past patterns of exposure differed before the cases actually developed the disease. A case–control study is not suitable for direct measurement of risk, because the sample of cases and controls is not proportional to the underlying population.

*Cross-sectional studies*

In a cross-sectional study, the prevalence of a particular disease, set of symptoms or other indication of ill-health is investigated at a single time-point (or over a relatively narrow period of time). Comparisons can then be made between the frequency of ill-health; for example, between workers exposed to a particular hazard and those not exposed. Alternatively, the study can compare workers suffering different degrees of exposure. A cross-sectional study can determine the prevalence rate, defined as the number of existing cases divided by the population at a given time point.

### 5.4.2 Health hazard and risk: assessing the potential health impact on the surrounding population through exposure assessment

A hazard is defined as the potential to cause harm, while a risk is defined as the likelihood that harm will indeed occur.

Aquifer recharge by recycled water may pose health hazards due to contaminants present in the treated wastewater, or in the water pumped from the recharged aquifer. Assessment of the health hazards in water pumped from the aquifer against quality criteria for intended use (e.g. irrigation or drinking-water) could disclose health risks that may otherwise be hard to assess.

The quality of the recharged groundwater is not always strictly correlated to the quality of the recycled water being infiltrated or pumped into the aquifer. Results from the mixing of recycled water and water already present in the aquifer can be further modified by several physical, chemical and biological interactions between water and the subsoil; such interactions are often unpredictable.

The presence of contaminants in groundwater does not necessarily imply that recycled water used for recharge is contaminated. As stated earlier, groundwater can be contaminated from sources that are geographically far removed from the point of recharge or abstraction. This spatial problem is further compounded by the very slow and variable flow rates (from tens to hundred of years) characteristic of many aquifers, so that unequivocal identification of a pollution source affecting a given aquifer is difficult. The presence of a source of pollution outside the aquifer or the recycled water can easily be verified by simultaneously investigating the quality of both waters. If the presence of an external source of pollution is confirmed, it is wise to investigate recharge areas first, followed by other potential sources of groundwater contamination (e.g. waste dumping, irrigation practices, industrial discharge and uncontrolled animal husbandry).

Potential hazards to health may occasionally derive from naturally occurring chemical substances or biological pathogens, but more usually originates from an anthropogenic source. A list of significant pathogens that relate to contaminated water is suggested in Table 5.2.

**Table 5.3 Orally transmitted waterborne pathogens and their significance in water supplies**

| Pathogen | Health significance | Persistence in water supplies[a] | Resistance to chlorine[b] | Relative infective dose[c] | Important animal reservoir |
|---|---|---|---|---|---|
| Bacteria | | | | | |
| *Campylobacter jejuni, E. colil* | High | Moderate | Low | Moderate | Yes |
| Pathogenic *E. colil* | High | Moderate | Low | High | Yes |
| *Salmonella typhi* | High | Moderate | Low | High[d] | No |
| Other *Salmonella* species | High | Long | Low | High | Yes |
| *Shigella* species | High | Short | Low | Moderate | No |
| *Vibrio cholerae* | High | Short | Low | High | No |
| *Yersinia enterocolitica* | High | Long | Low | High (?) | No |
| *Pseudomonas aeruginosa*[e] | Moderate | May multiply | Moderate | High (?) | No |
| *Aeromonas* species | Moderate | May multiply | Low | High (?) | No |
| Viruses | | | | | |
| Adenoviruses | High | ? | Moderate | Low | No |
| Enteroviruses | High | Long | Moderate | Low | No |
| Hepatitis A | High | ? | Moderate | Low | No |
| Enterically transmitted non-A, non-B, hepatitis viruses, hepatitis E | High | ? | ? | Low | No |
| Norwalk virus | High | ? | ? | Low | No |
| Rotavirus | High | ? | ? | Moderate | No (?) |
| Small round viruses | Moderate | ? | ? | Low (?) | No |
| Protozoa | | | | | |
| *Entamoeba histolytica* | High | Moderate | High | Low | No |
| *Giardia intestinalis* | High | Moderate | High | Low | Yes |
| *Cryptosporidium parvum* | High | Long | High | Low | Yes |
| Helminths | | | | | |
| Dracunculus medinensis | High | Moderate | Moderate | Low | Yes |

? — not known or uncertain
[a] Detection period for infective stage in water at 20°C: short = up to 1 week; moderate = 1 week to 1 month; long = over 1 month.
[b] When the infective stage is freely suspended in water treated at conventional doses and contact times. Resistance moderate implies that the agent may not be completely destroyed.
[c] Dose required to cause infection in 50% of health adult volunteers; may be as little as one infective unit for some viruses.
[d] From experiments with human volunteers (see Section X)
[e] Main route of infections is by skin contact, but can infect immunosuppressed or cancer patients orally
Source: WHO (2003)

Some chemicals of considerable chemical stability and hence long-lasting environmental persistence, which could be present in groundwater, are listed in Table 5.4.

**Table 5.4 Chemical contaminants of concern that could be present in groundwater**

| Contaminants | Media |
|---|---|
| Asbestos | Soil, groundwater, air |
| Chlorinated hydrocarbons | Soil, groundwater, air |
| Dioxins | Soil, groundwater, air |
| Metals | Soil, groundwater, air |
| Pathogens | Soil, groundwater, air |
| Total petroleum hydrocarbons | Soil, groundwater, air |
| Pesticides | Soil, groundwater |
| Pharmaceuticals | Soil, groundwater |
| Polychlorinated biphenyls | Soil |

Both microbial and chemical potential health hazards need to be reviewed to guide health surveillance and epidemiological studies. Pathogens generally offer a simple cause–effect relationship. The issue becomes more complex in the case of chemical contamination where the individual health outcome is the result of multifactorial effects, involving interplay of genetic, lifestyle, occupational and environmental factors. The long latency of many diseases further complicates the issue. Toxicological characteristics of potential hazards need to be investigated from recent scientific data; many international organizations currently make such information available online. Some health effects from persistent chemical contaminants are summarized in Table 5.5.

**Table 5.5 Health effects from persistent chemical contaminants**

| Health effect | Sensitive group | Some associated chemicals[a] |
|---|---|---|
| Cancer | All | Asbestos, PAHs, benzene, dioxins, some metals, some pesticides, carcinogens, some solvents and natural toxins |
| Cardiovascular diseases | Especially elderly | Carbon monoxide arsenic, lead, cadmium, cobalt, calcium and magnesium |
| Respiratory diseases | Children, especially asthmatics | Inhalable particles, sulfur dioxide, nitrogen dioxide, ozone, hydrocarbons, some solvents, terpenes |
| Allergies and hypersensitivities | All, especially children | Particles, ozone, nickel, chromium |
| Reproduction | Adults of reproductive age | PCBs, DDT, phthalates |
| Developmental | Fetuses, children | Lead, mercury, other endocrine disruptors |
| Nervous system disorders | Fetuses, children | PCBs, methyl mercury. lead, manganese, aluminium, organic solvents |

DDT = dichlorodiphenyltrichlorethane; PAHs = polycyclic aromatic hydrocarbons; PCBs = polychlorinated biphenyls
a Examples only:
Source: EEA, “Chemicals in the European Environment: Low Doses, High Stakes?Annual Message 2”, July 1999

Human exposure to hazardous substances requires a complete exposure pathway. Defining such a pathway requires the identification of the source, pathway, point of exposure, exposure routes and receptors.

*Source*

The source is the origin of the contaminants. Sources can be either localized (point source) or spread over a wide geographical area (diffuse source).

*Pathway*

A pathway can be defined as an existing or potential physical link between sources and receptors. Such a link can be direct, when the source is in direct contact with the receptor, or indirect, when the contaminant is transported from the source to the receptor through environmental media. Surface water, air, soil, subsoil and sediments can all be considered as environmental media that can carry contaminants to aquifers and hence form part of a pathway.

*Point of exposure*

The point of exposure is the location of potential or actual human contact with contaminated environmental media. Typical examples include drinking-water, irrigating wells And food grown through irrigation using recharged groundwater, especially when the food is eaten raw. An often-overlooked aspect of exposure is the biological availability of potentially harmful chemicals through the food-chain. Hunting, fishing, foraging and farming activities may bring people into contact with such contaminants. When contamination of edible plants or animals is suspected, specific data obtaind through sampling and biota studies are needed to evaluate any potential exposure pathway through the food-chain. Diffusion of contaminants to plants or animals can be evaluated referring to toxicological and ecotoxicological data. The latter will give information on the length of contamination (because biological organisms act as bioaccumulators of contaminants) and the capacity of the environment to react to contamination-induced stressors. An initial approach towards the investigation of water sources is shown in Table 5.6.

**Table 5.6 Collecting samples and environmental data and information of concern for investigation of water sources**

| | |
|---|---|
| Well survey: | ▪ well survey and inventory within the potential affected area, whichever is greater; |
| | ▪ an inventory of larger area downgradient of any known groundwater plumes, depending on site-specific hydrogeology and the extent of contamination; |
| | ▪ the well inventory should include the number, total depth, screen interval, use, yield, status, installation date, pump type and age, and location of all local wells and developed springs |
| Water sources: | ▪ monitoring wells; |
| | ▪ facilities water supply wells; |
| | ▪ municipal/utilities wells, springs and reservoirs; |
| | ▪ residential wells or springs or small. |
| Hydrogeology: | ▪ depth, thickness, extent, name and characteristics (including flow direction) of all groundwaters potentially affected by contaminations; |
| | ▪ depth, thickness, extent, name and characteristics (including flow direction) of all drinking-water aquifers; |
| | ▪ vertical and lateral extent of groundwater contamination. |

Adapted from ATSDR (1977).

*Exposure routes*

Although ingestion is the prevailing exposure route for drinking-water produced from recharged aquifers, dermal absorption and skin contact can also be considered. However, information on residence time is often lacking, which can compromise exposure assessment. In assessing exposure, the choice of the measurement methodology is very important. Duration, intensity and frequency of exposure all contribute to the calculation of the cumulative exposure. It may be important to evaluate intermittent or peak exposure as well as the mean exposure time. The period of exposure and the latency time must also be considered, depending on the particular health outcome of interest. Exposure is generally poorly characterized in past epidemiological studies. This makes it difficult to establish an unequivocal causal relationship between chemical contamination and health outcome.

*Receptors*

Receptors are organisms or environmental media that are exposed to the contamination. In the context of this chapter, the human population is the final receptor. Identification of receptors (e.g. workers, consumers and residents) is the last step in an exposure assessment. Population data must include vulnerable groups of interest such as children and elderly people). An exposure assessment needs to be undertaken on the smallest geographical distribution. This may help to determine whether health hazards affect predominantly certain groups or geographical areas.

### 5.4.3 Health impact assessment: suggested working procedures

Baseline data concerning population and health outcome may be collected from a variety of sources. In gathering population baseline data, care needs to be taken to avoid or correct the common problem of residence misclassification. The data obtained are generally useful not only for the initial study but also for future epidemiological work.

Health outcome databases will provide information on health conditions that may prevail in the area under investigation. Mortality rate databases address causes of death (e.g. cancer, infectious diseases and poisoning). Morbidity rates are of interest for health surveillance purposes for the identification of significant deviations from the baseline status. Unfortunately, health databases are often not recorded in a homogeneous way, and their usefulness is often limited to monitoring the frequency of events rather than estimating a disease rate. Therefore, in addition to baseline data on mortality and disease incidence rate, a cross-sectoral health survey of a random sample of the population may be useful to provide information on the prevalence and frequency of general health conditions and lifestyles.

Once baseline data are available, prospective studies can be performed to define any potential association between exposure and health outcome. A cohort study is not the first option for any health impact assessment because of the time and cost involved. Rather, health and environmental monitoring are recommended, followed (eventually) by case–control studies with detailed residential information.

Health surveillance is a routine system of capturing cases that have been recorded by existing health services. The analysis of the resulting data should indicate whether there has been a significant increase in the recorded number of cases of the health impact of interest.

Health services may be provided with specific tools for the purposes of the study. For example, in monitoring studies, databases referenced to a geographical information system (GIS) can be useful in integrating fragmented information from data sources; it also allows the detection of links between pollution sources and the location of particular health outcomes. Further integration of such information on a geographical map with other relevant data such as the geological characteristics, flow direction and extent of the aquifer can be helpful. Such data compilation will be useful in the (quite difficult) investigation and interpretation of environmental factors and health outcome.

