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Developing a foundation for reclaimed water use in Scotland

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Key Findings

- Guidelines for reclaimed water in other countries focus on safety for human health and the environment. They differ substantially in the end uses considered, the approach taken, and the number and types of parameters considered and their associated limit values.
- Many regulations and guidelines state that they are based on a risk assessment, but it is often not clear on what basis water quality parameters have been chosen and how the associated limits have been set.
- A review of potentially harmful agents present in final effluent has identified those likely to pose the greatest threat to human health (and the environment). A vast number may be present in final effluent at a range of concentration levels depending on the type and size of the wastewater catchment and the degree of treatment. Three groups of hazards are considered here: pathogenic microorganisms, potential toxic elements and organic contaminants.
- For all considered scenarios, pathogenic microorganisms are found to be the main human health concern, suggesting that a disinfection treatment would be required before use of effluent. End-use scenarios considered included urban irrigation (edible/non-edible); agricultural irrigation; domestic; and industrial.
- Most of the other considered hazards pose limited risks to human health.
- The irrigation scenarios present a higher risk (especially for consumption of raw vegetables) compared to the domestic and industrial scenarios. Workshops captured stakeholder views on benefits, savings, perceived risks and barriers to the use of reclaimed water in Scotland and raised concerns over governance and accountability.
- Stakeholders are sceptical about the readiness of Scotland for water reuse projects, largely due to the perceived absence of economic, legislative and resource scarcity drivers.
- Opportunity mapping was identified as an essential next step. Favourable conditions may already be present and pinch-points in the current system need to be identified to inform whether and in what form guidelines on effluent reuse are needed.

Background

Water resources in Scotland and across the EU are under increasing stress, especially in areas with intense agriculture. Global climate changes are likely to exacerbate water shortages and cause an increasingly unpredictable supply. Consequently, there is an increasing interest in wastewater reclamation. Wastewater reclamation has the potential to conserve freshwater by reducing abstraction rates for irrigation and the amount of drinking water used for non-potable purposes. In addition, reclamation has the potential to reduce energy requirements and costs for wastewater treatment and provides an opportunity to recover valuable nutrients. These would otherwise be discharged to the aquatic environment with potential for eutrophication. Despite the benefits, reusing treated wastewater has seen limited development across the EU. Barriers include: risks to public health and the environment, cost, technical and practical implementation challenges, lack of public support, and concerns over trade barriers on agricultural goods from land irrigated with effluent.

Research Undertaken

This project aims to develop a foundation for reclaimed water use in Scotland. The focus is on the intentional reuse of final effluents, which here is defined as the treated final effluent from municipal/urban wastewater treatment plants. The research has been undertaken in two stages: 1) supporting the development of national guidelines in Scotland with a review of existing guidelines in other countries, and by using risk assessment tools to help develop reclaimed water standards that will ensure public health; 2) engaging with stakeholders to evaluate the models developed in Stage 1 and to identify benefits, savings, risks and barriers to use of reclaimed water in Scotland.

A series of potential end-use scenarios for reclaimed water relevant to Scotland have been developed. For each scenario, exposure and risk assessment models have been developed to assess potential human health impacts.

Recommendations

The following are recommended as prerequisites for the development of a set of national guidelines on water reuse in Scotland.

I. Decide, in consultation with stakeholders, which underlying approach would best-suit Scotland. Should the standards be legally-binding in an approach similar to that of Spain or California; a more flexible risk management approach of Australia; or a completely different approach?

Scenario	Description
Urban irrigation (non-edible)	Irrigation of non-edible plants, e.g. golf courses, parks, road verges, nursery stock, etc.
Urban irrigation (edible)	Irrigation of urban agricultural areas such as allotments and gardens
Agricultural irrigation	Irrigation of agricultural crops. Agricultural irrigation could also include watering livestock where potable-quality water is not necessary.
Non-potable domestic	Non-potable uses within the domestic environment for example, toilet flushing.
Industrial (uncontained)	Industrial processes where personnel are likely to come into contact with the water and will wear personal protective equipment, for example, vehicle washing.

II. Identify the end uses of effluent to be covered in the guidelines and those end uses to be prohibited.

III. The types of wastewater to be covered in the guidelines need to be clearly defined. Here, final effluent from wastewater treatment plants was considered, but other sources of wastewater could be included.

IV. Agree on which water quality parameters and other indicators to monitor, their associated limit values and whether/how this can be tailored to specific end uses. As wastewater effluent contains a mix of hazards at a broad range of concentration levels, it is impractical to monitor them all.

V. The current research has focussed primarily on the human health risks associated with effluent reuse. Future research should investigate and assess potential risks to the environment associated with different effluent reuse scenarios.

VI. Other initiatives and instruments will be required to promote and establish effluent reuse in Scotland. Continuing to develop an open and meaningful dialogue with stakeholders is critical particularly as there is currently no consensus in Scotland about the necessity for guidelines in this area.

VII. In light of the absence of an overarching scarcity driver, there is a clear call for opportunity mapping to be undertaken to identify under what circumstances water reuse is appropriate and to identify favourable conditions that may already be present as pinch-points in the current system.

Cost benefit analyses would be a useful way to identify which potential end uses are most viable from an economic point of view and would identify areas of worthwhile investigation. This additional research would underpin further work to develop a foundation for water reuse in Scotland.

1 Introduction

Wastewater reclamation has been practiced for many years across the world, particularly in arid regions where there is a clear incentive to use scarce water resources effectively. Within the EU, recycling of treated wastewater is mainly practiced in the southern European countries, notably Spain, Greece, Italy and France. However, there is an increasing interest in wastewater reclamation even in countries like Scotland, which have historically enjoyed an abundant water supply. This is due to water resources coming under increasing stress, especially in areas with intense agriculture. Furthermore, it is widely accepted that global climate changes will exacerbate problems with water shortages and lead to an increasingly unpredictable supply. The EU has recognised that water security particularly in the Mediterranean countries is an increasing concern. In 2015 the Commission presented a new circular economy package with a focus on promoting further uptake of water reuse across the EU. The Commission is developing a guidance document in the context of the Water Framework Directive (WFD) Common Implementation Strategy (CIS), together with Member States and stakeholders. Based on existing practice in the EU and third countries, it will contain recommendations on how to better integrate water reuse in water planning and management within the EU policy framework and taking into account underlying environmental and socio-economic benefits. Alongside this, the Commission will propose at the beginning of 2017 legislation on minimum requirements for water reuse in irrigation and aquifer recharge. The technical proposal is under development by the Joint Research Centre and will be consulted with the independent Scientific Committee on Health and Environmental Risks (SCHER). Furthermore, the Commission will look into further integration of water reuse in the development and review of Best Available Techniques Reference Documents (BREFs) for relevant industrial sectors under the scope of the Industrial Emissions Directive (2010/75/EU).

Wastewater reclamation is associated with a range of benefits including:

- Reducing the energy requirements and costs for wastewater treatment: treating wastewater to the required effluent discharge quality standards is an energy intensive process. For certain reuse applications, current treatment standards may be unnecessarily high and hence an inefficient use of energy.

- Nutrient recovery: wastewater effluents can contain substantial amounts of useful nutrients. Effluent can therefore be a valuable resource for agriculture as irrigation water, potentially lessening the dependence on manufactured fertilizers.

- Conserving freshwater resources: for example, abstractions from groundwater and surface water for irrigation can be reduced. This is particularly relevant during dry summers and in drier parts of the UK, where the water balance can be in deficit, and where unregulated over-abstraction can cause damage to ecosystems and biodiversity.

- Reducing the amount of drinking water being used for non-potable purposes.

- Minimising wastewater discharges to aquatic environments.

- Providing a stable water source.

Despite the many benefits, the option of reusing treated wastewater has only been developed to a limited extent in Scotland and across the EU. There are a number of barriers impacting the uptake of reuse projects and schemes, including risks to public health and the environment, public opposition, costs, technical and practical viability of implementation, and concerns over trade barriers e.g. restrictions on agricultural crops grown on land irrigated with effluent. Inadequate water pricing (costs of conventional water resources often do not reflect their actual costs due to subsidies) also means there is a limited economic incentive for users of reclaimed water.

1.1 Background and aims of study

This project builds on a previous CREW project on the establishment of a pilot Water Restoration Park at a wastewater treatment works (WWTW) to reclaim, recycle and market the wastewater (Morris et al., 2013) which found that:

- Wastewater reclamation and recycling presents a significant opportunity to enhance sustainable water management in Scotland.

- Interest in using reclaimed water needs to be fostered through:

    - Developing national guidelines to ensure public health;

    - Engaging with potential user communities (e.g. farmers, local authorities, commercial enterprises) and the consumers of their products (e.g. supermarkets, the general public) to raise awareness of the economic and environmental benefits of reclaimed water.

The aim of this research is to develop the foundations for reclaimed water use. The research has been undertaken in two stages:

I. Stage 1: support the development of national guidelines for reclaimed water in Scotland by reviewing existing guidelines for reclaimed water in other countries and using proven risk assessment tools to help develop reclaimed water standards that ensure public health.

II. Stage 2: establish a national-level stakeholder group and work with them to: evaluate the models developed in stage 1; identify benefits, savings, risks, and barriers to the use of reclaimed water.

The focus is on the intentional reuse of final effluents. For the purpose of this project, final effluent is defined as the treated final effluent from municipal/urban wastewater treatment plants. Other sources and types of wastewater, such storm water or mine waters, are recognised but are not specifically considered.

2 Overview of international guidelines on effluent reuse

2.1 Reclaimed water regulation in the UK

In the UK, there are currently no specific regulatory guidelines or water quality standards for reclaimed water. The reuse of treated urban wastewater is also not directly regulated, although there are several EU water-related directives establishing quality standards and legal restrictions for certain applications and/or defined environmental receptors. Relevant EU Directives include (BIO by Deloitte, 2015):

I. Drinking Water Directive (98/83/EC), covers indirect reuse of wastewater effluent for potable water production;

II. Bathing Water Directive (2006/7/EC), concerns the management of bathing water quality and is relevant to effluent reuse for recreational purposes;

III. Groundwater Directive (2006/118/EC), sets out standards relevant to the potential reuse of effluent for aquifer recharge and augmentation;

IV. Environmental Quality Standards Directive (Directive 2008/105/EC), sets environmental quality standards for priority substances considered a threat to the aquatic environment and is relevant to the reuse of effluent for environmental purposes (e.g. wetlands and surface water augmentation);

V. Nitrate Directive (91/676/EEC), aims to protect groundwater and surface water quality by reducing nitrate contamination from agricultural sources and is particularly relevant to effluent reuse for agricultural purposes.

VI. Urban Waste Water Treatment Directive (98/15/EEC), concerns treatment and discharge of urban and certain industrial wastewater, e.g. Article 13 sets out requirements for biodegradable wastewater discharges from certain industrial sectors, where the wastewater is not treated by an urban waste water treatment plant (WWTP) before discharge to receiving waters.

VII. Sewage Sludge Directive (86/278/EEC), not directly related to reuse of wastewater effluent, but may still be relevant to effluent reuse in agriculture, where many of the human health and environmental risks associated with sewage sludge application to agricultural land are similar to those for effluent reuse.

2.2 Guidelines and standards on effluent reuse

While there are currently no guidelines in the UK, national regulations and guidelines for reclaimed water exist in many other countries, where reclamation is widespread. Examples include the USA (particularly California), Singapore, Australia, Israel, Japan, Saudi Arabia, China and the Mediterranean region. Within the EU, recycling of wastewater effluent is practiced in the southern European countries; for example, Spain, Greece, Italy and France. Most of the countries with guidelines and standards in place are climatically different to Scotland, often having arid climates and a clear incentive to preserve water due to regular water shortages.