Timing of follow-up studies for health surveillance depends on the latency time of the health outcome of interest. In the case of chemical contaminants, scheduling should take into account cumulative impacts.

Case–control studies are used when a significant deviation from the baseline has been registered during the monitoring phase. The limits of this study design and its relevance for the topic at hand are described above (Section 5.4.1). Data collected during the initial phases of a health impact assessment, such as baseline assessment and monitoring of health surveillance, may speed up a case–control study; however, the long latency of certain impacts will require a long observation time. Uncertainties in the assessment of risk factors, especially in cases of multiple exposures, make results difficult to interpret unequivocally. Thus, case–control studies are best used only in particular cases; for example, when teratogenic contaminants or microbial pathogens are present.

Further epidemiological investigations are recommended only when characterization of environment and populations, exposure pathways, environmental monitoring or exposure estimations have indicated that a completed exposure pathway is likely to exist, and when a health impact assessment has revealed a real deviation from the baseline.

## 5.5 Conclusions

Health impact assessment of aquifer recharge by recycled water needs to be considered in the overall context of groundwater management. Health impact assessment is recommended both in the preliminary assessment (when different technical options for aquifer recharge are being evaluated) and in the characterization of exposure to potential pathogens. Prospective data collection, possibly using GIS, environmental mapping of the baseline situation and health surveillance are strongly recommended.

Health professionals face a new challenge in assessing risks and health impacts by persistent chemical contaminants, whose health outcome often encompasses a wide range of causative factors. Improving health professionals' knowledge and creating mechanisms to enable daily cooperation with multidisciplinary teams will be neccessary in the future.

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# 6 Groundwater recharge with recycled municipal wastewater: health and regulatory considerations

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## 6.1 Introduction

The lack of specific criteria and guidelines governing the artificial recharge of groundwater with recycled municipal wastewater is currently hampering the implementation of large-scale groundwater recharge operations. Thus, policies and guidance for planning and implementing new groundwater recharge projects are needed, and would serve as the basis on which future and current groundwater projects would be designed and evaluated.

This chapter discusses some of the challenges for groundwater recharge. It presents three case histories of wastewater recycling for nonpotable reuse, with less stringent water quality goals than would be required if potable reuse was being considered. It also discusses risk assessment for drinking-water contaminants and presents the proposed State of California criteria for groundwater recharge as an illustration of a conservative approach to groundwater recharge.

## 6.2 Challenges for groundwater recharge

Groundwater recharge with recycled wastewater presents a wide spectrum of technical and health challenges that must be carefully evaluated. Some basic questions that need to be addressed include (Asano and Wassermann, 1980; Roberts, 1980; NRC, 1994):

- What treatment processes are available for producing water suitable for groundwater recharge?
- How do these processes perform in practice at specific sites?
- How does water quality change during infiltration–percolation and in the groundwater zone?
- What do infiltration–percolation and groundwater passage contribute to the overall treatment system performance and reliability?
- What are the important health issues to be resolved?
- How do these issues influence groundwater recharge regulations at the points of recharge and extraction?
- What benefits, problems and successes have been experienced in practice?

Pretreatment requirements for groundwater recharge vary considerably depending upon the purpose of groundwater recharge, sources of recycled wastewater, recharge methods and location. Although the surface spreading method of groundwater recharge is in itself an effective form of wastewater treatment, a certain degree of pretreatment must be provided to municipal wastewater before it can be used for groundwater recharge.

## 6.3 Case studies of groundwater recharge with soil aquifer treatment

This section describes two case histories that illustrate the use of soil aquifer treatment for groundwater recharge. The first case history looks at a variation of the treatment process in Ben Sergao, a suburb of Agadir, in Morocco. The pilot study is of interest not only for the Greater Agadir, where water resources are limited, but also for a number of cities in Morocco where reuse of treated wastewater constitutes an essential option for wastewater treatment and disposal. The second case history looks at infiltration–percolation as a tertiary treatment to meet the WHO's microbiological standards applying to unrestricted agricultural reuse (WHO, 1989).

### 6.3.1 Case history 1 — wastewater treatment at Greater Agadir, Morocco

With a population of over 350,000, the rapidly growing Greater Agadir faces an increasing need for wastewater treatment and an increasing demand for water supplies. The two main discharges of raw sewage — one into the port area, the other into the bed of the Souss wadi within a few kilometres of its mouth — are incompatible with a valuable tourist attraction.

In a cooperative project between Morocco and France, pilot wastewater treatment through dune sand infiltration–percolation is underway at Ben Sergao, a suburb of Agadir (Bennani et al., 1992). The initial chemical oxygen demand (COD) of raw sewage is 1190 mg/l, and the first treatment is by anaerobic stabilization pond. The pilot plant to treat wastewater by infiltration–percolation treats 1000 $m^3$/day of highly concentrated effluents in five infiltration basins of 1500 $m^2$ each, consisting of 2 m thick eolian sand. The anaerobic stabilization pond (1500 $m^3$ for a theoretical residence time of 2 days; depth of 3–4 m) is used to reduce suspended solids (40–50 %) and organic matter (50–60 %), increasing the rate of infiltration and reducing the surface area necessary for the infiltration basin. The basin is submerged for 8 hours and remains dry for 16 hours.

The wastewater is infiltrated at the rate of 1 m/day. Nearly 100% of suspended solids and 95% of COD are removed; 85% of nitrogen is in oxidized form and 56% is removed. Microbiological quality of raw sewage, pond effluent and percolated water are shown in Table 6.1. The percolated water will be used in growing tomatoes (a vegetable extensively cultivated in the Agadir region), public gardens and future golf courses.

Inasmuch as recharged groundwater may be an eventual source of potable water supply, groundwater recharge with recycled municipal wastewater may often involve treatment beyond the conventional secondary wastewater treatment level. In the past, several apparently successful groundwater recharge projects were developed and operated using primary and secondary effluents in spreading basins. However, because of the increasing concerns about protozoan cysts, enteric viruses, and trace organics in drinking-water, groundwater recharge with recycled wastewater in industrialized countries now generally entails further treatment after conventional secondary treatment. For example, surface spreading operations practiced in the USA to reclaim wastewater commonly include primary and secondary wastewater treatment, tertiary granular-medium filtration and, finally, chlorine disinfection.

**Table 6.1 Microbiological quality of raw sewage, pond effluent, and percolated water**

| | Raw sewage | Pond effluent | Percolated water | Overall removal efficiency |
|---|---|---|---|---|
| Fecal coliforms (No./100 ml) | $6x10^6$ | $5x10^5$ | 327 | 4.26 logs |
| Fecal streptococci (No./100 ml) | $2x10^7$ | $1.6x10^6$ | 346 | 4.78 logs |
| Nematode ova (No./l) | 139 | 32 | 0 | ~100% |
| Cestode ova (No./l) | 75 | 18 | 0 | ~100% |
| Total helminths ova (No./l) | 214 | 47 | 0 | ~100% |

Source: Bennani et al. (1992)

### 6.3.2 Case history 2 — disinfection of secondary effluents via infiltration and percolation

This case study deals with infiltration–percolation as a tertiary treatment to meet the microbiological standards applying to unrestricted agricultural reuse (Brissaud et al., 1999).

Because of the potential for direct human exposure, effluents from conventional wastewater treatment must be disinfected if they are to be used for irrigation of public parks, sports fields, golf courses and edible crops, to comply with relevant regulations. When the object is to meet WHO's unrestricted irrigation criteria (WHO, 1989), the additional treatment can be achieved through infiltration–percolation. An infiltration–percolation pilot plant constructed in Vall-Llobrega, Catalonia, Spain was intermittently fed with secondary effluents which percolate through 1.5 to 2 m of unsaturated coarse sand and recovered by under drains. In 1997, a 0.82 m/d hydraulic load was applied for more than six months. The mean fecal coliform removal was about four log units.

Infiltration–percolation allows oxidation and disinfection of the wastewater to occur. This is why soil aquifer treatment is used in Spain and France as a tertiary treatment with the aim of removing pathogens from the effluents of conventional wastewater treatment plants. It is a low-technology method that can be used to prepare wastewater for unrestricted irrigation (Brissaud et al., 1999). However, the reported disinfection performance provided by infiltration–percolation is uneven and dependent on the hydraulic loading of the system. The average water quality of secondary effluents and the percolated water is shown in Table 6.2.

**Table 6.2 Water quality of secondary effluent and percolated water**

| | Secondary effluent | Percolated water |
|---|---|---|
| Suspended solids (mg/l) | 18 | 1.2 |
| COD (mg/l) | 97 | 51 |
| $NH_3$-N (mg/l) | 28 | 0.5 |
| $NO_3$-N (mg/l) | | 47 |
| Fecal coliform (CFU/100 ml) | $6.1 \times 10^5 \sim 7.3 \times 10^6$ | Variable, in the range of 100–500 dependent on the hydraulic loading |

COD = chemical oxygen demand, CFU = colony forming units.
Source: Brissaud et al. (1999)

## 6.4 Health and regulatory aspects of groundwater recharge with recycled wastewater

Groundwater recharge with reclaimed water and direct potable water reuse share many of the public health concerns encountered in drinking-water withdrawn from polluted rivers and reservoirs. Ongerth and Ongerth (1982) have argued that the issue that has arisen in regard to groundwater recharge relates to the so-called "single standard" – meaning that all domestic water sources (natural and recycled water) should be subject to the same set of standards. The implication in the use of this term regarding wastewater reuse is that more stringent requirements may have been applied to recycled water for potable applications than for surface water or groundwater sources now in use. Clearly, whatever the source of a given array of chemical contaminants, the same health effects will occur and be of equal health consequences if the exposure is the same. There is, however, an important difference in the circumstances. With existing water sources, chemical contaminants should be identified and their harmful effects understood and mitigated as promptly as resources and public policy will allow. On the other hand, proposals to develop optional projects for potable reuse of wastewater that may deliberately introduce residues of uncharacterized substances into community water supplies should be carefully considered case-by-case to assure that benefits are significantly greater than the potential unquantifiable risks.

The current drinking-water standards and guidelines were not designed to deal with the mixtures and individual contaminants that may be unique to wastewater sources, so they alone cannot be used to completely evaluate the acceptability of a recycled water for potable reuse. However, it is within our capability to provide appropriate standards that may involve limits on both individual substances and aggregates, as well as treatment and source management specifications.

Tapping of polluted sources has potential effects that go beyond the increased cost of additional treatment. Incidental or "unplanned" indirect potable reuse of polluted water may expose people to health risks not associated with protected sources. The health concerns associated with drinking-water drawing upon polluted sources apply even more forcefully to treated wastewater reuse for portable purposes. In order to form a sound policy, the following questions should be considered: (a) is a water reuse option necessary as a water resource alternative; (b) what level of risk control is attained by a standard relative to the intended use; (c) how valid is the judgement of that level of risk, and, what is the acceptability of a given degree of risk? Risk analysis as applied to recycled water entails the same difficulties as that for other health hazards in the environment. Basically, the problem lies in quantifying the risks involved and agreeing upon what level of risk to accept (Ongerth and Ongerth, 1982).

### 6.4.1 Case history 3 — advanced wastewater treatment

Because of unspecified health concerns, several groundwater recharge projects in the USA are using microfiltration (MF) of secondary effluent followed by nanofiltration (NF) or reverse osmosis (RO) prior to subsurface injection into the aquifer. Drewes, Amy and Reinhard (2002) investigated advanced membrane treatment using NF and RO and found that membranes can efficiently reject high molecular weight organic matter. Approximately 40 to 50 per cent of the remaining TOC in permeates consisted of low molecular weight acids and neutrals representing a molecular weight range of ~500 Daltons and less. Based on carbon-13 NMR results and SEC-DOC analysis, NF and RO permeates still contained fulvic acid-related material that was not altered as compared to organic matter present in groundwater or surface water.