This section reviews existing international guidelines and standards to better understand current practices and regulations and identify main end uses in other countries. We focus on three exemplar countries, the USA (California), Spain and Australia, all of which have well-developed guidelines in place (especially California, which seems to be widely used as a model). Spain currently reuses the most water within the EU. We were particularly interested in democratic contexts where useful parallels were more likely to be found owing to similarities in governance arrangements and long established histories of citizen engagement. Non-democratic countries (for example, Singapore and China) have also developed reuse projects but within markedly different political contexts.

'Standards on water reuse' refers to different types of documents that provide requirements, specifications, guidelines or characteristics that can be used consistently to ensure water reuse projects achieve an acceptable level of health and/or environmental protection (BIO by Deloitte, 2015). Hence, existing standards focus on ensuring that any reuse of effluent is safe to human health and the environment, while barriers such as public perception and the financial feasibility are usually not explicitly covered by the standards. The Australian guidelines do, however, include consultation and communication, stressing the importance of communicating and engaging with communities and stakeholders to gain support for water recycling schemes. It should be noted that some countries have additional policy instruments (legislative and/or non-legislative) in place to govern, regulate and/or promote the reuse of wastewater (e.g. national targets). Water reuse projects in some cases, are governed by legislation that is not specific to water reuse (BIO by Deloitte, 2015).

In the following, the standards from California, Spain and Australia are summarised and compared. Table 2.1 presents an overview of this information.

2.2.1 Legal status

An important distinction needs to be made regarding whether the standards have legally-binding or guidance status. In much of the EU, including Spain, standards on wastewater reuse are legally-binding, while they typically have guidance status in countries outside the EU.

2.2.2 Types of wastewater

The existing standards define the types of wastewater covered. Most cover treated wastewater from urban/municipal WWTPs, but some also include other types of wastewater. For example, in Australia, specific guidelines exist for management of storm water and grey water.

2.2.3 Reuse categories

The standards usually define and describe which potential uses are allowed and which are prohibited in relation to effluent reuse. Although the existing standards cover many of the same uses, they often define their own reuse categories and application areas, which make it difficult to compare the different standards directly. Typical applications include agricultural irrigation, urban and industrial, domestic, and recreational and environmental. Potable uses are covered in a limited number of standards (including Australia). The number of application areas and reuse categories varies significantly between standards (see Table 2.1). Some standards are very specific in their definition of end-use categories (e.g. California) and may distinguish between, for example, restricted and unrestricted public access, or whether edible crops may come into direct contact with recycled water or not and are eaten raw or processed.

2.2.4 Approach

The exemplar guidelines differ in the approach upon which the standards are based. Three different approaches are distinguished:

i) Based on numerical limit values. Water quality limit values are defined for the different end uses. Health and/or environmental risks are deemed acceptable if the effluent complies with these numerical limit values. Limit values are defined at different points (e.g. at the outlet of the WWTP or at the point of delivery) depending on the standards. This type of approach is followed in Spain and most of the Mediterranean countries.

ii) Based on wastewater treatment requirements and numerical limit values. This approach is adopted in California. For each potential and approved end use, a specific and certified wastewater treatment technique is required, and for certain end uses, legally binding water quality criteria apply. Treatment requirement categories and associated water quality limits are defined. Other treatment methods not specified in the standards may be accepted if they are demonstrated to provide a satisfactory degree of treatment and reliability. Proposed groundwater recharge projects follow a different approach, where recommendations are made on a case-by-case basis by the State Department of Health Services based on all relevant aspects of each project, including treatment, effluent quality and quantity, spreading area operations, soil and hydrogeology, residence time, and distance to withdrawal.

iii) Based on implementation of a risk management system for each reuse project. This project-specific approach is followed in Australia. The Australian framework identifies and manages risks proactively and is intended to serve as a common basis for the establishment of state-level guidelines. The approach requires proposed reuse projects to identify and assess the associated health and environmental risks, implement measures to prevent and control these risks, and to put monitoring procedures in place to ensure the risks are reduced to an acceptable level. This approach is more proactive and flexible than both the Spanish and Californian guidelines and is designed for the treatment applied to the given wastewater source to be cost-effective while delivering the appropriate quality for the intended end use (BIO by Deloitte, 2015).

2.2.5 Number and types of parameters

The standards often specify how many and which parameters should be monitored and what their numerical limit values are for specific end uses. The number and types of parameter included in a standard is important, as it will influence the monitoring costs associated with a reuse project, which may in turn impact the uptake of reclaimed water schemes. Previous studies have suggested that countries that include fewer water quality parameters with legally-binding limit values recycle larger volumes of treated wastewater than countries having a large number of parameters (BIO by Deloitte, 2015).

The number of parameters with numerical limit values varies greatly between the exemplars. The Californian guidelines define numerical limit values for one main parameter (total coliforms) while the Spanish guidelines specify limit values for up to 90 parameters (BIO Deloitte, 2015); four of these (nematodes, E. coli, suspended solids and turbidity) are considered to be main parameters for which limit values apply to all use categories.

Direct comparison of parameter limit values between existing standards is complex because of the differences in the way reuse categories are defined, and because the standards focus on different parameters. Appendix III presents a summary of the main parameters and limit values considered in different standards on reuse of treated wastewater for agricultural irrigation.

2.2.6 Monitoring requirements/recommendations

The standards outline how often and where samples of reclaimed water should be taken and analysed as well as who is responsible for the monitoring (e.g. WWTP operator, final user). Determining the responsibility for the monitoring may influence the implementation of standards. The existing standards typically require a review process and quality control of the monitoring procedures to be in place.

Both the Californian and the Spanish guidelines specify legally binding minimum frequencies for sampling and analysis of the parameters of concern. In Australia, monitoring is defined on the basis of risk assessment, but guidance on types of monitoring (baseline, validation, operational and verification), parameters and suggested frequencies is provided. California, Australia and Spain all specify procedures for quality control and assurance of the monitoring program.

2.2.7 Application controls and additional risk management measures

Many standards recommend or require control measures for the use of reclaimed water. Typical measures include clear signalling of when effluent is being used, restrictions on crop types irrigated, safe irrigation distances (buffer zones), restrictions on application times, public access restrictions, avoidance of physical connections between recycled and potable water systems, and use of alarm devices and protective equipment for operators. In some cases, control measures are incorporated into the defined reuse categories. For example, California covers golf course irrigation with unrestricted and restricted access and also considers irrigation of ornamental nursery stock, where irrigation is either unrestricted or has to be ceased 14 days prior to harvesting or retail sale.

While the Spanish guidelines do not stipulate application controls as such, both the Australian and the Californian standards specify a number of use area requirements and preventive measures. The Australian guidelines estimate reductions in exposure to microbial pathogens by applying these preventive measures, although it states that these are associated with a great deal of uncertainty.

Table 2.1: Overview of the existing guidelines on reuse of treated wastewater in California, Spain and Australia (BIO by Deloitte, 2015)

	California	Spain	Australia
Reference document	CPDH (2009;2011)	Royal Decree 1620/2007	EPHC et al. (2006; 2008a,b; 2009a,b)
Legal status	Standards are legally binding	Standards are legally binding	Not legally binding; can be used for setting permits
Types of wastewater	Treated wastewater from municipal WWTPs.	Treated wastewater (source not specified)	Treated urban wastewater, storm- and greywater
Reuse categories	45 end-use types grouped into 4 categories: **Irrigation:** • Food crops for human consumption • Parks, playgrounds, school yards, cemeteries • Residential and freeway landscaping • Golf courses • Ornamental nursery stock and sod farms • Pasture and fodder crops for livestock • Any nonedible vegetation • Fruit trees (orchards, vineyards etc.) • Non-food-bearing trees **Impoundments:** • Recreational impoundments • Landscape impoundments not utilising decorative fountains **Cooling:** • Industrial cooling or air conditioning **Other purposes:** • Flushing toilets and urinals • Flushing sanitary sewers. • Priming drain traps • Industrial process water and boiler feed • Decorative fountains • Street cleaning and dust control • Commercial laundries • Commercial car washes • Construction (soil compaction, concrete mixing, consolidating backfill around pipes) • Fire-fighting • Artificial snow for outdoor use • Groundwater recharge of domestic supply	24 end uses grouped in 5 categories **Agriculture (irrigation):** • Food crops for human consumption • Grassland for livestock • Aquaculture • Fruit trees for human consumption • Ornamental flowers, nurseries etc. • Industrial crops/fodder not for humans **Urban uses:** • Residential (irrigation of private gardens; supply to sanitary appliances) • Services (urban green space irrigation, street cleaning, fountains, firefighting, car washing) **Recreational area:** • Golf course irrigation • Ornamental ponds/lakes **Environmental:** • Aquifer recharge • Woodland, green areas, silviculture etc. • Wetlands, streams etc. **Industrial uses:** • Water for industrial processes/ cleaning • Cooling towers & evaporative condensers Water reuse is prohibited for: potable water (except in emergencies), swimming water, hospitals, public fountains, and all other uses presenting health risks. Other reuses not listed can be permitted, if justified and approved by the water basin authority.	7 overall use categories: **Agricultural uses:** • irrigation for agriculture and horticulture **Residential and commercial property uses:** • garden watering • car washing and utility washing (paths, fences) • in-building (toilet flushing) • water features and systems (ponds, fountains) **Municipal uses:** • Irrigation of public greenspace, gardens, sports turfs • road making • street cleaning and dust control **Fire control uses:** • Firefighting & fire control systems **Managed aquifer recharge & indirect potable:** • Storage in aquifers to be extracted for later irrigation or as part of recycling schemes • Augmentation of drinking water supplies **Environmental:** • Streams and creeks • River • Lakes and dams **Industrial and commercial uses:** • Cooling water • Process water • Wash-down water

Table 2.1: Overview of the existing guidelines on reuse of treated wastewater in California, Spain and Australia (BIO by Deloitte, 2015)