Emerging contaminants relevant to groundwater recharge will include: trace organics such as potential endocrine-disrupting compounds (EDCs), pharmaceutically active compounds (PhACs), and N-nitrosdimethylamine (NDMA); some trace inorganics; and microbes, for example nanobacteria (≈0.1 μm). Wastewater indicators, EDCs, and PhACs selected for study usually are not detected in either NF or RO permeates at pilot- and full-scale. These findings indicate that advanced membrane treatment using NF or RO not only efficiently removes high molecular weight organic carbon compounds, but also selected organic wastewater indicators, such as EDCs and PhACs (Drewes et al., 2002).

## 6.5 Risk assessment in groundwater recharge

Risk avoidance or risk minimization should certainly be principal elements in the development of drinking-water standards and guidelines. However, technological and economic factors must also be taken into account. Aesthetic factors of taste, odour and appearance must be important considerations, even if they do not directly relate to the safety of the water, because consumer acceptance of, and confidence in, the quality and safety of drinking-water are essential. If water is esthetically unacceptable, consumers may switch to other, uncontrolled, waters that are actually less safe.

Risk assessment is fundamentally an attempt to quantify the possible health consequences of human exposure in particular circumstances. In the case of drinking-water, the conclusion would be expressed in terms of the probability (within specified levels of uncertainty) of cases of adverse effects, such as fatalities, in the reference population group. For example, an incremental upper limit risk of bladder cancer of one per million in a population typically consuming 2 litres of drinking-water per day for 70 years (see "Stockholm framework" discussion in Appendix A). The lower limit risk might well be zero, especially if one or more assumptions are invalid. All such computations and "conclusions" are limited in their reliability and credibility by the quality of the exposure and toxicological data, the mathematical expressions used, and the lack of scientific understanding of the mechanisms of carcinogenesis operating at low environmental doses in genetically diverse humans, compared to the high doses to which test animals are exposed. In addition, little is known of the effects of interactions of low doses of chemicals (Cotruvo, 1988).

In its simplest terms a risk assessment could be represented as follows:

RA = concentration distribution × persons exposed at each dose × risk per dose × time

The basic information required to perform a quantitative risk assessment includes quantitative information on: the occurrence, human exposure and toxicology of the substance. Although methodologies are available to attempt to quantify each of these factors, in practice, data limitations and analytic complexities usually lead to many simplifying assumptions.

Comprehensive quantitative data on the frequency and concentration range of the contaminants in public drinking-water supplies are needed to determine the potential for human exposure under a variety of conditions, and to predict the exposure consequences of any control options being considered. The same analyses should be done for other water-related exposures (see "Stockholm Framework" discussion in Appendix A); and air, occupational and other contributions to total human exposure. Typically, drinking-water data are the most feasible to obtain, given the finite nature of drinking-water sources and the availability of analytical methods of high sensitivity and reliability. Statistically based surveys can be designed to generate high-quality data on national distributions of drinking-water contaminants by source type, population, season or other variables.

Computing human exposure from occurrence data requires detailed information on water and food consumption patterns, and other lifestyle factors that often are very difficult to model. These factors will be dependent on age, size, season and location. Water consumption has been studied in several countries and reasonable distributional data are available. For example, the average drinking-water consumption estimated from eight studies was 1.63 l/day. A dietary study (Ershow & Cantor, 1989) concluded that the median daily water consumption in the USA was 1.2–1.4 l/day, that 80–85% of people consumed less than 2 l/day and about 1% consumed more than 4 l/day. This included all tap water, including coffee, tea, and reconstituted juices, soups and food water (e.g. from rice). These estimates are probably low for very warm climates.

Dietary patterns are much more complex, and databases that can be extrapolated to populations are not very extensive. Localized ambient air inhalation data are available for a few substances. Indoor air quality data are potentially of greatest interest but are also limited. Water can contribute to indoor air exposure to volatile substances such as trihalomethanes or radon, or even *Legionella* organisms from growth in plumbing systems. This indirect exposure should be considered when projecting total exposure and the drinking-water contribution. For volatile organic compounds in drinking-water this inhalation dose can be equivalent to the amount from ingestion of water.

Toxicology assessments require highly qualified health scientists to analyse the complex data describing the toxicology of a substance, select the appropriate valid studies from the often conflicting or incomplete data, and arrive at a judgement on the relevance of the information to human health risks. The analyses should include all relevant toxicology, including whole animal acute, subchronic (90-day), and chronic (lifetime) studies, reproductive and developmental studies, neurotoxicology and other relevant animal data, in vitro studies of mutagenesis and cytogenetics, and also human epidemiology, if available.

The rough relationship between postulated levels of risk and the ability to identify cancer risk in the human population is illustrated in Figure 6.1. It is clear that the risks that are postulated for exposure levels typical to drinking-water are usually well beyond what epidemiological studies can measure. Since regulatory policy generally strives to limit risks below 1/100,000 for life threatening diseases like cancer, these lower risks are estimated by making inferences about the shape of the dose–response curve and extrapolations from effects to humans at higher doses or animal testing. Imperfect though this system is, it attempts to incorporate all of the available information and creating usable (albeit unverifiable) low dose risk hypotheses that can be helpful for decision making.

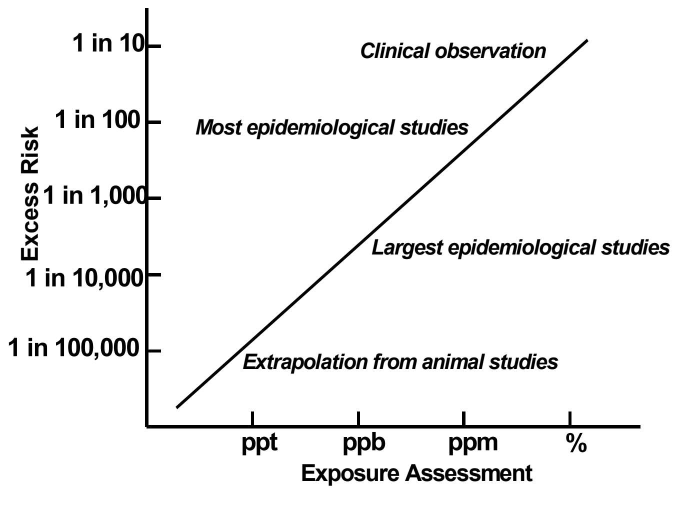


**Figure 6.1 Sensitivity of epidemiology in detecting risks of regulatory concern (NRC, 1993).**

Thus, WHO *Guidelines for Drinking-water Quality* (WHO, 1996) together with detection and evaluation methodologies aimed at source-specific contaminants, and site and technology-specific factors should be applied on a case-by-case basis. Such guidelines and techniques should be extended significantly to determine the design and operation of each specific project, to assure the suitability of the product water for its end use. They should also be expanded to include reuse and recharge applications. Indeed, the recommendations and methodologies described in the WHO *Guidelines for Drinking-water Quality* provide for appropriate authorities to make suitable water quality and safety determinations based upon the societal, economic and feasibility factors that bear upon the cost–risk–benefit balance that must be struck to assure access to water of both adequate quantity and quality.

A brief description of the type of process used by WHO and national regulatory agencies to determine acceptable concentrations of contaminants in drinking-water is provided in Appendix C (Cotruvo, 1999). The methodology is evolving and variations are commonly applied, but this appendix describes the basic thought processes that are involved.

## 6.6 Proposed State of California planned groundwater recharge and reuse criteria

The proposed California criteria for groundwater recharge with recycled municipal wastewater rightly reflect a cautious attitude toward short and long-term health concerns. The criteria rely on a combination of controls intended to maintain a microbiologically and chemically safe groundwater recharge operation. No single method of control would be effective in controlling the transmission and transport of contaminants of concern into and through the environment. Therefore, the criteria specify source control, wastewater treatment processes, water quality, recharge methods, recharge area, dilution, extraction well proximity, and monitoring wells. An illustration of this conservative approach for regulating planned reuse projects is given in Appendix C. The approach may need to be modified to deal with particular circumstances of water quantity, water quality, economic considerations and risk–benefit environments. However, the example given highlights the potential of this approach for a comprehensive and protective regulatory program.

These proposed groundwater recharge criteria have undergone several iterations since the early 1990s (Hultquist et al., 1991), and, while several refinements have been made to improve the criteria, many of the requirements specified in earlier drafts remain unchanged (see Table 6.1). More recent revisions emphasized dilution and monitoring of unregulated organics and groundwater.

## 6.7 Summary and conclusions

Groundwater recharge with recycled wastewater presents a wide range of technical and health challenges. Case histories of groundwater recharge with soil aquifer treatment in Morocco and Spain illustrate the degree of removal of contaminants that can be achieved by this type of treatment.

Risk assessment is important to quantify the possible health consequences of human exposure. However, the data needed to assess the exposure that might result from contaminants in recycled water are often lacking, and assessment of the risk is complex. The proposed California criteria for groundwater recharge with recycled municipal wastewater reflect a cautious attitude to health concerns. These criteria, which continue to be revised, illustrate the potential of this approach for a comprehensive and protective regulatory program.

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# 7 Public concerns and risk communication

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## 7.1 Introduction — the need for a communication policy

Management of groundwater affects many different aspects of social development and well-being; thus, a wide range of different groups are stakeholders in the management of water resources. Stakeholders include the public at large (e.g. residents and consumers), public servants (e.g. policy-makers and their coworkers, members of the legal system, health and environmental workers, institutional and organizational representatives) and the private sector (e.g. farmers, industrialists, workers and water managers).

When changes in water resource management policy are being considered, stakeholders have expressed several concerns, such as:

- environmental health concerns related to the continued availability of water for all community needs;
- water safety concerns (e.g. perception of a health hazard related to water use);
- potential threats to socioeconomic development, particularly in areas that are highly water dependent (e.g. irrigated agriculture and certain industrial processes)
- water pricing;
- demand management options.

Subgroups of stakeholders may have their own specific concerns, but health impact from changes to water resource management policy is an issue that cuts across all groups. This chapter therefore focuses mainly on concerns about health impacts, to provide guidance to policy-makers on how to deal with the associated communication challenge. It looks at the principles of assessing the risks that have to be communicated and planning a risk communication campaign.

## 7.2 Basic principles of risk communication

### 7.2.1 Introduction to risk communication

Risk communication on any environmental health issue can be defined as the professional method of understanding scientific and technological risk associated with an environmental policy decision, and the communication of this risk within a given sociopolitical structure. Risk communication was first recognised as a specific scientific issue in 1969. In the 1980s, a number of authors developed a new approach to risk communication that viewed risks according to their perceived threat to familiar social relationships and practices, rather than simply by threshold numbers defined by (public) acceptability (Vlek & Stallen, 1981; Douglas, 1986; Slovic, 1987).

Experience over recent decades has shown that a risk communication strategy needs to focus on understanding how the public perceives risk, how the media translate information received from scientists or public policy-makers, and how representatives of the public and private sector can better relate risk information over a wide range of disciplines. Intensive industrial development, especially of the chemical industry, has often required the communication of technological risk (Covello et al., 1988). In democratic societies, decision-making processes have increasingly involved the public as legitimate partners, often creating a 'risk' that the communication would be driven by nonexperts. In addition, the implementation of the right-to-know principle now enshrined in many national and international laws and regulations has meant that many assessment procedures now include public participation. This evolution created a need for a systematic approach to risk communication in public policy implementation.

Successful risk communication is a complex art that requires skill, knowledge, training and practice, as well as funding. An appropriate communication policy needs to be costed and should be part of any cost evaluation of an environmental resource management policy. Effective communication of risk needs to involve not only disseminating information but also communicating the complexities and uncertainties associated with risk assessment and management. Well-managed efforts will help to ensure that messages are constructively formulated, transmitted and received, and that they correspond to actions perceived to be meaningful and justified.

### 7.2.2 Definitions and perceptions

Since 1983, the model developed in the USA by the National Research Council (NRC) of the National Academy of Sciences (NAS) has explicitly distinguished between different stages of risk analysis. "Risk management" applies to assimilating nonscientific factors to reach a policy decision; "risk communication" applies to communicating a policy decision.