	California	Spain	Australia
Approach	Based on defined WWT requirements and limit values. Treatment requirement categories are defined, each with specific water quality limits: • *Disinfected tertiary recycled water:* filtered and disinfected wastewater. Quality criteria for the disinfection process must be met. • *Disinfected tertiary recycled water subject to conventional treatment:* A treatment chain utilizing sedimentation between the coagulation and filtration processes and produces effluent that meets the definition for disinfected tertiary recycled water . • *Disinfected secondary-2.2 recycled water & Disinfected secondary-23 recycled water:* oxidized and disinfected effluent. • *Undisinfected secondary recycled water:* oxidized wastewater. Criteria for wastewater oxidization, coagulation and filtration are outlined and must be met.	Based on numerical water quality limit values. 14 water quality levels are defined, each with specific numerical limit values for a range of parameters. The required water quality level is specified for each use category.	Based on project-specific risk management approach. The guidelines require: i) to identify and assess the main health and environmental risks, and ii) to implement preventive measures and monitoring to reduce those risks to an acceptably low level. The standards furthermore specify: i) indicative log reductions of various enteric pathogens for different treatment processes; ii) treatment (log-reduction) sufficient for the different intended end uses; iii) an indicative list of relevant parameters. For the management of risks to human health, the guidelines define health-based targets based on DALYs (Disability-Adjusted Life Year).
Number and types of parameters	Main parameter: total coliform. Total coliform limit values are defined for the different treatment categories. Limit values defined for: i) 7-day median, ii) maximum, iii) a value that only one sample in any 30 day period may exceed. Turbidity limit values are specified in the definition of 'filtered wastewater'. Other parameters may be contained in the permissions granted to individual schemes. Other parameters specifically mentioned are i) F-specific bacteriophage MS2 and polio virus (in relation to specific disinfection requirements); and ii) Cryptosporidium, enteric viruses and Giardia (in relation to monitoring requirements for some impoundment schemes).	A total of approx. 90 parameters • 4 main parameters with numerical limit values for each use category: nematodes, E. coli, suspended solids, turbidity. • 14 physicochemical parameters specific to agricultural uses. • 67 hazardous substances (same limit values for all uses) • Some end uses require monitoring of other parameters, e.g. salmonella, legionella, taenia saginata/solium, nitrogen. Other substances are included in the treated effluent disposal permit. Water basin authorities may set limits to other parameters in effluent, depending on intended use.	A number of indicative parameters are mentioned, but not associated with numerical limit values, e.g.: • Suspended solids, turbidity, BOD, microbial quality (incl. faecal pathogens and indicators), chemical quality (e.g. salinity, nutrients, heavy metals, pesticides and other organics), algal counts, organic matter, colour, pH and disinfectant residuals. The standards mainly focus on microbial hazards for management of human health risk and on chemical hazards for management of environmental risks. A list of microbial and chemical hazards of potential concern is provided.
Monitoring requirements/ recommendations	Minimum required frequency for sampling and analysis is defined (daily sampling for total coliform and continuous sampling of turbidity). Samples are taken from the disinfected effluent and analysed by an approved laboratory. Supplier of the recycled water conducts the sampling. Monitoring results, operational problems, corrective actions etc. are recorded and filed.	Minimum monitoring frequencies required for the various parameters and quality controls are outlined. Reference methods for analysis of measurements are recommended. Holder of permit for reclaimed water use is responsible for the monitoring from the time reclaimed water enters the reuse distribution network to point of delivery. The user of reclaimed water is responsible from point of delivery until its points of use.	Sites potentially affected by the use or discharge of recycled water may need to be monitored. Monitoring is defined based on the risk assessment. Guidance on types of monitoring (baseline, validation, operational and verification), parameters and suggested frequencies is provided. It is recommended to review documentation and reliability of data, user satisfaction and to implement corrective responses. Data should be reviewed over time and after specific events (e.g. heavy rainfall).

Table 2.1: Overview of the existing guidelines on reuse of treated wastewater in California, Spain and Australia (BIO by Deloitte, 2015)

	California	Spain	Australia
Application controls and/or additional risk management measures	• Setback distances to domestic water supply wells (distances depend on treatment degree and type of use) • Setback distances to residences and public places (e.g. parks) when using spray irrigation. • Confine spray/runoff to effluent use area. • Protect drinking water fountains against contact with recycled water. • Signalling of all areas where recycled water is used that are accessible to the public. • Physical connections between any recycled and potable water systems not allowed. • Only quick couplers, in public areas • Preventive maintenance programs • Alarm devices to warn in case of any failures.	Not specified.	• Signage • Control of application methods (e.g. spray vs drip), times (irrigate only at night-time, withholding periods before harvest etc.) and rates (e.g. moisture sensors) • Use of buffer zones • Control of public access • Restricting uses of recycled water • Control of plumbing and distribution system (e.g. use backflow prevention devices, pipes for recycled water must comply with COP and be separated from potable systems, labelling etc.) • Operator and end-user awareness/ training Provides estimates of reduction in exposure to microbial pathogens associated with some of the above measures.
Possible requirements related to the granting of permits for water reuse	The process is defined by California Water Code. Any person recycling or using recycled water must file a report with the appropriate regional water board (RWB). This report must be prepared by a qualified engineer and comply with Californian regulations. RWB may prescribe further requirements to protect public health, safety etc., but must consult with and consider recommendations of the Department of Health Services if doing so.	The decree outlines the procedure for obtaining water reuse concessions and permits and provides an application form for a water reuse concession/permit. Note that more comprehensive guidelines are developed by certain regions (e.g. Andalucía).	Defined at state level. Each state and territory usually has their own specific guidelines which should also be adhered to.

2.3 Summary

The literature review has shown that the existing guidelines and standards on effluent reuse differ substantially in terms of the end uses considered, the approach taken, and the number and types of monitoring parameters considered. The literature also suggests that the statutes by the State of California, the World Health Organisation (WHO) and the US Environmental Protection Agency (CPDH, 2009; US EPA, 2012; WHO, 2006) constitute the basis for many of the legal guidelines proposed in other countries (Becerra-Castro et al. 2015). Regulations are often specifically for water reuse in agriculture rather than industry.

Standards are generally developed with the aim of ensuring that any effluent reuse is safe to human health and the environment. Many regulations and guidelines state that they are based on a risk assessment framework, but it is often not clear on what basis water quality parameters have been chosen and how the associated limit values have been set.

The review has highlighted a number of aspects that need to be addressed in order to develop guidelines for effluent reuse in Scotland. Key aspects include:

I. To identify which end uses and potential markets of reclaimed water are of most relevance to Scotland and therefore should be covered in the guidelines,

II. To decide which and how many water quality parameters and indicators to monitor, their associated limit values and whether this can be tailored to specific end uses and WWTPs. Studies indicate that including too many parameters and/ or specifying too strict limit values can lead to low uptake of water reuse schemes.

III. To decide on an underlying approach, i.e. should Scotland adopt legally binding requirements concerning water quality and/or treatment, similar to Spain or California, or should they follow the more flexible risk management approach of Australia. Both approaches have advantages and disadvantages, and further work is needed to determine the most appropriate model for Scotland.

3 Risk assessment of reclaimed water end uses

The aim of the risk assessment is to estimate appropriate or 'safe' levels of potentially harmful agents in water for end-use scenarios. This will provide an estimate of how strict water quality parameters need to be in final effluent (and by analogy how much energy is required to get the water to that quality) for a given end use. The risk assessment has been conducted in three stages:

I. Identification of potential end uses for reclaimed water.

II. Review of the literature to identify potentially harmful agents in final effluent and the range of concentrations expected

III. Construction of conceptual models for each of the identified end uses, and expansion on each model to develop scenarios to estimate the 'flow' of potentially harmful agents from the final effluent to vulnerable receptors.

3.1 General principles and methodology

The Quantitative Risk Assessment (QRA) calculations are based on a standard source-pathway-receptor principle (Figure 3.1). For risks to be realised, there has to be a complete pathway-linkage between the source and the receptor. The source is considered to be the final effluent containing certain amounts of hazardous residuals depending on the degree of treatment the wastewater has undergone. The characterisation of the source aims to identify which hazards remain in the effluent and at what concentrations, and is presented in Chapter 4. The pathways and receptors depend on the specific end-use scenarios. For each scenario, a conceptual model is developed, which aims to identify the key pathways and receptors. The conceptual models for the different end-use scenarios and the results of the risk assessment are presented in Chapter 5.

The aim of the QRA approach used here is to estimate the realistic worse-case doses of specific agents/hazards that a given receptor is being exposed to and compare these to estimated or reported safe (no-effect) doses. In calculating these, conservative values of the inputs are assumed in line with the precautionary approach. Input values are therefore taken to be the upper or lower 5th percentile of available data (depending on the type of input), where possible. The QRA consists of three stages: exposure assessment (Section 3.1.1), toxicity assessment (Section 3.1.2) and risk characterisation (Section 3.1.3).

3.1.1 Exposure assessment

The exposure assessment estimates the Average Daily Dose (ADD; mg kg^{-1} d^{-1} for chemicals and CFU kg^{-1} d^{-1} for pathogens) of a specific agent (for example, bacteria, organic contaminant) to a specific receptor (here: human or animal). The calculation of ADD depends on the specific exposure route. Three exposure routes will be considered; ingestion, inhalation and dermal uptake. The calculation of the dose through each of these routes is described in the following.

Exposure through direct ingestion

The Average Daily Dose through ingestion (ADDig) is calculated as:

$$ADD_{\mathrm{ig}} = \frac{C_m R_{\mathrm{ig}}}{BW} F_{\mathrm{ig}} \tag{3.1}$$

where Cm is the concentration (mg kg-1 for chemicals or CFU kg-1 for pathogens) of the specific agent in the ingested medium (e.g. soil, water, and/or crops), Rig is the ingestion rate of the media (kg day-1), BW is the body weight (kg) of the receptor (e.g. juvenile, adult), and Fig is the fractional time of exposure. The main challenge for Eq. 3.1 is to estimate the concentration of the specific agent in the ingested carrier media, Cm, and the amount of contaminated media ingested by the receptor over time for each of the considered exposure pathways. The concentration in the exposure media will here be estimated through reported concentrations in the source (i.e. the effluent) and by use of simple environmental transport and fate models. The modelling of the latter is described separately in the sections for each scenario.

Exposure through inhalation

The inhalation exposure model can be simplified to the following

$$ADD_{\mathrm{ih}} = \frac{C_a R_{\mathrm{ih}}}{BW} F_{\mathrm{ih}} \tag{3.2}$$

where ADD_{ih} is the Average Daily Dose through inhalation, C_a is the hazard concentration in air (mg m^{-3} for chemicals or CFU m^{-3} for pathogens), R_{ih} is the inhalation rate (m^3 day^{-1}) of the exposure medium, and F_{ih} is the fractional time of exposure through inhalation. The main challenge for Eq. 3.2 is to estimate the hazard concentrations in the air being inhaled, which depends on the specific end-use scenario considered.

Exposure through dermal uptake

The chemical absorption through the skin largely occurs by diffusion from the contaminated media (water, soil, etc.) in contact with the skin into the body tissue. The driving force for this process is the concentration gradient from the media in contact with the skin to the body tissue. As the concentration of most contaminants in the body tissue is usually considered negligible, the rate of adsorption is roughly proportional to the concentration in the media in contact with the skin (EA, 2009). A number of factors affect skin absorption of contaminants, including media type (e.g. soil or water), physicochemical properties of the contaminant (e.g. lipophilicity) and skin-specific

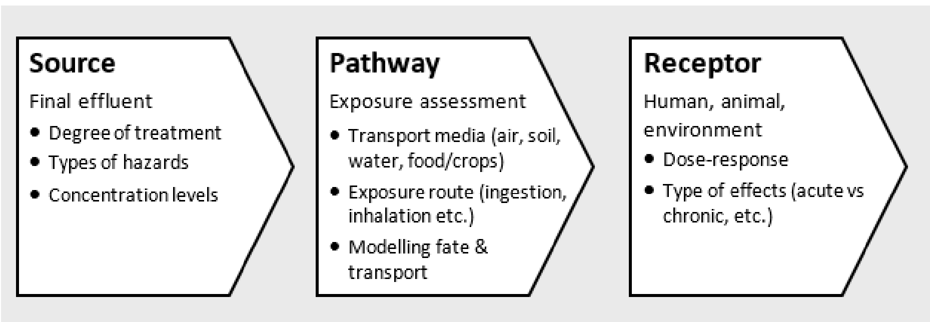

Figure 3.1: General source-pathway-receptor concept for the risk assessments.

factors such as thickness, ageing and hydration (US EPA, 1992). Some studies suggest that contaminants that are highly soluble in both fat and water are most likely to be adsorbed through skin, but the process is complex and affected by a number of competing factors (e.g. lipophilic contaminants are more likely to penetrate skin, but are also more likely to adsorb to soil, if this is the media considered).