The very definition of "risk" varies depending on the user. Covello and Merkhofer (1994) define risk as "the possibility of an adverse outcome, and uncertainty over the occurrence, timing, or magnitude of that adverse outcome". Scientists generally define risk as the nature of the harm that may occur, the probability that it will occur and the number of people that will be affected (Groth, 1991). Most citizens, on the other hand, are concerned with broader, qualitative attributes, such as the origin of the risk (natural or technological), whether a risk is imposed or can voluntarily be assumed, the equitable distribution of risk over a population, alternatives and the power of individuals to control the risk (Sandman, 1987).

Perception of risk is also important. According to Covello (1992a; 1983), research has identified 47 known factors that influence the perception of risk, including such factors as perception of risk, control, benefits accrued from accepting risk and, most importantly, trust. Two basic considerations should drive risk communication (Covello et al. 1988; U.S. National Research Council, 1989):

- perception can change even if the actual risk does not;
- perception is the reality you have to deal with.

Fischhoff et al. (1981) and Covello and Merkhofer (1994) discuss risk perception in detail.

### 7.2.3 Breaking down barriers to risk communication

An inappropriate approach to risk communication can be a barrier to communication. Such barriers often originate in linguistic differences between scientists and laypersons. For example, members of the general public, particularly consumer organizations, often address risk communication in a "court context" (i.e. in an adversarial manner). Conversely, scientists often approach the topic with the aim of educating people; for example, explaining the scientific aspects of the matter but not actively listening and responding to legitimate concerns voiced by the lay audience. Thus, an unhelpful confrontation is generated between what Filshoff calls "opinion of experts against expert opinion". This further confuses the issues of trust and credibility in communication.

To compound matters further, there is a large and growing distrust of experts, and of science in general. A survey of 2000 doctoral students and 2000 university faculty from the largest graduate departments in chemistry, civil engineering, microbiology and sociology in the USA (Swasey, Anderson & Louis, 1993), found that:

- 6–9% of both students and faculty stated that they had direct knowledge of faculty who had plagiarized or falsified data;
- almost one-third of faculty claimed to have observed student plagiarism;
- 22% of faculty claimed to have witnessed instances of their colleagues overlooking sloppy use of data;
- 15% stated that they knew of cases where data that would contradict an investigator's own previous research had not been presented;
- 7–23% of both faculty and students claimed to have firsthand information about faculty misuse of research funds, unauthorized use of privileged information and conflicts of interest involving failure to disclose involvement in firms whose products are based on a faculty member's own research.

Credibility of communicators is mainly built by transferring and demonstrating knowledge about hazards and risks associated with the proposed policy, and by trust in public policy-makers. In this context it is worth noting that risk communication experts argue with the classical models of risk assessment, in which hazard identification in a part of the process. Covello and Merkhofer (1994) regard hazard identification as an altogether separate process that is of necessity conducted before risk assessment. These authors argue that treating hazard identification as merely one component of risk assessment underplays its importance. They also encourage the release of the hazard assessment as a separate step for important types of risk such as industrial accidents or failures involving large technological systems.

### 7.2.4 The risk management cycle

Soby et al. (1993) developed the concept of the risk management cycle in a review of risk communication research and its applicability, mainly in relation to food-related risks. This model, which calls for concerns of the public and other stakeholders to be actively sought at each stage of the management process, including assessment, is now widely adopted. A revised form is shown in Figure 7.1.

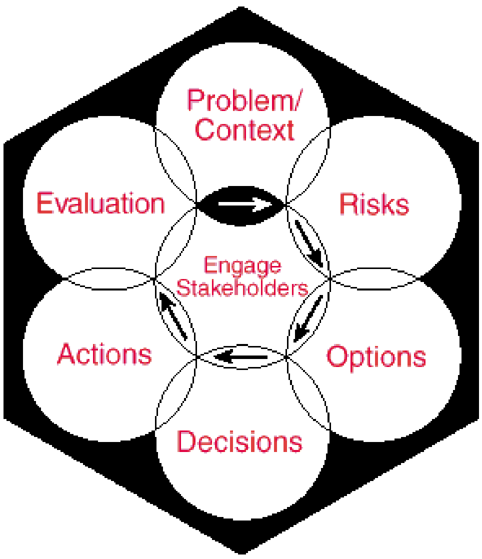


**Figure 7.1 Review of risk communication research and its applicability Source: Soby et al. (1993)**

Effective risk communication requires knowledge of the nature of the risk and of the benefits associated with acceptance of the risk, and knowledge of uncertainties in risk assessment and risk management, as outlined in Table 7.1. Communication programmes need to be tailor-made for specific policies, and adapted to the unique needs of specific audiences and concerns.

**Table 7.1 Effectiveness of risk communication**

| |
|---|
| The nature of risk:<br>▪ the characteristics and importance of the hazard of concern;<br>▪ the magnitude and severity of the risk;<br>▪ whether the risk is becoming greater or smaller (trends) and the urgency of the situation;<br>▪ the probability of exposure to the hazard and the distribution of exposure;<br>▪ the amount of exposure that constitutes a significant risk;<br>▪ the nature and size of the population at risk and who is at the greatest risk. |
| The nature of benefits:<br>▪ who benefits and in what ways<br>▪ the actual or expected benefits associated with each risk.<br>▪ the magnitude and importance of the benefits<br>▪ where the balance point is between risks and benefits. |
| Uncertainties in risk assessment:<br>▪ the methods used to assess the risk;<br>▪ the assumptions on which estimates are based;<br>▪ the importance of each of the uncertainties;<br>▪ the weaknesses of, or inaccuracies in, the available data;<br>▪ the sensitivity of the estimates to changes in assumptions;<br>▪ the effect of changes in the estimates on risk management decisions. |
| Risk management issues:<br>▪ the action(s) taken to control or manage the risk;<br>▪ the action individuals may take to reduce personal risk;<br>▪ the justification for choosing a specific risk management option;<br>▪ the effectiveness of a specific option;<br>▪ the benefits of a specific option;<br>▪ the cost of managing the risk, and who pays for it;<br>▪ the risks that remain after a risk management option is implemented. |

## 7.3 Dealing with public concerns

Risk communication in the context of implementing policy options requires communication goals to be set before the communication strategy is put into practice. Stakeholders concerned about health issues are likely to seek information on:

- the nature of the risk;
- the presence of effective and rapid surveillance systems;
- the existence of a credible, open and responsive regulatory system as a basis for policy implementation;

- effective communication on individual and community benefits;
- demonstrable efforts to reduce levels of uncertainty and risk;
- evidence that actions match words.

Statements addressing these concerns, and if possible providing information on achievements in these areas, constitute the core of any communication strategy.

Implementation of the communication strategy will be most effective if it follows a two-phased approach:

- Phase 1 — pre-assessment
  In this phase stakeholders are evaluated within their social context, to determine what their concerns are, how they perceive risk, whom they trust, etc. It may be useful to rank stakeholders and their concerns. For example, if the result of implementing the policy will be an increase in the price of water, and the main final use of the groundwater is agricultural irrigation, farmers will be the group most concerned, and communication strategies can be targeted to their specific needs. Pre-assessment campaigns may also usefully explore current understandings and risk perception, and test the communication channels chosen to reach the target group.
- Phase 2 — operation
  In this phase, attention will mainly focus on the ways and means available to implement the communication campaign, once a communication programme has been developed. It will involve organization of the people involved, selection of the communicators and selection of the range of communication channels to be used (e.g. Internet, pamphlets, public forums or presentations and newspaper articles).

The seven cardinal rules of risk communication, developed by Covello and Allen (1998) and shown in Box 7.1, have to be borne in mind at all stages.

**Box 7.1 Cardinal rules of risk communication**

- Accept and involve the public as a partner. The ultimate goal of the communication strategy is to produce an informed public, not to defuse public concerns or replace actions.
- Plan carefully and evaluate the outcome of the communication efforts. Different goals, audiences and media require different actions.
- Listen to the public's concerns. People often care more about trust, credibility, competence, fairness and empathy than about statistics and details.
- Be honest, frank and open. Trust and credibility are difficult to obtain; once lost, they are almost impossible to regain.
- Work with other credible sources. Conflicts and disagreements among organizations make communication with the public much more difficult.
- Meet the needs of the media. The media are usually more interested in politics than in risk, in simplicity than in complexity, and in danger than in safety.
- Speak clearly and with compassion. Never let efforts prevent acknowledgement of the tragedy of an illness, injury or death. People can understand risk information, but they may still not agree. Some people will not be satisfied.

Source: Covello & Allen (1998)

People's perception of the magnitude of the risk is influenced by factors other than numerical data and, as stressed earlier, in the public domain, perception equals reality. General principles on which factors influence risk perception are summarized below (Fischhoff et al., 1981):

- Risks perceived to be under an individual's control are more accepted than risks perceived to be controlled by others. Water resource management is typical for an activity perceived as being outside the individual's control.
- Risks perceived to have clear benefits are more accepted than risks perceived to have little or no benefits.
- Risks perceived to be voluntary are more accepted than risks perceived to be imposed (e.g. through public policy).
- Risks perceived to be fairly distributed are more accepted than risks perceived to be unfairly distributed.
- Risks perceived to be natural are more accepted than risks perceived to be human-made.
- Risks perceived to be statistical are more accepted than risks perceived to be catastrophic.
- Risks perceived to be generated by a trusted source are more accepted than risks perceived to be generated by a suspected source.
- Risks perceived to be familiar are more accepted than risks perceived to be exotic.
- Risks perceived to affect adults are more accepted than risks perceived to affect children.

## 7.4 Conclusion

A communication campaign should be based on the results of risk analysis, and should be adapted to the social context of the target group. Health concerns should be addressed mainly by showing the capacity of the health services in monitoring health impacts, and assessing health risks when there is a significant deviation from the health outcome baseline.

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# Appendix A:
# Stockholm framework

Following a major expert meeting in Stockholm Sweden, the World Health Organization (WHO) published the document *Water Quality — Guidelines, Standards and Health: Assessment of Risk and Risk Management for Water-related Infectious Disease* (Bartram, Fewtrell & Stenström, 2001). This document creates a harmonized framework for the development of guidelines and standards, in terms of water-related microbiological hazards. The framework involves:

- assessing health risks before setting health targets;
- defining basic control approaches;
- evaluating the impact of these combined approaches on public health status (Figure A1).

The framework allows countries to adjust guidelines to local social, cultural, economic and environmental circumstances; and to compare the associated health risks with the risks that may result from microbial exposures through wastewater use, drinking-water and recreational or occupational water contact. This approach requires that diseases be managed as a whole package and not in isolation. Disease outcomes from one exposure pathway, or from one illness to another, can be compared by using a common measure, such as disability adjusted life years (DALYs). Water-related guidelines developed by WHO in the future will accord with the framework.

This appendix contains a summary of the framework components (Table A1), and a discussion of DALYs and tolerable risk. For a more detailed discussion of the framework, see Fewtrell and Bartram, 2001.