The Average Daily Dose through dermal uptake (ADDd) is calculated differently depending on what contaminated media the skin is in contact with. This study only considers dermal uptake from water and soil, while uptake from vapours is not covered. We are also only considering dermal uptake of chemicals, while uptake of microbial hazards through the skin is assumed to be negligible. The human exposure to hazardous agents via dermal uptake from water is calculated as:

$$ADD_d^w = \frac{C_w \times K_p^W \times t_{event} \times EV \times A_{skin}}{BW} F_d \tag{3.3}$$

where C_w (mg/ml) is the concentration of the specific hazardous agent in water, K_p^w (cm hr^{-1}) is a contaminant-specific permeability coefficient through skin from water, t_{event} (hr/event) is the duration of the exposure event, EV is the number of daily contact events, A_{skin} is the exposed skin area (cm^2) and F_d is fractional time of dermal exposure.

Experimentally derived K_p values for different contaminants can be found in the literature (e.g., US EPA, 1992; Bogen, 2013). A number of regression models have also been developed to estimate K_p^w, mainly for organic contaminants, based on K_{ow} and molar weight (MW, g mol-1). We have here used the regression model by the US EPA (1992) to estimate K_p^w for organic contaminants for which experimentally derived K_p^w values do not exist. A default value of K_p^w = 0.001 cm/hr is recommended for inorganic contaminants, for which measured values are not available.

The human exposure to hazardous agents via dermal uptake from contaminated soil adhered to the skin is calculated as (EA, 2009; US EPA, 1992):

$$ADD_d^s = \frac{C_s \times AF \times ABS_d \times EV \times A_{skin}}{BW} F_d \tag{3.4}$$

where Cs is the hazard concentration in soil adhered to the skin (mg/kg soil), AF is a soil-to-skin adherence factor (mg soil/cm2 skin), ABSd is the dermal adsorption fraction, and EV is the number of daily soil contact events. To calculate the dermal uptake from soil, the concentration in soil needs to be estimated. This is described in later sections for the relevant scenarios.

3.1.2 Toxicity assessment

The aim of the toxicity assessment is to determine a hazard-specific acceptable dose, i.e. the maximum daily uptake level of a hazard that is not likely to result in any adverse effects and hence is considered 'safe'. The toxicity assessment depends on the specific contaminant/hazard and the exposure route (whether the contaminant enters the receptor through e.g. inhalation, ingestion or dermal contact), but also on the type of response (cancer vs. non-cancer) and whether chronic or acute effects are considered.

For non-carcinogenic agents, it is standard practice to assume that a threshold of effect exists, and that the safe dose can be expressed as a so-called reference dose (RfD). RfDs are usually estimated based on dose-response relationships, which describe what the adverse effects are at different exposure levels, when no effects are observed and when responses start to appear. Many methodological approaches exist for deriving RfDs from available dose-response data (Appendix I). It should be noted that RfDs are often determined conservatively from dose-response data by applying a number of uncertainty factors to account for the uncertainties associated with extrapolating from the experimental population to the study population at risk (e.g., extrapolating from an experimental rat population to a human), and the variability within receptor populations (e.g., differences in the amount of contaminated media consumed, differences in the inherent susceptibility of different members of the population) (Barnes and Dourson, 1988).

For carcinogenic agents, it is often assumed that a threshold of effect does not exist. Instead, a so-called cancer slope factor is derived from the dose-response data (Subramariam et al., 2006). RfDs are also not usually defined for pathogens and microbial hazards, as it is not possible to perform a direct study to assess dose corresponding to an acceptably low risk. Instead, the dose-response assessments of pathogens are typically based on mathematical dose-response models for specific pathogens and pathways. Two models exist, which are often applied to describe the dose response relationship of pathogen in QRA: the Beta-Poisson model and the exponential model (Haas et al., 1999). The Beta-Poisson model is used to describe the dose-response relationship between exposure to pathogens such as *Campylobacter, L. monocytogenes* and *E. coli O157* and subsequent human infection (Gale, 2005). This model assumes that pathogens act independently and that the minimum infectious dose is one pathogen. This approach is taken to reflect the most precautionary assumption. The model is defined as:

$$p = 1 - \left[1 + \frac{N}{N_{50}}(2^{1/\alpha} - 1)\right]^{-\alpha} \tag{3.5}$$

where p is the probability of infection following exposure to N pathogens and N_{50} is the ID_{50} (the dose expected to cause infection in 50% of the total population when given to each and every member of that population).

The exponential model is often used to describe the dose-response relationship for pathogens such as *Cryptosporidium parvum* and *Salmonella Enteritidis*. It is defined as:

$$p = 1 - \exp(-rN) \tag{3.6}$$

where r is an approximation of the probability of a single particle to survive and cause infection after ingestion (Haas et al. 1996). Values of the N*50*, a and r parameters values for different pathogens reported in the literature are presented in Appendix II.

Appendix II presents acceptable dose values for hazards considered as part of this project. It should be noted that the dose-response model parameters in Appendix II are typically defined for a specific exposure route (e.g. ingestion). It should also be noted that the two dose-response models predict the probability of infection. However, infection may not always lead to illness/disease, depending instead on many factors such as an individual's immunity status, age, medical conditions, and nutrient intake (Lim et al., 2015). The dose-response parameters are associated with large uncertainties, as is apparent from the wide variation in reported parameter values. For example, the N50 (dose expected to cause infection in 50% of the total population) for *E.coli O157:H7* varies many orders of magnitude. Similar dose-response models for microbial pathogens are recommended in the Australian guidelines for effluent reuse (EPHC et al., 2006). Finally, it should be noted that, within this study, all hazardous agents are assessed on an individual basis. There is an increasing focus on the issue of mixtures and their actions, but the evidence is still in its infancy and comprehensive data are not yet available for all of the combinations of chemicals possibly present in effluent.

3.1.3 Risk characterisation

The final stage of the QRA is risk characterisation, where the ratio of the exposure (i.e. ADD, mg kg^{-1} d^{-1}) to the appropriate reference dose (RfD, mg kg^{-1} d^{-1}) is determined:

$$HQ = \frac{ADD}{\mathrm{RfD}} \quad (3.7)$$

If the ADD exceeds the RfD, the Hazard Quotient (HQ) will be greater than 1.0 and we might expect to see deleterious effects. Risk in this study was expressed either as 'negligible' ($HQ \leq 1.0$) or potentially requiring further investigation ($HQ > 1.0$).

To assess the risk from exposure to pathogens, the calculated probability of infection (Eq. 3.5 or 3.6) needs to be compared against some acceptable level of probability. We will here assume an acceptable probability level of less than one infection per year per 10,000 people, in line with the goal of the US EPA Surface Water Treatment Rule developed for *Giardia lamblia* (US EPA, 2002). To calculate the annual risk of infection, the following equation is used:

$$p_{annual} = 1 - \left(1 - p_{daily}\right)^{365} \quad (3.8)$$

where the daily probability of infection is calculated with Eq. 3.5 or 3.6, depending on the type of dose-response model being used. Based on the acceptable probability level of 0.0001 infections per person per year and the parameters in Table A2.2, Appendix II, it is possible to estimate the corresponding "bacterial reference dose".

The Australian guidelines suggest a similar approach to risk characterisation for management of human health risk from pathogens. However, these guidelines recommend the use of disability adjusted life years (DALYs) to convert the likelihood of infections/illnesses and set a tolerable risk at 10^{-6} DALYs per person per year. A similar approach is used in the WHO guidelines.

3.1.4 Backward risk calculations

The approach outlined above (Fig. 3.1) describes a forward QRA, where the risks are calculated in the 'flow' from the source through the considered pathways to the receptor. For the different end-use scenarios considered in the later sections, the risk assessments will be performed backwards, as illustrated in Figure 3.2, i.e. the assessment will start from a safe/acceptable risk level at the receptor and the risk calculations performed in reverse to determine what the corresponding concentration levels of the given hazards need to be in the final effluent under the considered scenario. Such assessment will inform the level of water treatment required to achieve a safe risk level.

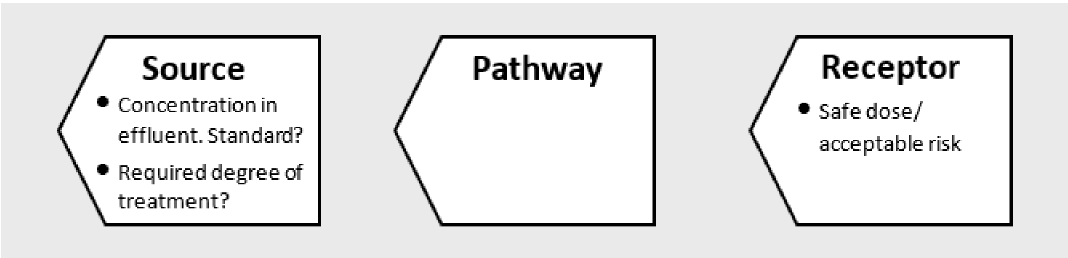

Figure 3.2: Backward risk assessment.

4 Hazard identification: Review of potentially harmful agents in effluent

As part of the review of the existing standards and regulations (Chapter 2), key contaminants of concern that are presently subject to regulation in other countries were identified. However, since we are assessing the safety of final effluent for various uses, and not whether it complies with current standards, it is important not to ignore emerging hazards and other potentially harmful agents that are not subject to regulation. Current water quality standards were therefore supplemented by a literature review to establish what potentially harmful agents could be present in the final effluent.

The review also provides data on measured concentrations of these agents, as well as safety standards and toxicological information of identified agents where these are available.

4.1 Potentially harmful hazards in wastewater effluent

The reviewed hazards have been grouped into the following categories (Table 4.1): microbial hazards; potentially toxic element (PTEs); organic contaminants (including PAHs, PCBs, PCDD/Fs and pesticides); and environmental and physical hazards. Within each category a number of specific hazardous agents have been listed. The list is not exhaustive. Wastewater effluent can contain a very large mix of hazards. The types of hazard depend, among other things, on the type and size of the wastewater catchment. A good understanding and characterisation of the source water is therefore important, as this will influence the types and amounts of hazards found.

In the following, each hazard category is described, focussing on selected hazards considered in the risk assessment scenarios (see Chapter 5). Because the risk assessments here mainly focus on human (and animal) health, the environmental and physical hazards will not be described further in this review, despite these potentially having harmful and unwanted effects on the environment if present in high concentrations.

Hazard group	Specific hazards
Microbial hazards (pathogens)	Bacteria (e.g., E. coli O157:H7, Campylobacter, Salmonella, Listeria, Legionella)
	Viruses (e.g. rotavirus, adenovirus)
	Protozoa (e.g. Cryptosporidium parvum, giardia)
	Parasites, helminths (e.g. nematodes, taenia)
Potentially Toxic Elements (PTEs)	Cadmium, chromium, copper, lead, nickel, zinc, mercury etc.
(Persistent) organic contaminants	Polycyclic Aromatic Hydrocarbons (PAHs) (e.g. naphthalene, chrysene, benzo-a-pyrene)
	Polychlorinated Biphenyls (PCBs) (e.g. PCB28, PCB95, PCB101, PCB153, PCB180)
	Pesticides and herbicides (e.g. Clopyralid, Fenoxycarb, Imazalil, Pentachlorophenol)
	Polychlorinated dibenzo-dioxins and –furans (e.g., 2,3,7,8-TeCDD; 1,2,3,4,7,8-HxCDF)
	Pharmaceuticals and Personal Care Products (PPCPs) (e.g., antibiotics, Clofibric acid, Carbamazepine)
	Sex/steroidal hormones and other endocrine disruptors
	Surfactants (e.g., Alkane ethoxy sulfonates)
	Water disinfection by-products
	Algal toxins
	Caffeine
Environmental and physical hazards	Nutrients (e.g. nitrogen, phosphorous, potassium, sodium, DOC, boron)
	Dissolved inorganics and salts (e.g. Na, Ca, Mg, Cl, B, total dissolved solids, electrical conductivity)
	pH
	Suspended solids
	BOD, COD
	Total organic carbon (TOC)
	Turbidity
	Temperature

Table 4.1: Overview of hazards present in sewage and wastewater effluent, which could pose a risk to human health and/or the environment (Kopec et al., 1993; Radcliffe, 2004; Toze, 2006; Roberts and Thomas, 2006; Zhang et al., 2008; EPHC et al., 2006).