**Table A1 Elements and important considerations of the Stockholm framework**

| Framework component | Process | Considerations |
|---|---|---|
| Assessment of health risk | Hazard assessment<br>Environmental exposure assessment<br>Dose-response analysis<br>Risk characterization | Best estimate of risk — not overly conservative<br>Equivalence between risk of infection and risk of disease<br>Health outcomes presented in disability adjusted life years (DALYs); facilitates comparison of risks across different exposures and priority setting<br>Risk assessment is an iterative process — risk should be periodically reassessed based on new data or changing conditions<br>Risk assessment is a tool for estimating risk and should be supported by other data (e.g. outbreak investigations, epidemiological evidence, microbiological risk assessment and studies of environmental behaviour of microbes)<br>Process depends on quality of data<br>Risk assessment needs to account for short-term under-performance |
| Tolerable risk/health targets | Health-based target setting based on risk assessment<br>Define water quality objectives | Need to be realistic and achievable within the constraints of each setting<br>Set using a risk-benefit approach; should consider cost-effectiveness of different available interventions<br>Should take sensitive subpopulations into account<br>Index pathogens should be selected for relevance to contamination, control challenges and health significance (more than one index pathogen may be needed) |
| Risk management | Based on health-based targets:<br>Define other management objectives<br>Define measures and interventions<br>Define key risk points and audit procedures<br>Define analytical verifications | Risk management strategies need to address rare or catastrophic events.<br>A multiple barrier approach should be used.<br>Monitoring — overall emphasis should be given to periodic inspection/auditing and to simple measurements that can be rapidly and frequently made to inform management.<br>Hazard analysis critical control point (HACCP)-like principles should be used to anticipate and minimize health risks. |
| Public health status | Public health surveillance | Need to evaluate effectiveness of risk management interventions on specific health outcomes (both through investigation of disease outbreaks and evaluation of background disease levels)<br>Establish procedures for estimating the burden of disease, to facilitate monitoring of health outcomes due to specific exposures<br>Burden of disease estimates can be used to place water-related exposures in the wider public health context, to enable prioritization of risk management decisions<br>Public health outcome monitoring provides the information needed to fine-tune risk management through an iterative process |

Source: Adapted from Bartram, Fewtrell & Stenström (2001)

## A1 Disability adjusted life years

DALYs are a measure of the health of a population or burden of disease due to a specific disease or risk factor. They attempt to measure the time lost through disability or death from a particular disease, by comparing it to a long life free of disability in the absence of the disease. DALYs are calculated by adding the years of life lost (YLLs) to premature death to the years lived with a disability (YLDs). YLLs are calculated from age-specific mortality rates and the standard life expectancies of a given population. YLDs are calculated from the number of cases multiplied by the average duration of the disease and a severity factor, which ranges from 1 (death) to 0 (perfect health), based on the disease. For example, watery diarrhoea has a severity factor ranging from 0.09 to 0.12, depending on the age group (Murray and Lopez, 1996a; Prüss and Havelaar, 2001). DALYs are an important tool for comparing health outcomes because they account not only for acute health effects but also for delayed and chronic effects, including morbidity and mortality (Bartram, Fewtrell & Stenström, 2001).

When risk is described in DALYs, different health outcomes can be compared (e.g. cancer can be compared to giardiasis) and risk management decisions can be prioritized.

## A2 What is an acceptable (tolerable) risk?

According to Hunter and Fewtrell (2001) the following criteria can be used to judge whether a risk is acceptable:

- it falls below an arbitrary defined probability;
- it falls below some level that is already tolerated;
- it falls below an arbitrary defined attributable fraction of total disease burden in the community;
- the cost of reducing the risk would exceed the costs saved;
- the cost of reducing the risk would exceed the costs saved when the "costs of suffering" are also factored in;
- the money would be better spent on other, more pressing public health problems;
- public health professionals say that it is acceptable;
- the general public say that it is acceptable (or more likely, do not say that it is not acceptable);
- politicians say that it is acceptable.

Tolerable risks are not necessarily static. As tools for managing water-related disease transmission improve, the levels of risk that are tolerable may decrease. Tolerable risks can therefore be set with the idea of continuous improvement. For example, smallpox and polio were eradicated because it was technologically feasible to do so, not because of the continually decreasing global burden of disease attributed to these pathogens.

The control of *Listeria monocytogenes* in ready-to-eat food products provides another example. Listeria causes listeriosis, a serious, although rare, foodborne disease; 99% of listeriosis is estimated to be foodborne (Mead et al., 1999). In the 1980s, effective management procedures were developed that could produce finished, ready-to-eat food products with either no *L. monocytogenes* present, or very low levels. The United States of America (USA) adopted a policy of "zero" tolerance for listeria in food products, as a result of which, the incidence of foodborne listeriosis in the USA declined by 44% and the mortality it caused declined by 49% over a period of four years

(Billy, 1997). In many developed countries, similar reductions in foodborne listeriosis have occurred due to (CODEX, 1999):

- adoption of good management practices and hazard analysis critical control point (HACCP) programmes by food processors;
- improvement in the integrity of the cold chain in food storage and transport, etc;
- better communication of risk to susceptible consumers

## A3 Tolerable microbial risk in water

For water-related exposures, WHO has determined that a disease burden of $1 \times 10^{-6}$ DALYs (i.e. one micro-DALY) per person per year from a disease (caused by either a chemical or an infectious agent) transmitted through drinking-water is a tolerable risk (WHO, 2003). This level of health burden is equivalent to a mild illness (e.g. watery diarrhoea) with a low fatality rate (e.g. 1 in 100 000) at an approximately 1 in 1000 annual risk of disease to an individual, which is equivalent to a 1 in 10 risk over a lifetime (WHO, 1996; Havelaar & Melse, 2003). In relation to *Giardia intestinalis* infection (rather than disease), the US Environmental Protection Agency (US EPA) sets a tolerable risk of less than 1 in 10 000 people per year (a $10^{-4}$ risk) from drinking-water (Regli et al., 1991). However, based on background rates of gastrointestinal disease in the general population, Haas (1996) argued that an acceptable risk of $10^{-4}$ of infection per person per year was too low, and that even a risk of $10^{-3}$ of infection per person per year would be too low.

The US EPA set the tolerable risk level using the risk of infection rather than the manifestation of disease. This is an important distinction, because there are a number of factors that determine whether infection with a specific pathogen will lead to a disease, including the virulence of the pathogen and the immune status of the individual (see Prüss and Havelaar, 2001 for a further discussion of infection versus disease). For example, hepatitis A infections in children are predominantly asymptomatic (have no apparent symptoms) but the same infection in adults often leads to disease symptoms (WHO, 2000). Asymptomatic infection can be confirmed by microbiological examination of stool specimens and, in some cases, by detection of a serological response (Teunis et al., 1996). However, infection is harder to detect in the general population because there are no obvious disease symptoms to track. For this reason, it is more difficult to measure compliance with, and enforce, a guideline value set with infection as an end-point than one based on disease. Also, a value based on infection is less precise in terms of public health protection.

Tolerable risk can be looked at in the context of total risk from all exposures; risk management decisions can then be used to address the greatest risks first. For example, if 99% of cases of salmonellosis were related to food, then halving the number of cases attributed to drinking-water would have very little impact on the disease burden.

For water-related exposures to microbial contaminants, diarrhoea or gastrointestinal disease is often used as to represent all waterborne infectious diseases. Mead et al. (1999) estimated that, including all age groups, the average person in the USA suffers from 0.79 episodes of acute gastroenteritis (characterized by diarrhoea, vomiting or both) per year, equivalent to a $7.9 \times 10^{-1}$ yearly risk. The rates of acute gastroenteritis among adults worldwide are generally within the same order of magnitude, as shown in Table A2. However, children (especially those living in high risk situations where poor hygiene, sanitation and water quality prevail) generally have more frequent gastrointestinal illnesses than adults. Kosek, Bern & Guerrant (2003) found that children under the age of five in developing countries experienced a median of 3.2 episodes of diarrhoea per child year, equivalent to a 3.2 yearly risk.

**Table A2 Diarrhoea cases per year by age group and country income level**

| Age | No. of diarrhoea cases per year<br>Country income level[a]<br>Low | Middle | High |
|---|---|---|---|
| 0–4 | 4.5–5.0 | 2.3–4.0 | 1.8 |
| 4–15 | 0.6–0.9 | 0.1–1.2 | 0.1 |
| 15–80+ | 0.2–0.3 | 0.2–0.3 | 0.1 |
| **Average** | 0.8–1.3 | 0.6–1.0 | 0.18–0.22 |

a As defined by the World Bank (www.worldbank.org/data/countryclass/countryclass.html)
Source: Adapted from Murray and Lopez (1996b)

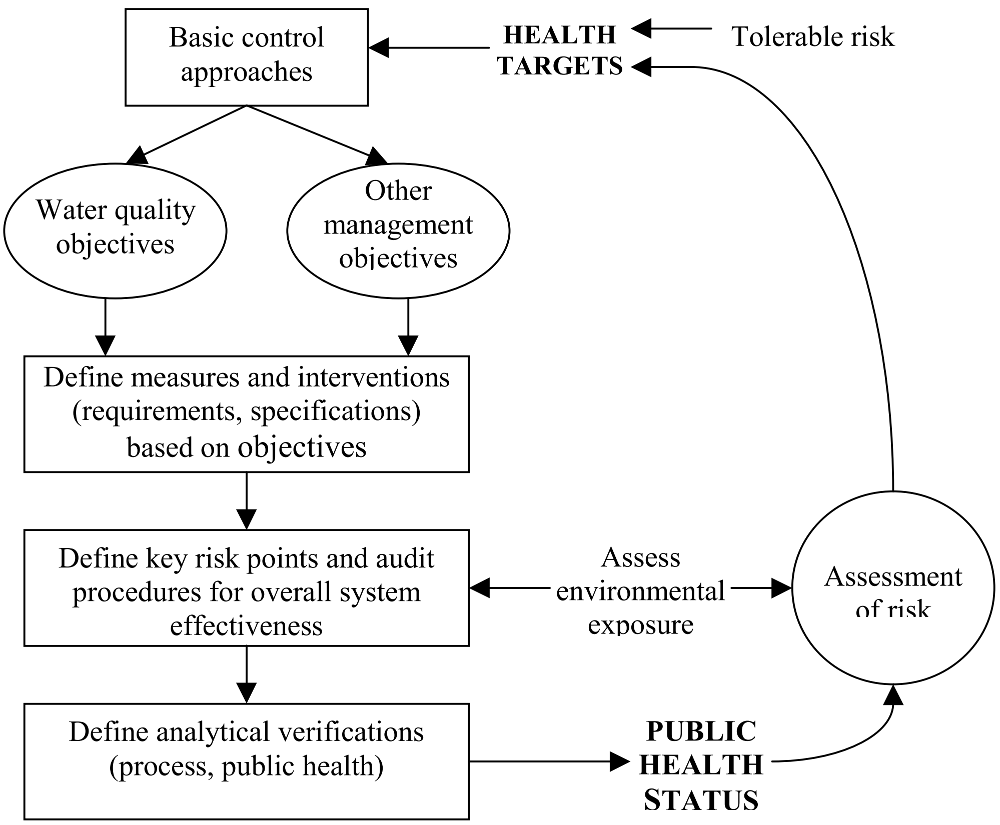


**Figure A1 Stockholm framework for assessment of risk for water-related microbiological hazards**

Source: Bartram, Fewtrell & Stenström (2001)

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## Appendix B:
## Balearic Islands — an example of regulation of wastewater reuse

A recent draft of Spanish regulation on wastewater reuse included some requirements on water used for aquifer recharge. The indicators of injectant quality taken into account are nematode eggs, *Escherichia coli*, suspended solids, turbidity and total nitrogen (IME, 1999). Artificial groundwater recharge by recycled treated wastewater had previously been addressed in the recommendations for water reuse issued by the government of the Balearic Islands, Spain in 1995 (Table B.1), one of the very few examples of such recommendations in Europe. The recommendations distinguish between potable and nonpotable aquifers.

**Table B.1 Summary of water reuse recommendations in Balearic Islands[a]**

| Requirements on the injectant | Requirements on the aquifer water |
|---|---|
| Groundwater recharge into nonpotable water aquifers by spreading, infiltration percolation or infiltration wells — protection against seawater intrusion | |
| pH = 6–9<br>$BOD_5$ < 40 mg/l<br>COD < 120 mg/l<br>SS < 60 mg/l<br>*E. coli* < 10 000/100 ml | No quality objective, with the exception of N and P limitations |
| Indirect potable water reuse — groundwater recharge by spreading or percolation | |
| pH = 6–9<br>$BOD_5$ < 40 mg/l<br>COD < 120 mg/l<br>SS < 60 mg/l<br>*E. coli* < 10 000/100 ml | Water should meet potable water standards after infiltration<br>Daily monitoring of pH, total and thermotolerant coliforms<br>Other potable water parameters monitored every 3 months<br>Enterovirus or bacteriophages controlled monthly (0/10 l)<br>Control on other organic and inorganic compounds, or classes of compounds, that are known or suspected to be toxic, carcinogenic, teratogenic or mutagenic and are not included in the potable water standards<br>Minimum detention time : 1 yr. |
| Indirect potable water reuse — groundwater recharge by injection wells | |
| Potable water quality required at the point of injection<br>Daily monitoring:<br>pH = 6.5–8.5, *E. coli* absent/100 ml<br>Continuous control:<br>$Cl_2$ residual = 0.5 mg/l (30 min contact), NTU < 2<br>Monthly monitoring:<br>No detectable enterovirus or bacteriophages in 10 l<br>All potable water standards monitored every 3 months<br>Control on other organic and inorganic compounds, or classes of compounds, that are known or suspected to be toxic, carcinogenic, teratogenic, or mutagenic and are not included in the potable water standards | Minimum detention time: 1 yr |

BOD = biochemical oxygen demand; COD =carbon oxygen demand; NTU = nephelometric turbidity unit; SS = suspended solids.
a Informe técnico sobre reutilización de aguas residuales. Consejería de Sanidad y Seguridad Social del Gobierno Balear, Nov. 1995

For nonpotable aquifers, the regulations summarized in Table B.1 make no explicit mention of the intended uses of the aquifer water (likely to be irrigation) or of any relationship between these uses and the quality required of the aquifer water. One may assume that the water quality is covered by regulations applying to wastewater reuse at the end of pipe.