4.1.1 Microbial hazards (pathogens)

The risks associated with microbiological hazards, such as pathogens and antibiotic resistance genes, are a particular concern when considering reuse of reclaimed effluent. A wide range of microbial hazards can be found in sewage and sewage effluent. The types of pathogens present will depend on the state of public health, and on the presence of nearby risk factors, for example, hospitals, tanneries, meat-processing factories, and slaughterhouses (Dumontent et al., 2001). Foodborne pathogens are generally among the most important cause of sewage contamination in developed countries.

The guidelines in place in other countries mostly define standards for some microbiological hazards, focussing on the presence of potential human pathogens and parasites. Because it is impractical to test wastewater effluent for all pathogens, the presence of pathogens is typically assessed by use of faecal indicator organisms (FIOs) (e.g. total coliform, faecal coliform and *Escherichia coli*) and nematode eggs (Norton-Brandão et al, 2013; Becerra-Castro et al., 2015). Several studies exist in which wastewater effluents have been analysed for various FIOs. For example, Kay et al. (2008) analysed sewage effluents from 162 discharge sites across 12 study areas in the UK for different FIOs (total coliform, faecal coliform and Enterococci) for different levels of treatment (untreated, primary, secondary and tertiary treatment) and individual types of sewage-related effluents under different flow conditions (base vs high flow). They found that FIO concentrations decreased with increasing level of treatment with e.g. total coliform concentrations ranging from around 3 x 10^7 CFU/100 ml after primary treatment only, to 1 x 10^6 CFU/100 ml after secondary treatment and 5.5 x 10^3 CFU/100 ml after tertiary treatment. It should be noted that some studies have found poor correlations between FIOs and presence of specific pathogens (Aquarec, 2006), with some pathogens being less susceptible to the different wastewater treatment processes, raising questions about the adequacy of standards based on FIOs to protect human health (Harwood et al., 2005). There is therefore a need to define more suitable indicators to better characterise the entire biological quality of different types of wastewater (Aquarec, 2006).

For this study, the following microbial hazards (Table 4.2) have been selected for the risk assessment scenarios in Chapter 5: *Escherichia coli O157*, *Campylobacter*, *Salmonella*, *Cryptosporidium* and *Listeria*. These are all common foodborne pathogens and considered an important cause of sewage contamination in developed countries (Dumontent et al., 2001). Major sources of these pathogens include faeces of healthy cattle, sheep and pigs; animal waste from abattoirs and animal treatment plants; and raw poultry and pork (Hutchison et al. 2004, Jones, 2001, Dumontent et al. 2001; Robertson et al., 2000).

The existing guidelines in other countries on effluent reuse do not define threshold values for these specific pathogens, but instead specify threshold values for certain indicator organisms. For example, many guidelines specify total *E. coli* (which include the non-pathogenic strains) as an indicator of microbial quality, and specify threshold values generally ranging between 100 – 1000 CFU/100 ml for unrestricted irrigation, while the threshold levels for some restricted irrigation uses can be up to 10^5 CFU/100 ml (Becerra-Castro et al., 2015).

Pathogen	Concentration in effluent	Comment	References
***E. coli* (total)**	1.3 x 10^6 CFU/100 ml	Effluent from 32 septic tanks, NE Scotland	Richards et al. (2016)
	1.1 x 10^7 (0.04 x 10^7 – 6.9 x 10^7) CFU/100 ml	Effluent from small WWTP, Scotland (P)	Personal comm.
	1.7×10^5 (1.3×10^4 – 7.5×10^5) CFU/100 ml	Effluent from WWTP in Rome (P, S)	Bonadonna et al. (2001)
***E. coli* O157:H7**	0.1 – 10 PFU/100 ml	Sewage from different countries	Muniesa and Jofre (2000)
Campylobacter	262-79000 cells/100 ml	Effluent from WWTW in Lancaster (P)	Jones et al. (1990a,b; 2001)
Salmonella	1.5x10^4 (0.03x10^3 –50x10^3) MPN/100 ml	Effluent from WWTP in Rome (P, S)	Bonadonna et al. (2001)
	1x10^5 CFU/100 ml	Effluent from municipal WWTPs, South Africa (P, S, T)	Odjadjare et al. (2010)
	45 MPN/100 ml	Effluent from WWTP in Brazil (P,S,T)	Howard et al. (2004)
Listeria	300-2100 MPN/100 ml	Effluent from a WWTP in France (P,S)	Paillard et al. (2005)
	500-240000 CFU/100 ml (max: 1.2x10^7 CFU/100 ml)	Effluent from municipal WWTP, South Africa (P, S, T)	Odjadjare et al. (2010)
	From 7×10^4 to >1.8×10^6 MPN/100 ml	Industrial effluent from UK WWTP (P)	Watkins and Sleath (1981)
Cryptosporidium	0.5-6 oocysts/100 ml	Sewage from six sewage treatment works within Strathclyde Region in Scotland	Robertson et al. (2000)
	3.7 oocyst/100 ml	Effluent from WWTP in Rome	Bonadonna et al. (2002)
	0.002 – 0.1 oocyst/100 ml	Effluent from 6 WWTPs in the US (P,S,T)	Harwood et al. (2005)

Table 4.2: Microbial pathogens selected for the Quantitative Risk Assessment and examples of their reported concentration levels in treated effluent. (P)= primary treatment; (S)= secondary treatment; (T)=tertiary treatment. CFU: Colony Forming Units; PFU: Plague Forming Units; MPN: Most Probable Number

4.1.2 Potentially Toxic Elements (PTEs)

Wastewater effluent, especially from urban and industrial WWTPs, will contain heavy metals and other potentially toxic elements (PTEs) such as cadmium, chromium, lead, mercury, arsenic, and copper. Although heavy metals in raw sewage are easily removed during standard treatment processes with the majority of the PTEs ending up in the sewage sludge and only low concentrations present in the treated effluent, the impact of heavy metals on the environment and human health is still a concern to regulators and the public (Toze, 2006; Smith et al., 1996). This is particularly the case when considering irrigation with effluent, as heavy metals present in effluents used for irrigation tend to accumulate in the soils and may become bioavailable for crops (Toze, 2006).

For the QRAs carried out here, the following six PTEs are considered: copper (Cu), chromium (Cr), cadmium (Cd), lead (Pb), nickel (Ni) and zinc (Zn). In Table 4.3 an overview of the selected metals and reported concentration levels in effluent is presented.

PTE	Concentration in effluent (μg/l)	Comment	References
Cu	109 (5-637)	Effluent from 32 septic tanks, NE Scotland	Richards et al. (2016)
	11 (0-70)	Effluent from small WWTP, Scotland (P)	Personal comm.
	25-100	Municipal effluent WWTP, Australia (P, S)	Smith et al (1996)
	58 ± 37 (P) 33 ± 5.8 (S)	Effluent from WWTP in Greece (P,S)	Karvelas et al. (2003)
	10 (2-20)	Effluent from WWTP in Brazil (P,S,T)	da Silva Oliveira et al. (2007)
Cr	1.05 (0.25-3.49)	Effluent from 32 septic tanks, NE Scotland	Richards et al. (2016)
	5-20	Municipal effluent WWTP, Australia (P, S)	Smith et al (1996)
	25 ± 12 (P) 20 ± 3.5 (S)	Effluent from WWTP in Greece (P,S)	Karvelas et al. (2003)
	5.7 (1.7-13.5)	Effluent from WWTP in Brazil (P,S,T)	da Silva Oliveira et al. (2007)
Cd	< 6	Municipal effluent WWTP, Australia (P, S)	Smith et al (1996)
	2.3 ± 0.9 (P) 1.5 ± 0.7 (S)	Effluent from WWTP in Greece (P,S)	Karvelas et al. (2003)
	0.06 (0.04-0.1)	Effluent from WWTP in Brazil (P,S,T)	da Silva Oliveira et al. (2007)
Pb	1.68 (0.5-6.67)	Effluent from 32 septic tanks, NE Scotland	Richards et al. (2016)
	1.5 (0-25)	Effluent from small WWTP, Scotland (P)	Personal comm.
	2-8	Municipal effluent WWTP, Australia (P, S)	Smith et al (1996)
	31 ± 12 (P) 27 ± 3.6 (S)	Effluent from WWTP in Greece (P,S)	Karvelas et al. (2003)
	22.5 (4.2-76.4)	Effluent from WWTP in Brazil (P,S,T)	da Silva Oliveira et al. (2007)
Ni	66 (0-380)	Effluent from small WWTP, Scotland (P)	Personal comm.
	3.0-6.0	Municipal effluent WWTP, Australia (P, S)	Smith et al (1996)
	600 ± 270 (P) 430 ± 97 (S)	Effluent from WWTP in Greece (P,S)	Karvelas et al. (2003)
Zn	150 (18-287)	Effluent from 32 septic tanks, NE Scotland	Richards et al. (2016)
	47 (0.3-300)	Effluent from small WWTP, Scotland (P)	Personal comm.
	40-82	Municipal effluent WWTP, Australia (P, S)	Smith et al (1996)
	380 ± 50 (P) 270 ± 53 (S)	Effluent from WWTP in Greece (P,S)	Karvelas et al. (2003)
	43.6 (22.8-76.3)	Effluent from WWTP in Brazil (P,S,T)	da Silva Oliveira et al. (2007)

Table 4.3: PTEs selected for the Quantitative Risk Assessment and examples of their reported concentration levels in treated effluent. (P)= primary treatment; (S)= secondary treatment; (T): tertiary treatment

4.1.3 Persistent organic pollutants

Wastewater and wastewater effluent can contain a large number of organic pollutants such as pharmaceuticals, pesticides, hydrocarbons, and endocrine disruptors. For example, Paxeus (1996) analysed effluent from the three largest WWTPs in Sweden for presence of organic pollutants and identified 137 different compounds, with concentrations of individual compounds in the range of 0.5–50 μg/l. Roberts and Thomas (2006) analysed the occurrence of 13 priority pharmaceutical compounds in the final, pre-UV disinfected and raw effluent from a WWTP in Newcastle (UK). Most of the compounds were measured in the raw effluent in concentrations of up to 70,000 ng/l. Zhang et al. (2008) analysed sewage effluent from a WWTP in West Sussex (UK) for a range of emerging organic contaminants, including endocrine disrupting chemicals and pharmaceuticals, with concentrations of up to around 600 ng/l. Pal et al. (2010) present a review of the occurrence and concentration levels of various pharmaceuticals (including antibiotics, beta-blockers, anti-inflammatory and antiepileptic compounds) and hormones in wastewater effluent. They report concentration levels from below 1 ng/l to concentrations in the µg/l level.

Due to the vast range and mixture of organic contaminants present in wastewater effluent, it is not possible to carry out QRAs for all organic pollutants that have been monitored in effluent. In a project for WRAP (2013) on assessing the risks from applying green waste compost to land, a comprehensive review of potentially harmful organic pollutants present in compost was carried out, and following engagement with a large stakeholder group, a number of organic contaminants were identified and selected for a detailed risk assessment based on the stakeholders' concerns and priorities. The specific contaminants are listed in Table 4.4. Most of these figures on the EU list of priority substances in the field of water policy (Annex X of the Water Framework Directive). For the risk assessments here, the same organic contaminants have been selected.

5 Risk assessment of reclaimed water end-use scenarios

Quantitative Risk Assessment (QRA) calculations for a number of end-use scenarios of reclaimed water (i.e. final effluent) have been carried out. An overview of the selected end-use scenarios is presented in Table 5.1. These scenarios represent five common areas of effluent reuse and were chosen based on the review of the existing guidelines (Section 2) and other literature as well as through consultation with the project steering and stakeholder groups (see Section 6).

The QRA calculations are based on the methodology presented in Section 3. The source term is the same for the different risk assessment scenarios and was covered in the previous section. However, it should be noted that for some scenarios, some types of hazards might be more relevant to consider than others. The pathways and receptors depend on the specific end-use scenarios. For each scenario, a source-pathway-receptor conceptual model is therefore developed. Table 5.2 lists a number of assumptions regarding the source and the receptors that are in common for most of the considered scenarios.

Organic pollutant group	Specific hazard
Polycyclic Aromatic Hydrocarbons (PAHs)	Naphthalene (NAP)
	benzo-a-anthracene (B[a]A)
	chrysene (CHR)
	benzo-b-fluoranthene (B[b]f)
	benzo-k-fluoranthene (B[k]f)
	Benzo-a-pyrene (B[a]P)
	indeno(1,2,3-cd)pyrene (IPY)
Polychlorinated Biphenyls (PCBs)	PCB28, PCB52, PCB95, PCB101, PCB118, PCB132, PCB138, PCB149, PCB153, PCB174, PCB180
Polychlorinated dibenzo-dioxins and –furans (PCDD/Fs)	2,3,7,8-TeCDD; 1,2,3,7,8-PeCDD; 1,2,3,4,6,7,8-HpCDD
	2,3,4,7,8-PeCDF; 1,2,3,4,7,8-HxCDF; 1,2,3,6,7,8-HxCDF; 2,3,4,6,7,8-HxCDF
Pesticides and herbicides	Clopyralid
	Fenoxycarb
	Imazalil
	Pentachlorophenol

Table 4.4: Selected organic pollutants for the risk assessment scenarios

	Scenario	Specific hazard
1	Urban irrigation (non-edible)	Final effluent used for irrigating non-edible (or at least not intended for human consumption) plants, e.g. golf courses, parks, road verges, nursery stock.
2	Urban irrigation (edible)	Use of final effluent for the irrigation of urban agricultural areas such as allotments, community gardens, back gardens.
3	Agricultural irrigation	Final effluent used for irrigation of agricultural crops. Agricultural irrigation could also include the use of final effluent for watering livestock where potable-quality water is not necessary
4	Non-potable domestic	Final effluent used for non-potable uses within the domestic environment. The most likely use under this scenario is for operations such as toilet flushing.
5	Industrial (uncontained)	Final effluent used for industrial processes where personnel are likely to come into contact with the water and are likely to have to wear personal protective equipment. Vehicle washing is a good example of such a process.

Table 5.1: Overview of the different reclaimed water end-use scenarios

<table>
<tr><th>Stage: Variable</th><th>Assumption</th></tr>
<tr><td>Source:
Final effluent</td><td>Concentrations of potentially hazardous agents in final effluent. Three groups of hazards are considered, as discussed in section 4: i) microbial hazards, ii) Potentially Toxic Elements, and iii) organic contaminants. Hazard specific inputs required for the risk assessments are:

1. Concentration in effluent: Only needed for forward risk calculations. Used in Eq. 5.3, 5.4, 5.6, 5.7 and Eq. 3.1 (direct water ingestion).
2. Dermal uptake: Permeability coefficient through skin from water (K_p^w; Eq. 3.3) and dermal adsorption fraction (ABS_d; Eq. 3.4)
3. Henry's constant: Only for used for organics. Used in Eq. 5.3 and 5.4.
4. First-order decay constant (λ). Is used in Eq. 5.3 and 5.4. However, in all scenarios, decay is assumed to be zero as a precautionary assumption.
5. Adsorption to soil: Soil-water equilibrium distribution coefficient (K_d) is needed for inorganic and organic contaminants (Eq. 5.3). For pathogens, adsorption and desorption constants k_{att} and k_{det} are needed (Eq. 5.4)
6. Bio-concentration factor (BCF). Needed for plant uptake (Eq. 5.5).

Input values used for the chosen hazards are presented in Appendix II.</td></tr>
<tr><td>Receptor: Statistics</td><td>1. Human health: Three population sub-groups were identified (Hough et al. 2012; 2004):
i. 'Average Person' – age 39.5, body weight 74 kg
ii. '95 %ile vulnerable person' – age 39.5, body weight 36.7 kg
iii. 'Highly exposed infant' – age 5, body weight 13 kg

2. Animal health (agricultural scenario only): Three population sub-groups of livestock considered (Hough et al. 2012; 2004; WRAP, 2013):
i. Sheep (40 kg)
ii. Cattle (500 kg)
iii. Poultry (3 kg)</td></tr>
<tr><td>Receptor: Exposure</td><td>1. Soil ingestion rates by human receptors (Rig,soil; Eq. 3.4) are assumed to be (Sheppard et al., 1995; EA, 2009):
i. 'Average Person' = 20 mg d^{-1}
ii. '95 %ile vulnerable person' = 50 mg d^{-1}
iii. 'Highly exposed infant' = 500 mg d^{-1}

2. Dietary intake rates (kg d^{-1}) by human receptor for different vegetable types (adapted from Hough et al., 2004):
<table>
<tr><th>Produce</th><th>Average Person</th><th>95%ile Person</th><th>HEI</th></tr>
<tr><td>Root crops</td><td>0.1</td><td>0.20</td><td>0.15</td></tr>
<tr><td>Leafy green</td><td>0.1</td><td>0.25</td><td>0.01</td></tr>
<tr><td>Fruit</td><td>0.05</td><td>0.10</td><td>0.07</td></tr>
<tr><td>Grains</td><td>0.2</td><td>0.28</td><td>0.25</td></tr>
</table>
3. Inhalation rate, $R_{inhalation}$, is assumed to be 15 l min^{-1} for all human receptor groups (Lim et al., 2015; US EPA, 2011).

4. Livestock intake rates (kg dry matter d^{-1}) during grazing (Abrahams, 2013; WRAP, 2013; Hough et al. 2012; Phillip and Leaver, 1987; Singh et al. 2013):
<table>
<tr><th></th><th>Sheep</th><th>Cattle</th><th>Poultry</th></tr>
<tr><td>Grass (kg DM d-1)</td><td>1.25</td><td>12.5</td><td>0.04</td></tr>
<tr><td>Soil intake while grazing (% DM)</td><td>16</td><td>9</td><td>10</td></tr>
</table></td></tr>
<tr><td>Receptor: Risk characterisation</td><td>List of safe doses for the different hazards are presented in Appendix II. Safe doses for PTEs and organic contaminants are taken from literature. For pathogens, an acceptable probability level of less than 1 infection per year per 10,000 people is assumed (US EPA, 2002).</td></tr>
</table>

Table 5.2: Assumptions common to all QRA end-use scenarios.

5.1 Scenario 1: Urban Irrigation, non-edible

This scenario aims to assess the risks and human exposure from use of final effluent in urban irrigation of public green spaces and sports turf (e.g., golf courses, football pitches). This is a practice currently adopted in several other countries, for example, in the United States (Carrow, 1997).

Although the demand for irrigation in the UK is lower than elsewhere, irrigation is still important. For example, based on a survey of golf courses in England and Wales and analyses of Environment Agency abstraction data, Knox et al. (2008) estimated that the depth of irrigation applied in 2003 (a relatively dry year) was on average around 250 mm (with a standard deviation of 150 mm), corresponding to an irrigation demand of about 4500 m^3 per year for a typical 18-hole golf course. They found that peak demand in dry summers is 600 m^3 per week for a typical golf course. According to Knox et al. (2008), the total licensed volume for golf course irrigation was around 10,112 million litres in 2003, representing 3% of the total volume licensed for spray irrigation in England and Wales, with agricultural spray irrigation accounting for the vast majority of the remainder. It should be noted that the Environment Agency abstraction records for golf course spray irrigation do not include water taken via the public mains supply for irrigation, so the actual water demand for golf course irrigation is likely to be higher. Although golf irrigation constitutes a relatively minor abstraction, it predominantly occurs during the driest periods when water resources are scarcest.

The exposure scenario here relates to occupational exposure (gardeners, grounds people, greenkeepers) as well as potential exposure of children and adults through inadvertent ingestion and contact with effluent and soil (Figure 5.1). Specific assumptions used are presented in Table 5.3.

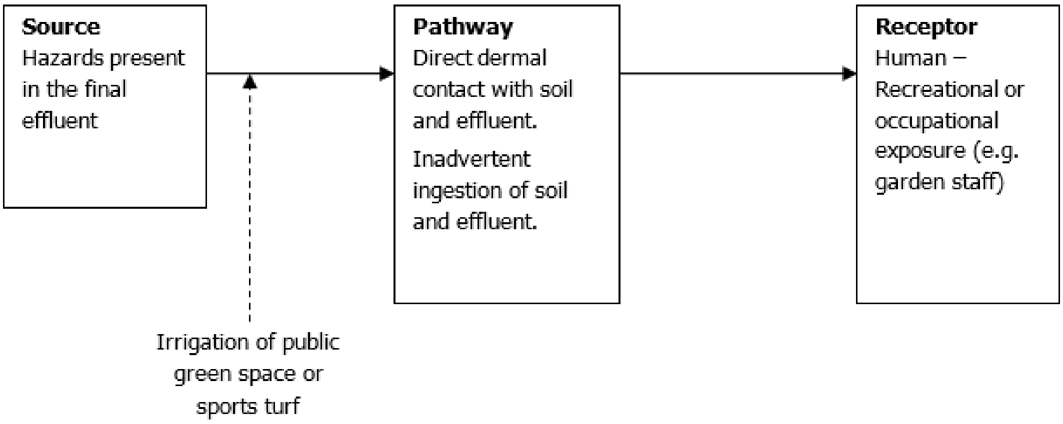

Figure 5.1: Source-pathway-receptor schematic of Scenario 1 – Urban irrigation (non-edible).