The regulations covering indirect potable reuse are more consistent, with water quality required to be potable in the aquifer (after percolation through the vadose zone for indirect recharge) or in the injectant (for direct recharge).

# Appendix C:
# Process used to develop WHO guidelines for chemical drinking-water quality

## C1 Introduction

This appendix provides a summary of the methodology used in developing the World Health Organization (WHO) *Guidelines for Drinking-water Quality* (WHO, 1996 and 2003), to illustrate how numerical guideline values are generated for chemical contaminants. The primary aim of these WHO guidelines is the protection of public health. The guidelines are intended to be used as a basis for the development of national standards that, if implemented, will ensure the safety of drinking-water supplies by eliminating (or reducing to an acceptable concentration) constituents of water that are known to be hazardous to health.

Recommended guideline values are not mandatory limits. To define such limits, it is necessary to consider the guideline values in the context of local or national environmental, social, economic, and cultural conditions.

The main reason for not promoting the adoption of international standards for drinking-water quality is that there are advantages to using a risk–benefit approach (qualitative or quantitative) to the establishment of national standards and regulations. The standards that individual countries develop can thus be influenced by national priorities and economic factors. However, considerations of policy and convenience must never be allowed to endanger public health. It is up to each country to decide whether the benefit of adopting any of the guideline values as standards justifies the cost. The guideline values have a degree of flexibility and allow judgments to be made regarding the provision of drinking-water of acceptable quality.

The problems associated with chemical constituents of drinking-water arise primarily from their ability to cause adverse health effects after prolonged periods of exposure. Of particular concern are contaminants that have cumulative toxic properties, such as heavy metals and carcinogenic substances . There are few chemical constituents of water that can lead to acute health problems except through massive accidental or deliberate contamination of a supply. Moreover, experience shows that in such incidents the water usually becomes undrinkable owing to unacceptable taste, odour and appearance.

Guideline values have been set for numerous potentially hazardous water constituents and provide a basis for assessing drinking-water quality. A guideline value represents the concentration of a constituent that does not result in any significant risk to the health of the consumer over a lifetime of consumption.

The quality of water defined by the WHO *Guidelines for Drinking-water Quality* (WHO, 1996) is such that it is suitable for human consumption and for all normal domestic purposes, including personal hygiene. However, water of a higher quality may be required for certain uses, such as renal dialysis. When a guideline value is exceeded, this should be a signal to:

- investigate the cause with a view to taking remedial action;
- consult with, and seek advice from, the authority responsible for public health.

The establishment of guideline values does not imply that the quality of drinking-water may be degraded to the recommended level; every effort should be made to maintain drinking-water quality at the highest possible level. Also, short-term deviations above the guideline values do not necessarily mean that the water is unsuitable for consumption.

When a guideline value is exceeded, the surveillance agency (usually the authority responsible for public health) should be consulted for advice on suitable action. The agency should take into account the intake of the substance from sources other than drinking-water (for chemical constituents), the toxicity of the substance, the likelihood and nature of any adverse effects, the practicability of remedial measures, the population at risk and other similar factors.

National drinking-water standards based on the WHO guideline values will need to take into account a variety of geographical, socioeconomic, dietary and other conditions affecting potential exposure. These conditions may lead to national standards that differ appreciably from the guideline value. It is important that recommended guideline values are both practical and feasible to implement, as well as protective of public health. Guideline values are not set below concentrations that can be detected under routine laboratory operating conditions. Moreover, guideline values are recommended only when control techniques are available to remove or reduce the concentration of the contaminant to the desired level.

In some instances, provisional guideline values have been set for:

- constituents for which there is some evidence of a potential hazard, but where the available information on health effects is limited;
- substances for which the calculated guideline value would be below the practical level for quantification, or below the level that can be achieved through practical treatment methods;
- substances for which guideline values are likely to be exceeded as a result of disinfection procedures.

Aesthetic and organoleptic characteristics are subject to individual preference as well as social, economic, and cultural considerations. No guideline values have been set for such substances where they do not represent a potential hazard to health.

## C2 Assumptions in setting guideline values

A number of basic assumptions are made in setting guideline values. These include assumptions related to the following issues:

- drinking-water consumption and body weight;
- inhalation and dermal absorption;
- mixtures;
- health risk assessment;
- tolerable daily intake;
- uncertainty factors;
- allocation of intake.

The assumptions related to these issues are discussed below.

### C2.1 Drinking-water consumption and body weight

In studies carried out in Canada, the Netherlands, the United Kingdom, and the United States of America (USA), the average daily per capital consumption of water was usually found to be less than 2 litres, but there was considerable variation between individuals. Water intake is likely to vary with climate, physical activity and culture; therefore, the above studies, which were conducted in temperature zones, can give only a limited view of consumption patterns throughout the world.

In developing the guideline values for potentially hazardous chemicals, a daily per capita consumption of 2 litres by a person weighing 60 kg was generally assumed. The guideline values set for drinking-water using this assumption generally err on the side of caution. However, such an assumption may underestimate the consumption of water per unit weight (and thus exposure) for those living in hot climates, and for infants and children, who consume more fluid per unit weight than adults.

### C2.2 Inhalation and dermal absorption

The contribution of drinking-water to daily exposure includes both direct and indirect ingestion. Examples of indirect routes of injestion are inhalation of volatile substances and dermal contact during bathing and showering. That portion of the total tolerable daily intake (TDI) allocated to drinking-water is generally sufficient to allow for these additional routes of intake. When there is concern that potential inhalation of volatile compounds and dermal exposure from various indoor water uses (such as showering) are not adequately addressed, authorities could adjust the guideline value.

### C2.3 Mixtures

Any chemical contaminants of drinking-water supplies will be mixed with numerous other organic and inorganic constituents. The guideline values were calculated separately for individual substances, without specific consideration of the potential for interaction of each substance with other compounds present. The large margin of safety incorporated in the majority of guideline values is considered to be sufficient to account for such potential interactions. In addition, the majority of contaminants will not be present at concentrations at or near their guideline value.

There may, however, be occasions when a number of contaminants with similar toxicological effects are present at levels near their respective guideline values. In such cases, decisions concerning appropriate action should be made, taking into consideration local circumstances. Unless there is evidence to the contrary, it is appropriate to assume that the toxic effects of these compounds are additive.

### C2.4 Health risk assessment

Two principal sources of information on health effects resulting from exposure to chemicals can be used in deriving guideline values: human epidemiology and animal toxicology. Epidemiology is often limited due to lack of quantitative information on the concentrations to which people are exposed or on simultaneous exposure to other agents. Also, the tools of epidemiology are relatively insensitive to low-risk situations, due to confounders. Animal studies are generally limited because of the small number of animals used and the high doses administered. There is also need to extrapolate the results to the low doses to which human populations are usually exposed.

To derive a guideline value to protect human health, it is necessary to select the most suitable experimental animal study on which to base the extrapolation. Data from well-conducted studies, where a clear dose-response relationship has been demonstrated, are preferred, and expert judgement is essential to the process.

### C2.5 Tolerable daily intake

For most kinds of noncancer toxicity, it is generally believed that there is a dose to individuals below which no adverse effects will occur. For chemicals that give rise to such toxic effects, TDI can be derived as follows:

TDI = <u>NOAEL or LOAEL</u> (C1)

UF

where:

NOAEL = no-observed-adverse-effect level

LOAEL = lowest-observed-adverse-effect level

UF = uncertainty factor

The guideline value (GV) is then derived from the TDI as follows:

| | | |
|---|---|---|
| GV = | TDI × bw × P | (C2) |
| | C | |

where,

bw = body weight (60 kg for adults, 10 kg for children, 5 kg for infants),

P = fraction of the TDI allocated to drinking-water

C = daily drinking-water consumption (2 l for adults, 1 l for children, 0.75 l for infants).

The TDI is an estimate of the amount of substance in food or drinking-water, expressed on a body weight basis (mg/kg or μg/kg of body weight), that can be ingested daily over a lifetime without appreciable health risk. Principles established by the Joint Expert Committee on Food Additives (JECFA) and the Food and Agriculture Organization of theUnited Nations (FAO)/WHO Joint Meeting on Pesticide Issues (JMPR) have been adopted where appropriate in the derivation of TDIs.

Short-term exposure to levels exceeding the TDI is not a cause for concern, provided the individual's intake, averaged over longer periods of time, does not appreciably exceed the level set. The large uncertainty factors generally involved in establishing a TDI (see below) serve to provide assurance that exposure somewhat exceeding the TDI for short periods is unlikely to have any deleterious effects upon health. However, consideration should be given to any potential acute toxic effects that may occur if the TDI is substantially exceeded for short periods of time.

The calculated TDI was used to derive the guideline value, which is then rounded to one significant figure to reflect the uncertainties in toxicity data and exposure. In some instances, TDI values with only one significant figure set by JECFA or JMPR are used to calculate the guideline value. More than one significant figure was used for guideline values only where extensive information on toxicity and exposure to humans provided greater certainty.

### C2.6 Uncertainty factors

In the derivation of WHO drinking-water quality guideline values, uncertainty factors were applied to the lowest NOAEL or LOAEL for the response considered to be the most biologically significant, and were determined by consensus among a group of experts using the approach outlined in Table C.1.

**Table C1 Determination of uncertaintly factors**

| Source of uncertainty | Factor |
|---|---|
| Interspecies variation (animals to humans) | 1–10 |
| Intraspecies variation (individual variations) | 1–10 |
| Adequacy of studies or database | 1–10 |
| Nature and severity of effect | 1–10 |

Situations in which the nature or severity of effect might warrant an additional uncertainty factor include studies in which the end-point was malformation of a fetus or in which the end-point determining the NOAEL was directly related to possible carcinogenicity. In the latter case, an additional uncertainty factor was applied for carcinogenic compounds for which a guideline value was derived using a TDI approach. Factors lower than 10 were used, for example, for interspecies variation in cases where humans are known to be less sensitive than the animal species studies.

The total uncertainty factor should not exceed 10 000. If the risk assessment leads to a higher uncertainty factor, then the resulting TDI would be so imprecise as to lack meaning. For substances for which uncertainty factors were greater than 1000, guideline values are designated as provisional, to emphasize the high level of uncertainty inherent in these values. For contaminants for which there is relatively little uncertainty, the guideline value was derived using a small uncertainty factor. For most contaminants, however, there is great scientific uncertainty and therefore a larger uncertainty factor was used.

Hence, there may be a large margin of safety above the guideline value before adverse health effects result. The derivation of the uncertainty factor used in calculating a guideline value should be clearly presented as part of the rationale. This helps authorities in using the guidelines, because the safety margin in allowing for local circumstances is clear. It also helps to determine the urgency and nature of the action to be taken in the event that a guideline value is exceeded.

### C2.7 Allocation of intake

In many cases, the intake of a substance from drinking-water is small in comparison with that from other sources such as food and air. Guideline values derived using the TDI approach take into account exposure from all sources by apportioning a percentage of the TDI to drinking-water. This approach ensures that the TDI from all sources (including drinking-water containing concentrations of the substance at or near the guideline value) does not exceed the TDI level set as acceptable.

Wherever possible, guideline values were derived from data concerning the proportion of total intake normally ingested in drinking-water (based on mean levels in food, air and drinking-water) or intakes estimated on the basis of consideration of physical and chemical properties. Where such information was not available, an arbitrary (default) value of 10% for drinking-water was used. This default value is, in most cases, sufficient to account for additional routes of intake (i.e. inhalation and dermal absorption) of contaminants in water.

Exposure from different media may vary with local circumstances and conditions. The derived guideline values apply to a typical exposure scenario or are based on default values that may not be applicable for all areas. Where relevant data on exposure are available, authorities are encouraged to develop context-specific guideline values that are tailored to local circumstances and conditions.

## C3 Derivation of guideline values for potential carcinogens.

Evaluation of the potential carcinogenicity of chemical substances is usually based on long-term animal studies. Sometimes data are available on carcinogenicity in humans, mostly from occupational exposure. On the basis of the available toxicological evidence, the International

Agency for Research on Cancer (IARC) categorizes chemical substances with respect to their potential to be carcinogenic to humans into the following groups:

- Group 1 — the agent is carcinogenic to humans
- Group 2A — the agent is probably carcinogenic to humans
- Group 2B — the agent is possibly carcinogenic to humans
- Group 3 — the agent is not classifiable as to its carcinogenicity to humans
- Group 4 — the agent is probably not carcinogenic to humans

The initiating event in the process of chemical carcinogenesis is generally considered to be the induction of a mutation in the genetic material (deoxyribonucleic acid, DNA) of somatic cells (i.e. cells other than ova or sperm). Because this genotoxic mechanism theoretically does not have a threshold, there is a probability of harm at any level of exposure, albeit vanishingly small when the substance is present at extremely low levels. Therefore, the development of a TDI is considered inappropriate, and mathematical low-dose risk extrapolation is applied instead. On the other hand, some carcinogens are capable of producing tumours in animals or humans without exerting genotoxic activity, but acting through an indirect mechanism. It is generally believed that a threshold dose exists for these nongenotoxic carcinogens.

To make the distinction with respect to the underlying mechanism of carcinogenicity, each compound that has been shown to be a carcinogen was evaluated on a case-by-case basis, taking into account the evidence of genotoxicity, range of species affected, and the relevance to humans of the tumours observed in experimental animals. For carcinogens for which there is convincing evidence to suggest a nongenotoxic mechanism, guideline values were calculated using the TDI approach.

In the case of compounds considered to be genotoxic carcinogens, guideline values were determined using a mathematical model, and the guideline values are presented as the concentration in drinking-water associated with an estimated excess lifetime cancer risk of $10^{-5}$ (one additional cancer case per 100 000 of the population ingesting drinking-water containing the substance at the guideline value for 70 years). Concentrations associated with estimated excess lifetime cancer risks of $10^{-4}$ and $10^{-6}$ can be calculated by multiplying and dividing, respectively, the guideline value by 10. In cases in which the concentration associated with a $10^{-5}$ excess lifetime cancer risk was not practical because of inadequate analytical or treatment technology, a provisional guideline value was set at a practicable level and the estimated associated cancer risk presented.

A linearized, multistage model was generally adopted in the development of the guidelines, although other models were considered more appropriate in a few cases. Guideline values for carcinogenic compounds computed using mathematical models must be considered at best as a rough estimate of the cancer risk. These models do not usually take into account a number of biologically important considerations, such as pharmacokinetics, DNA repair or immunological protection mechanisms. However, the models used are conservative and probably err on the side of caution.

To account for differences in metabolic rates between experimental animals and humans — the former are more closely correlated with the ratio of body surface areas than with body weights — a surface area to body weight correction is sometimes applied to quantitative estimates of cancer risks, derived on the basis of models for low-dose extrapolation. Incorporation of this factor increases the risk by approximately one order of magnitude (depending on the species on which the estimate is based) and increases the risk estimated on the basis of studies in mice relative to that in rats. The incorporation of this factor is considered to be overly conservative, particularly in view of the fact that linear extrapolation most likely overestimates risk at low doses; indeed, Crump, Allen

& Shipp (1989) concluded that "all measures of dose except dose rate per unit of body weight tend to result in overestimation of human risk." Consequently, guideline values for carcinogenic contaminants were developed on the basis of quantitative estimates of risk that were not corrected for the ratio of surface area to body weight.

## C3 References

Crump K, Allen B, Shipp A (1989). Choice of dose measures for extrapolating carcinogenic risk from animals to humans: an empirical investigation of 23 chemicals. *Health Physics*, 57(Supp. 1):387–393.

WHO (1993). *Guidelines for drinking water quality.* 1st ed. Geneva, World Health Organization.

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WHO (2003). *Guidelines for drinking-water quality*, 3rd ed. Geneva, World Health Organization.

# Appendix D:
# Summary of proposed State of California criteria for groundwater recharge and reuse projects[12]

## D1 Source control

Concerns have been raised about the discharge of industrial pollutants not normally removed by municipal wastewater treatment processes. A well-operated and strictly enforced source control program will contain these pollutants onsite at the point of generation for proper disposal. The program will prevent these contaminants from entering the municipal wastewater treatment plant by severely limiting, restricting or prohibiting the discharge of these compounds into municipal wastewater treatment plants. The proposed criteria require a comprehensive program for the control of toxic wastes from point sources, which must be approved by the Regional Water Quality Control Board are shown in Table D1.

12 This appendix was derived from draft regulations dated April 23, 2001 and a publication by Crook, Hultquist, Sakaji and Wehner (2002) based on the background document developed by the State of California, Department of Health Services.

**Table D1 Proposed requirements for groundwater recharge with recycled water**

| | Type of recharge | |
|---|---|---|
| **Contaminant type** | **Surface spreading** | **Subsurface injection** |
| Pathogenic microorganisms | | |
| Secondary treatment | ≤ 30 mg/l | |
| Filtration | ≤ 2 NTU | |
| Disinfection | 4-log virus inactivation, ≤ 2.2 total coliform per 100 | |
| Retention time underground | 6 months | 12 months |
| Horizontal separation | 153 m | 610 m |
| Regulated contaminants | Meet all drinking-water maximum contaminant levels | |
| Unregulated contaminants | | |
| Secondary treatment | BOD ≤ 30 mg/l, TOC ≤ 16 mg/l | |
| Reverse osmosis | Four options available (see Chapter 6, Fig. 6.1) | 100% treatment to $TOC \leq \frac{1 mg/L}{RWC}$ |
| Spreading criteria for SAT 50% TOC removal credit | Depth to groundwater at initial percolation rates of: | NA |
| | <0.2 in/min = 10 ft. | |
| | <0.3 in/min = 20 ft. | |
| Mound monitoring option | Demonstrate feasibility of the mound compliance point | NA |
| Recycled water contribution | ≤ 50 % | |

BOD = biochemical oxygen demand; NA = not applicable; NTU = nephelometric turbidity unit; RWC = the percent recycled water contribution in groundwater extracted by drinking-water wells; SAT = soil aquifer treatment; TOC = total organic carbon.

## D2 Treatment processes

The definition of "filtered disinfected wastewater" in the proposed revisions to the existing regulations for nonpotable uses of recycled water now includes the use of membranes to meet the filtration requirements. This includes, and does not distinguish between, microfiltration (MF), ultrafiltration (UF), nanofiltration (NF) and reverse osmosis (RO). Although the performance requirement for membranes (average of 0.2 nephelometric turbidity units, NTU) is more stringent than that for granular medium filtration (2 NTU), the work done by the City of San Diego, California indicates that a filtered wastewater turbidity of more than 0.1 NTU signals a breach in the integrity of the membranes. An extra 0.1 NTU performance margin was given to allow for variations in turbidity measurements and associated sources of interference, such as sloughing from pipe walls.

Also included in the definition of filtered disinfected wastewater is the requirement that the wastewater be oxidized to a total organic carbon (TOC) concentration of 16 mg/l or less. The current California Wastewater Reclamation Criteria defines "oxidized wastewater" as "wastewater in which the organic matter has been stabilized, is nonputrescible, and contains dissolved oxygen". The TOC requirement of 16 mg/l is a performance-based water quality standard. A survey of several existing wastewater treatment plants associated with groundwater recharge projects indicated that a TOC concentration not exceeding 16 mg/l should be easily met by a well-operated wastewater treatment plant.

To address the issue of unregulated organics, the previous drafts of the proposed criteria allowed the use of granular activated carbon (GAC) or RO for organics removal. Although these processes can be complementary in terms of the fraction of organics they remove, GAC is generally regarded as being less efficient than RO for organics removal. RO provides a physical barrier to restrict the passage of molecules, whereas removal by GAC depends on the interaction between the solute and the GAC surface. Both processes depend on water chemistry and solute properties (ionic radius, hydrophobicity of the solute or membrane, or conformation) for removing organics; however, the interaction between the solute and GAC surface depends more on the interaction between the solute and the adsorbent, rather than providing a physical barrier to transport. RO provides the additional benefit of removing inorganics, which GAC does not. Consequently, the proposed groundwater recharge regulations reflect the conclusion that GAC alone is not effective for controlling unregulated organics.

## D3 Disinfection

The disinfection requirement in the proposed California regulations, for nonpotable reuse where a high degree of public exposure is expected, is also required for all groundwater recharge projects. This is because disinfection is effective in reducing the concentration of viruses, the only pathogenic microorganisms not effectively removed by the aquifer. Many groundwater recharge projects also provide nonpotable water for other urban uses, and the disinfection requirement is readily achievable with the reclamation technologies commonly in use in California. The two options for compliance are:

- filtration followed by chlorination with a modal chlorine contact time multiplied by the chlorine residual of 450 mg/min $l^{-1}$;
- any combination of filtration and disinfection that has been demonstrated, and is operated, to achieve a 5-log virus reduction.

## D4 Water quality

While the application of an organics removal requirement would appear to solve a plethora of water-quality issues, several issues remain, such as that of the nitrogen requirement. A conservative total nitrogen standard of 10 mg N/l was proposed, to ensure that, should all ammonia forms of nitrogen be converted to nitrate, the effluent nitrate concentration would approach, but never exceed, the nitrate maximum contaminant level (MCL). Dilution underground is not considered to be a reliable method for controlling the nitrogen content of the water for a chemical that poses such an acute public health threat. Therefore, the total nitrogen standard must be met above ground.

The main issue is the nitrite drinking-water MCL of 1 mg N/l. Since biological nitrification and denitrification processes produce nitrite as an intermediate product, it is not known how protective the 10 mg N/l standard would be of the nitrite MCL. As the oxidizing and reducing environment under which nitrite is formed and can persist in the environment appears to be relatively narrow, it is not anticipated that his will be a major issue. However, the current literature provides very little information on the microbial ecology and environmental conditions that might lead to a potential nitrite problem. It is difficult to develop specific criteria to ensure that such conditions are prevented from occurring or excluded from potential recharge sites. This issue is currently under investigation.

## D5 Dilution and unregulated organics

The draft criteria use the percent of the drinking-water supply that comes from recycled municipal wastewater as a factor in determining the required degree of unregulated organic removal. This fraction is the recycled water contribution (RWC). The previous drafts set separate organic chemical removal requirements for subsurface injection and surface spreading projects with a 20% RWC and a 50% RWC. Four treatment goals for organics removal have been provided: one for subsurface injection projects and three for surface spreading operations.

Subsurface injection projects are required to treat 100% of the recycled water by RO to provide organics removal. The goal of organics removal is to achieve a TOC concentration that is:

$$\mathrm{TOC} \leqslant \frac{1\,\mathrm{mg\,TOC/L}}{\mathrm{RWC}} \qquad \text{(D1)}$$

where:

RWC = the percent recycled water contribution in groundwater extracted by drinking-water wells.

For all surface spreading operations, the degree of organics removal required before recharge is dictated by the project sponsor’s ability to meet all the spreading criteria before the recharged water reaches the native groundwater (see Figure D1). If all the spreading criteria are not met, and monitoring in the groundwater mound before the recycled water is mixed with the native groundwater) is not possible, a project sponsor could provide RO treatment that would produce a recycled water meeting a TOC performance concentration that is less than or equal to the TOC concentration resulting from Equation C1. However, if a project sponsor could develop a program to monitor the TOC concentration in the groundwater mound before the recycled water reached the native groundwater, then RO treatment to produce a recycled water that contains a TOC concentration as shown in Equation D2 before recharge would be acceptable.

$$\mathrm{TOC} \leqslant \frac{1.5\,\mathrm{mg\,TOC/L}}{\mathrm{RWC}} \qquad \text{(D2)}$$

In addition, the recycled water leaving the groundwater mound would be required to contain a TOC concentration that was equal to or less than the TOC concentration given by Equation D1.