Stage: Variable	**Assumption**
Pathway: Irrigation of public green space and sports turfs	1. Irrigation rate is 4 mm/day and takes place 4 months every year (Knox et al., 2008). The irrigation rate should not exceed the soil's infiltration rate. If irrigation is assumed to take place for 4 hours every day, then the irrigation rate is 1 mm/hr when the system is in operation, which should be well below the infiltration rate of even quite low-permeable soils. 2. Assume irrigation rate is uniform across both grass and (planted) soil areas. 3. Assume irrigation occurs early morning/late evening and direct inhalation of water vapour unlikely. 4. Assume that hazards/contaminants applied to land via effluent irrigation are mixed uniformly and instantaneously in the top 10 cm of the soil according to mass balance and fugacity principles. 5. Assume soil bulk density is 1.5 kg/l, organic carbon content is 1%, soil porosity is 0.4, and soil water content is at field capacity of 0.3.
Pathway: Direct dermal contact	1. Assuming that the human receptor wears T-shirt and shorts, the maximum exposed skin area is around 1500 cm^2 for adults and 500 cm^2 for children (EA, 2009). It is assumed that 10% of the exposed skin area will be in direct contact with effluent water and soil during the entire exposure period. 2. Soil-to-skin adherence factors (AF; Eq. 3.4) are assumed to be 1 mg soil/cm^2 skin for children and 0.3 mg soil/cm^2 skin for adults (US EPA 2004a; EA, 2009). Moya et al. (2004) report amount of adhered soil on children hands per event to be 0.26 g $hand^{-1}$ (the 95ile value). If it is assumed that the surface area of hands is 1% of the total body skin area (6600 cm^2 for a child), this corresponds to an adherence factor of about 4 mg soil cm^{-2}, which compares well to the chosen values. 3. It is assumed the concentration of the hazardous agent in the body tissues is negligible, thus rate of absorption is approximately proportional to the concentration in the adhered soil (USEPA 1992, Paustenbach, 2000). 4. Unless contaminant-specific dermal adsorption fractions (ABSd; Eq. 3.4) are available from literature, ABSd values of 0.1 for organic contaminants, 0 for inorganic contaminants and 0 for pathogens was assumed, due to limited data (Environment Agency 2009; USEPA 2004a, b). 5. Frequency of contact is assumed to be 1 event per day, similar to the risk assessment guidelines presented in US EPA (2004a) and EA (2009). 6. Contaminant-specific permeability coefficients through skin from water (K_p^w; Eq. 3.3) are taken from literature or estimated using available regressions (e.g. US EPA, 1992). It is assumed that K_p^w is 0.001 cm/hr for inorganics and 0 cm/hr for pathogens (i.e. no dermal uptake).
Pathway: Ingestion	1. Soil ingestion rates by human receptors as in Table 5.1. Inadvertent ingestion of effluent water ($R_{ig,wate}$r; Eq. 3.4) is assumed to be 5 ml/day. This corresponds to over 10% of the amount of water a child ingests during a swimming session (Dufour et al., 2006)
Receptor: Exposure	1. Time spent in the park/public green space by the different receptor groups have been based on statistics for 24-hour cumulative number of minutes spent outdoors at a park/golf courses presented in Tsang and Klepis (1996): i. 'Average Person' = 150 min day^{-1} ii. '95 %ile vulnerable person' = 580 min day^{-1} iii. 'Highly exposed infant' = 660 min day^{-1} 2. It is assumed that the different receptor groups spent 50 days every year in parks/sports turfs irrigated with effluent.

Table 5.3: Specific assumptions used in the QRA for Scenario 1 – Urban irrigation (non-edible).

5.1.1 Pathway and exposure calculation method

Key to the calculation of the exposure and risk for the irrigation scenarios (scenarios 1, 2 and 3) is to determine the likely pollutant/hazard concentrations in the soil following the irrigation with effluent. The procedure used for calculating the resulting concentrations in the soil is briefly described in the following. For more detail, please refer to Appendix I.

The calculations are based on a simple mass balance approach, as illustrated in Figure 5.2. It is assumed that effluent with solute pollutant concentration C_0 (mg/l or CFU/l) is applied to land using an irrigation rate of I (mm/day). From this, the daily amount of pollutant added with irrigation can be calculated and it is assumed that added pollutant mass is mixed uniformly and instantaneously in the soil over a pre-defined depth d_r. To determine how the pollutant concentrations in the soil develop over time following multiple irrigation events, the following general mass balance can be set up:

$$\rho_b \frac{dC_T}{dt} = \left(\theta_w \frac{dC_W}{dt} + \theta_a \frac{dC_a}{dt} + \rho_b \frac{dC_s}{dt}\right) = \frac{I}{d_r} C_0 - \left(\frac{I}{d_r} + \lambda\theta_w\right) C_W \quad (5.1)$$

where C_T is total hazard concentration in soil (mg/kg soil dry weight), $C_w/C_a/C_s$ are hazard concentrations in soil water/air/soil, O_w and O_a are the water and air contents in the soil volume with depth d_r, P_b is the dry soil bulk density and λ is a hazard-specific first-order degradation constant (day-1).

The mass balance in Eq. 5.1 assumes that a proportion of the hazard added to the soil via irrigation is removed from the system through water losses (plant transpiration and leaching) and degradation. All water flows and the soil water content within the considered volume are assumed constant with time. To solve Eq. 5.1, expressions describing the link between the concentrations in the different phases (soil, water, air) are needed. Different options for doing this exist depending on the type of pollutant considered, as described in detail in Appendix I.

For both organic and inorganic contaminants (i.e. PTEs), the Eq. 5.1 has been solved by assuming linear soil-water-air equilibrium partitioning, resulting in the following expression for the total concentration in soil (see Appendix I for details):

$$C_T(t) = \left(1 - exp\left(-\frac{k}{d_r} t\right)\right) \frac{I}{\rho_b k} C_0 \quad (5.2)$$

Where $k = \frac{I + \lambda d_r \theta_w}{(\theta_w + \theta_a K_H + K_d \rho_b)}$

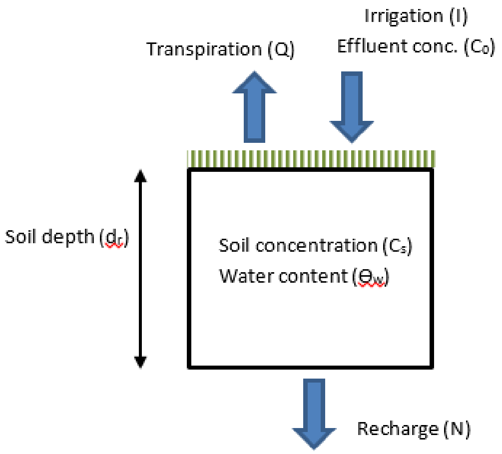

Figure 5.2: Conceptual model for the irrigation of land with effluent.

where K_d (l/kg) is an equilibrium distribution coefficient between soil and water, and K_H is the contaminant-specific Henry law's constant (only applicable to volatile contaminants). Eq. 5.2 describes how the contaminant concentration will develop over time in soil irrigated with effluent with concentration C_0. If irrigation is continued for a long time (i.e. when *t* approaches infinity), the system will eventually reach a steady state and the resulting concentration in soil will be constant:

$$C_{T,steady} = \frac{I}{\rho_b k} C_0 \quad (5.3)$$

For compounds that are non-volatile and do not undergo degradation (e.g. PTEs), the term $O_a K_H$ can be removed and is λ conservatively assumed to be zero in Eq. 5.2 and 5.3. As noted in Appendix I, the distribution coefficient K_d for metals is typically dependent on the soil pH, with metals tending to be in a more soluble form at acidic conditions. The effect of pH (and other soil parameters) on the sorption of metals will not be considered here and K_d values for metals are taken from the literature (Allison and Allison, 2005). However, various regressions exist in the literature to predict K_d for metals given pH and other soil parameters, which could be used.

The transport and retention of microbes through soil is more complex, as described in more detail in Appendix I. For the work here, it is assumed that the removal of microbes present the infiltrating water by retention follows a first-order process and that the air phase can be neglected, in which case the steady-state pathogen concentration in soil can be calculated as (cf. Appendix I):

$$C_{T,steady} = \left(1 + \frac{k_{att}}{k_{det}}\right) \frac{\theta_w I}{\rho_b (I + \lambda d_r \theta_w)} C_0 \quad (5.4)$$

where k_{att} and k_{det} are pathogen-specific first-order adsorption and desorption constants. As shown in Eq. 5.4, the amount of microorganism being retained in the soil will depend strongly on the microbe-specific ratio k_{att}/k_{det}. The higher this ratio is, the larger the number of microbes retained in the soil. Values for this ratio are, however, associated with considerable uncertainty. Schijven and Hassanizadeh (2000) reviews and discusses variations in reported k_{att}/k_{det} ratios for various viruses and find values ranging orders of magnitude (from 0.1 to over 1000). For E.coli, the adsorption coefficient k_{att} is usually found to be in the range between 10-4 to 10-1 min-1 (Engstrom, 2015), while the desorption coefficient k_{det} is often found to be about 2 - 100 times lower than k_{att} (Jiang et al., 2007; Bradford et al., 2006). We assume k_{att}/k_{det} to be 50 for all of the considered microbial hazards.

Note that Eq. 5.2 expresses the concentration development with time following consecutive effluent irrigation events assuming that the only water input to the soil is through effluent irrigation and hence does not account for dilution due to rainfall (i.e. 'irrigation' with clean water). A simple way of accounting for such dilution effects is to assume that irrigation repeatedly takes places for t_1 consecutive days followed by a period of t_2 days where irrigation does not take place, but where it rains with rate *I*. In this case, the concentrations in the soil will eventually reach and fluctuate around a steady level equal to the steady-state concentration in Eq. 5.3 (for chemicals) and Eq. 5.4 (for pathogens) multiplied by $t_1/(t_1+t_2)$.

5.1.2 Results

Given the assumptions presented above and in Table 5.3, it is possible to back-calculate what the concentration of the different hazards need to be in the final effluent in order for the receptors not to experience an unacceptable risk. Table 5.4 shows the results of the urban irrigation scenario for the selected microbial hazards. With the exception of *Listeria monocytogenes*, the concentration in the effluent of all of the selected pathogen needs to be below 1 CFU/l to avoid an unacceptable risk from exposure to effluent water and soils that have been irrigated with effluent. Very low concentrations of *Salmonellae Enteritidis* are required, because of its high r value (Eq. 3.6 and Appendix II), meaning that *S. Enteritidis* have high probability of causing infection once ingested. Relatively high concentrations of this pathogen in the final effluent would be required to experience an unacceptable exposure because of the high N_{50} value for *L. monocytogenes*.

For pathogens, dermal uptake is assumed negligible, and the most significant exposure route is through the inadvertent direct ingestion of effluent. For the Average Person, the dose experienced through direct effluent ingestion is about 100 times higher than that for soil ingestion. For infants, the pathogen dose through water ingestion is only about 5 times higher than through soil ingestion due to the relatively high soil ingestion rate assumed for this receptor group (Table 5.2).

Similar results can be produced for both the inorganic (PTEs) and organic contaminants, but for these two groups of hazards, dermal uptake is also accounted for. For the PTEs, the most significant exposure route is again through ingestion. For the Highly Exposed Infant receptor, the exposure of PTEs through dermal uptake is 80 times lower than via ingestion, which is mainly due to low PTE permeability coefficients (K_p^w) and the fact that they adsorb strongly to soil. For most of the organic contaminants considered, the most significant exposure route is via dermal uptake from effluent water with exposures that are generally 10 – 100 times higher than those for direct ingestion. However, for most of the pesticides considered, the main exposure route is found to be the direct ingestion of water, due to their lower K_p^w values and higher solubility in water.

The estimated maximum allowable concentrations in effluent of the considered hazards will be dependent on their toxicity (i.e. acceptable dose) and the extent to which they adsorb and are retained in the soil (as expressed through the distribution coefficient K_d for the contaminants or the k_{att}/k_{det} for the pathogens). The maximum concentration in effluent will also depend on how persistent the hazard is (i.e. whether it undergoes degradation). However, for the calculations here, degradation has been set to 0 for all of the organic contaminants. Figure 5.3 plots different maximum acceptable concentration limits in effluent ($C_{effluent}$) as a function of different K_d values and acceptable dose values. This graph shows that the higher the K_d value and the lower the acceptable dose value are for a given hazard, the lower the acceptable concentration limit has to be in order to avoid unacceptable health risk. This approach can be used as a quick guide to identify hazards that might be of concern for using effluent for urban irrigation.