The final unregulated organics treatment option is used if a proposed project meets all the recharge criteria for surface spreading. In this case, the treatment train would apply RO treatment as needed to provide a TOC concentration as shown in Equation D3.

$$\mathrm{TOC} \leqslant \frac{1\,\mathrm{mg\,TOC/L}}{(1\quad \mathrm{SAT\,TOC\,removal})\quad (\mathrm{RWC})} \qquad \text{(D3)}$$

where:

SAT TOC removal = the percent TOC removal credit given to the soil aquifer treatment system.

For surface spreading projects meeting all the criteria, the soil aquifer system is assumed to achieve 50% removal of TOC, as discussed later in this paper.

The proposed criteria now contain one set of requirements (in a continuum) for projects with a RWC up to 50%. Although there are provisions for allowing a RWC of up to 100%, the criteria establish, in effect, a dilution requirement for most groundwater recharge reuse projects. The rationale for maintaining this dilution requirement has not changed.

An alternative to the 50% maximum RWC criterion is proposed that will assure an equal level of public health protection. The project must demonstrate the effectiveness of the alternate criterion in reducing all the potentially harmful components of wastewater TOC. It must also demonstrate the practically of the criterion as a regulation to support its eventual inclusion in the criteria. There must be objective measures for determining when a project is or is not in compliance. A regulation must also be designed such that all projects in compliance are providing the intended public health protection. During the demonstration phase, the project must provide evidence that the public is not at risk. Some combination of water quality monitoring and incremental increases in the RWC may accomplish this.

While it is acknowledged that the 50% dilution requirement does not provide an order of magnitude reduction in the long-term risk of chemical exposure, it does ameliorate the impact of variations in effluent water quality. There is no treatment process for organic chemical removal that effectively removes all classes of potentially harmful organics.

Dilution is an effective barrier to all the unidentified contaminants in wastewater. Dilution also provides a margin of safety against the discharge of unknown and unwanted organics, such as pharmaceutically active compounds, whose impacts on the microbial ecology and potential human health implications was not considered by a science advisory panel (State of California, 1987).

In their summary report, the advisory panel:

- acknowledged that analytical chemistry was not capable of routinely identifying organic substances in the part per trillion levels that might occur;
- acknowledged that the toxicology of those compounds could not, and probably never would, be precisely characterized in the part per trillion range;
- concluded that dissolved organic carbon would be removed to "…below 1 mg/l by reverse osmosis and essentially all identifiable trace organic compounds of significance should be absent in detectable concentrations."

While this level of removal may be adequate for most organic compounds that produce an adverse impact on humans, the observation does not consider a class of compounds of growing concern that may not produce a recognized toxicological health effect (e.g. carcinogenic, mutagenic or teratogenic) on humans. This group of compounds, referred to as pharmaceutically active compounds, may contain antibiotics, hormones, anti-inflammatories, endocrine homologs or other constituents. Of most concern are the antibiotics, because the low concentrations found in discharges may allow microorganisms, some of which may be opportunistic pathogens, to develop resistance to these drugs. In fact, researchers are beginning to use bacterial resistance to antibiotics (multiple antibiotic resistance) as a means of associating bacteria with potential sources. Anecdotally, it has been observed that bacteria isolated in the environment near wastewater treatment plant discharges appear to be resistant to a greater range of antibiotics than those located some distance from the discharges. While the cause and long-term consequences have not been evaluated, the observation does raise the issue of the long-term impacts of low level discharges of pharmaceutically active compounds on the microbial ecology, which may in turn lead to potential human health problems.

### D5.1 Percolation rates

To ensure the control of unregulated organics in surface spreading operations, the initial percolation rates are still tied to a required depth-to-groundwater (i.e. the unsaturated soil between the bottom of the spreading basin and the top of the groundwater mound). Based on a preliminary review of work underway and historical information, the removal of organics by the soil mantle is credited with 50% TOC removal and is not applicable to subsurface injection projects.

The use of streambeds in one proposed surface spreading operation raised issues with respect to percolation rates. Drop structures disturb the streambed, and percolation rates in the region immediately downstream of the drop structures were higher than allowed due to scouring of the streambed bottom by the water falling over the drop structure. Consequently, concerns were raised that the nonhomogenous percolation rates in these areas were too great to provide adequate soil treatment, which had been demonstrated only at the lower percolation rates. As an alternative, the project proponents will be allowed to monitor in the groundwater mound when surface spreading recycled municipal wastewater to ensure compliance with the organics removal requirement before the water leaves the mound.

### D5.2 Groundwater mound monitoring

Originally, based on work conducted by the County Sanitation District of Los Angeles County, the soil mantle was credited with organics removal in surface spreading projects. There have been concerns that there were no means of evaluating the performance of the soil mantle in-situ during treatment. As noted previously, mound monitoring is allowed as a compliance point to meet the organics removal requirement. However, project proponents must provide physical evidence that the sampling system is capable of providing a representative sample of the groundwater mound. Based on a preliminary review of new groundwater monitoring systems, the California Department of Health Services (DHS) will allow the use of continuous low-flow sampling for the collection of compliance samples from the mound. DHS stakeholders group has recommended that any anticipated soil aquifer treatment should be verified by monitoring.

Continuous low-flow dedicated bladder-pump systems have been designed and built for groundwater monitoring specifically to avoid sample collection and handling problems that have plagued the traditional purge, pump and collect systems. It is presumed that, due to water quality differences between the recharge water applied and the native groundwater, the elevated water surface will contain only water from the spreading basin. Using the new sampling technology it is possible to collect water quality samples from the mound above the point at which it mingles with the native groundwater. This allows for compliance with TOC and other water quality standards at a point within the groundwater basin without interference from the native groundwater.

### D5.3 Time in the underground and horizontal separation

The distance between the edge of a recharge basin or point of injection and a water supply well is defined as the horizontal separation. The term, “retention time underground: refers to the period of time the recharged water remains underground en route to the well. The purpose of establishing these two criteria was to ensure minimal migration of viruses through the soil system and to allow time for the natural die-off or attenuation of viruses to take place. A minimum residence time underground was established with a minimum horizontal separation distance to control those cases where the time of travel calculation showed that a short distance was sufficient. While studies have shown that natural attenuation of viruses in the environment takes place in subsurface systems, other studies have shown viruses to be capable of migrating long distances in the underground. When the observed attenuation of viruses is based primarily on environmental samples, it is not known whether the attenuation was due to some form of inactivation or dilution from extraneous underground water sources. As outlined in Table D.1, the horizontal separation and retention time requirements have been simplified to 153 m and 6 months for surface-spreading projects and 610 m and 12 months for injection projects, eliminating the horizontal separation requirement of 305 m and the retention time requirement of 12 months specified in earlier versions of the draft regulations for some categories of surface-spreading operations.

### D5.4 Monitoring wells

The proposed regulations require the installation of monitoring wells between the groundwater recharge project and the nearest potable water supply well. The monitoring wells are to be located at one quarter and one half of the distance between a spreading basin or injection well and the closest water supply well (as determined by physical distance). This will allow for the collection of water quality samples to assess the impact of previously unknown water quality problems from constituents such as N-nitrosodimethylamine.

### D5.5 Alternatives

DHS recognizes that there may be methods of achieving the public health goals of the groundwater recharge criteria that were not considered and are not allowed by the specific requirements. A section will be included in the criteria, authorizing alternatives for many of the criteria when it has been demonstrated that the alternative is as effective and reliable in reducing the public exposure to contaminants. In addition, although DHS recognizes the value of toxicological testing, it concurs with the finding in a recent publication (National Research Council, 1998) on potable reuse that states "The requirements for toxicological testing of water derived from an alternative source should be inversely related to how well the chemical composition of the water has been characterized. If very few chemicals or chemical groups or concern are present, and the chemical composition of the water is well understood, the need for toxicological characterization is lowered and may be safely neglected altogether."

## D6 References

National Research Council (1998). *Issues in potable water reuse : the viability of augmenting drinking water supplies with recycled water*. Washington DC, National Academy Press.

## Glossary

**aquifer**: a geological area that produces a quantity of water from permeable rock.

**biofilm:** microbial populations that grow on the inside of pipes and other surfaces.
**biotest**: biological testing using test organisms (in vivo) or modified cell tissues (in vitro).

***Cryptosporidium***: A protozoan commonly found in lakes and rivers that is highly resistant to disinfection. *Cryptosporidium* has caused several large outbreaks of gastrointestinal illness, with symptoms that include diarrhoea, nausea and stomach cramps. People with severely weakened immune systems (ie severely immunocompromised people) are likely to have more severe and more persistent symptoms than healthy individuals (adapted from United States Environmental Protection Agency).

**cyanobacteria**: Bacteria containing chlorophyll and phycobilins, commonly known as 'blue-green algae'.

**disinfection:** the process designed to kill most microorganisms in water, including essentially all pathogenic (disease-causing) bacteria. Chlorine is the disinfection method most frequently used in water treatment.

**disinfection by-product**: products of reactions between disinfectants (particularly chlorine) and naturally occurring organic material.

**direct recharge**: injection of water into an aquifer via injection wells.

***Escherichia coli***: a bacterium found in the gut, used as an indicator of faecal contamination of water.

**exposure**: contact of a chemical, physical or biological agent with the outer boundary of an organism (eg through inhalation, ingestion or dermal contact).

**exposure assessment**: the estimation (qualitative or quantitative) of the magnitude, frequency, duration, route and extent of exposure to one or more contaminated media.

***Giardia lamblia***: a protozoan frequently found in rivers and lakes. If water containing infectious cysts of *Giardia* is ingested, the protozoan can cause a severe gastrointestinal disease called giardiasis.

**groundwater**: water contained in rocks or subsoil.

**hazard**: a biological, chemical, physical or radiological agent that has the potential to cause harm.

**hazard analysis critical control point (HACCP) system**: a systematic way to control safety hazards in a process, by first identifying hazards, their severity and likelihood of occurrence; then identifying critical control points and their monitoring criteria to establish controls that will reduce, prevent, or eliminate the identified hazards.

**health impact assessment**: the estimation of the effects of any specific action (plans, policies or programmes) in any given environment, on the health of a defined population.

**heterotrophic plate count (HPC)**: the number of colonies of heterotrophic bacteria grown on selected solid media at a given temperature and incubation period, usually expressed in number of bacteria per millilitre of sample.

**indicator**: a specific contaminant, group of contaminants or constituent that signals the presence of something else (eg *Escherichia coli* indicate the presence of pathogenic bacteria).

**indicator organisms**: microorganisms whose presence is indicative of pollution or of more harmful microorganisms.

**indirect recharge**: involves spreading surface water on land so that the water infiltrates through the vadose zone (the unsaturated layer above the water table) down to the aquifer.

**multiple barriers**: use of more than one preventive measure as a barrier against hazards.

**pathogen**: a disease-causing organism (eg bacteria, viruses and protozoa).

**pH**: an expression of the intensity of the basic or acid condition of a liquid (natural waters usually have a pH between 6.5 and 8.5).

**raw water**: water in its natural state before any treatment; or the water entering the first treatment process of a water treatment plant.

**reference dose**: 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**: the likelihood of a hazard causing harm in exposed populations in a specified time frame, including the magnitude of that harm.

**risk assessment**: the overall process of using available information to predict how often hazards or specified events may occur (likelihood) and the magnitude of their consequences (adapted from AS/NZS 4360:1999).

**risk management**: the systematic evaluation of the water supply system, the identification of hazards and hazardous events, the assessment of risks, and the development and implementation of preventive strategies to manage the risks.

**source water**: water in its natural state, before any treatment to make it suitable for drinking.

**surface water**: all water naturally open to the atmosphere (eg rivers, streams, lakes and reservoirs).

**toxicology**: study of poisons, their effects, antidotes and detection.

**turbidity**: the cloudiness of water caused by the presence of fine suspended matter.

**vadose zone**: the unsaturated layer above the water table.

**wastewater reclamation**: the treatment or processing of wastewater to make it reusable.

**wastewater (or water) reuse**: the beneficial use of treated water.