In general, it is found that the considered hazards pose a limited risk to human health through any of the exposure routes. For example, the determined safe concentration levels for all of the PTEs are far greater than what is normally measured in untreated wastewater effluent; and this is despite the many conservative risk assumptions made in this assessment. This is also the case for most of the organic contaminants. A large variation in the acceptable concentration levels in effluent have been found for the different organic contaminants before an unacceptable human health risk will be experienced for this scenario. The considered dioxins can only be tolerated at very low concentrations, at levels down to just 1 ng/l. This is mainly due to their high toxicity (i.e. the acceptable dose is low for these contaminants).

5.1.3 Summary and other considerations

The risk assessment results suggest that the human health risks associated with using reclaimed water for irrigating public green spaces are very low. Some of the considered pathogens (especially S. Enteritidis) may have a potential to cause unacceptable risk of infections suggesting that a disinfection treatment may be required before effluent is used for irrigation, in line with the recommendations of, for example, the Californian standards. However, the assessment here is highly uncertain due to the many conservative assumptions that have been incorporated. For example, the assessment ignores degradation/decay, which could reduce the risks substantially, notably for the pathogens. The model used here accounts for exposure through ingestion of soil and effluent as well as dermal contact. This is a more risk averse approach than the one recommended in the Australian guidelines, where only direct ingestion of effluent is considered (assumed to be approx. 1 ml per exposure event).

Overall the findings here are consistent with other published

Microbial hazard	Max allowable concentration level in effluent (CFU/l)		
	Average person	*Vulnerable person*	*Highly exposed infant*
E. coli O157:H7	0.04	0.04	0.03
Campylobacter	0.02	0.02	0.02
Salmonella	6 x 10^{-4}	6 x 10^{-4}	5 x 10^{-4}
Cryptosporidium	0.09	0.09	0.07
Listeria	81	80	69

Table 5.4: Estimated maximum allowable concentration levels of the selected pathogens in the final effluent for the urban irrigation (non-edible) scenario. Based on steady-state concentrations

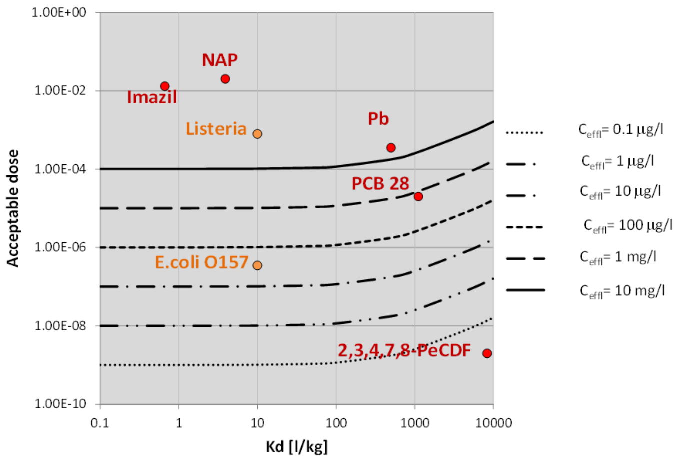

Figure 5.3: Scenario 1 – Urban irrigation (non-edible): Different maximum allowable concentration limits in treated effluent as a function of K_d and acceptable dose. The calculated concentration limits are for the "Average person" receptor using inputs from Table 5.1 and do not consider degradation and volatilization. Seven hazards have been added to the plot for illustration. Note that the K_d for the two pathogens (*Listeria* and *E. Coli O157*) have been estimated from the assumed k_{att}/k_{det} as: $K_d = O_w(k_{att}/k_{det})/pb$. Note that for pathogens, 1 mg/l corresponds to 1 CFU/l in the plot. Refer to Tables 4.2-4.4 for information on specific parameters and concentrations in effluent.

studies. For example, based on information obtained from the literature and other sources, Crook (2005) found no incidences of illness or disease from either microbial pathogens or chemicals, concluding that risks of using reclaimed water for irrigation of parks, playgrounds, athletic fields, and schoolyards with highly treated and disinfected reclaimed water are the same as using potable water. Alonso et al. (2006) also concluded that reclaimed wastewater should be submitted to tertiary treatment and disinfection processes before being used for irrigation purposes on golf courses to ensure that no health impacts resulted from microbial pathogens.

To reduce any potential human health risk, irrigation should take place at times when people are absent from the greenspace (e.g., during non-daylight hours) to reduce the potential of direct contact and warning signs could be put up to announce that irrigation with reclaimed water is taking place (Kopec et al., 1993). Table 2.1 lists other relevant application controls.

A concern in relation to the applicability of reclaimed water for irrigation of sports turfs and public parks is how the effluent water impacts on turf quality and appearance. This depends on a number of physical and chemical water quality parameters, such as dissolved salts (which can accumulate in the soil and inhibit moisture/nutrient uptake and impair soil structure), pH, toxic elements, organic carbon, suspended solids and nutrients (e.g. Toze, 2006; Kopec et al. 1993; Norton-Brandao et al. 2013). It has been demonstrated that the irrigation of soils with wastewater can significantly decrease infiltration (Thomas et al., 1966). Effluent water also typically contains substantial amounts of nutrients. Although these are beneficial to crops up to certain concentrations and potentially can lessen dependence on manufactured fertilizers, excessive nutrient concentrations in effluent applied to land can also lead to leaching and pollution of groundwater and surface waters. Effluent with high nutrient content may also stimulate growth of soil microorganisms and the production of biofilm, which can reduce the hydraulic conductivity of the soil (Toze, 2006). The impact and application of nutrients to land are already covered by existing regulations, for example, designation of Nitrate Vulnerable Zones as required under the EU Nitrates Directive (91/676/EEC).

5.2 Scenario 2: Urban irrigation – edible

In this scenario, final effluent is used to as a source of irrigation for urban agriculture (i.e. gardens and allotments used to grow edible produce, Figure 5.4). Specific assumptions used in this QRA are presented in Table 5.5. The scenario is similar to Scenario 1, but includes the risks from exposure to food crops grown on land irrigated with effluent.

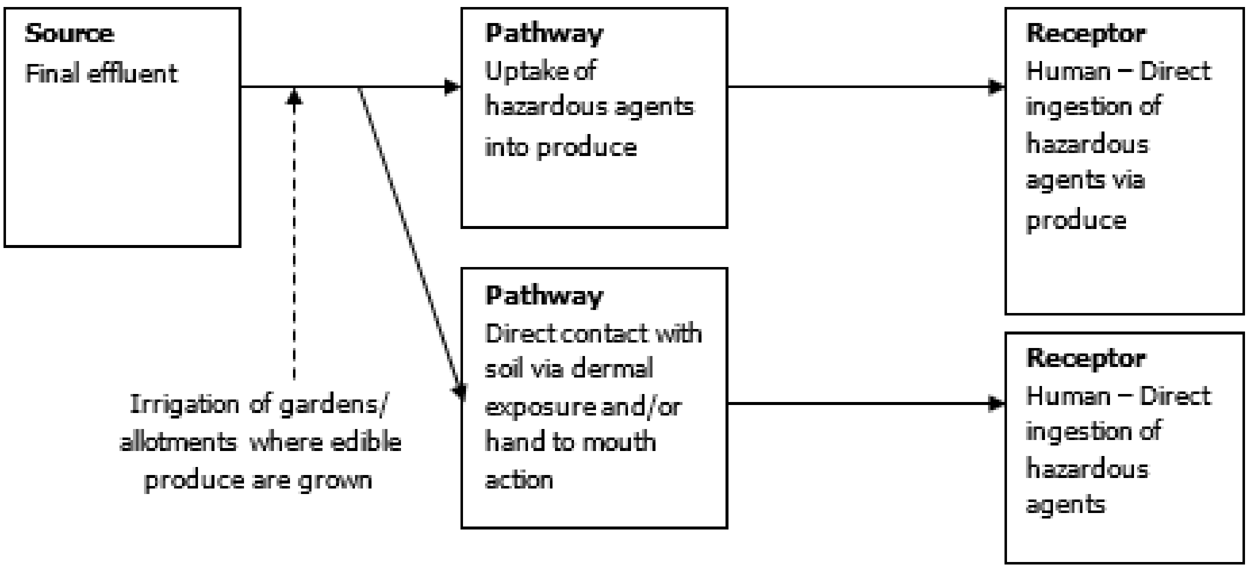

Figure 5.4: Source-pathway-receptor schematic of Scenario 2 – Urban irrigation (edible)

Stage: Variable	**Assumption**
Pathway: Irrigation of gardens/ allotments where edible produce are grown	1. Irrigation assumed to be the same as for Scenario 1, i.e. the irrigation rate is 4 mm/day and irrigation takes place 4 months every year. 2. Assume irrigation rate is uniform across planted soil areas. 3. Assume direct ingestion of water or inhalation of water vapour can be neglected and that effluent water is not applied directly on crops. 4. Assume that hazards/contaminants applied to land via effluent irrigation are mixed uniformly and instantaneously in the top 10 cm of the soil according to mass balance and fugacity principles (Mackay, 2001). 5. Assume soil bulk density is 1.5 kg/l, soil organic carbon content is 4%, soil porosity is 0.4, and soil water content is equal to a field capacity of 0.3.
Pathway: Uptake of hazardous agents into produce	1. Produce assumed to be grouped into root vegetables, leafy vegetables and fruit (similar to Hough et al., 2012). 2. All potentially hazardous agents have reached equilibrium in the soil. 3. Uptake of hazards into produce is modelled using hazard specific Bio Concentration Factors (BCFs) (see section 5.2.1) from the literature. Where BCFs are not available, the modelling of plant uptake of PTEs is done as per the methodology of Hough et al. (2004), while the method of Trapp & Legind (2011) is used for organic contaminants. Uptake of pathogens into produce modelled as in WRAP (2013)
Pathway: Direct dermal contact	1. All assumptions regarding dermal uptake are the same as in Scenario 1 (Table 5.3), except only dermal contact with soil is considered. It is also assumed that 20% of the total exposed skin area will be in direct contact with soil during the entire exposure period.
Receptor: Exposure	1. All produce is assumed eaten raw. 2. Soil and particles deposited on plant surfaces, which are not removed by washing assumed to be 5 mg/g plant. 3. Dietary intakes (kg d^{-1}) for human receptors as in Table 5.1 4. Direct water ingestion is not taken into account. Soil ingestion rates as in Table 5.1. 5. Time spent in the garden/working with soil are based on human activity survey data and statistics pre sented in Tsang and Klepis (1996): i. 'Average Person' = 100 min day^{-1} ii. '95 %ile vulnerable person' = 470 min day^{-1} iii. 'Highly exposed infant' = 300 min day^{-1} 6. It is assumed that the different receptor groups spent 52 days every year gardening (Tsang and Klepis, 1996)

Table 5.5: Specific assumptions used in the QRA for Scenario 2 – Urban irrigation (edible).

5.2.1 Pathway and exposure calculation method

A key input for this scenario is the estimation of the resulting hazard concentrations in crops grown on land irrigated with effluent. The uptake of different hazards into plants is complex and depends on a number of plant, soil and hazard-specific properties and processes. Plant uptake can occur via various pathways, the main ones being: i) uptake with soil water, ii) diffusion from soil or air, and iii) deposition of soil or airborne particles. The importance of the different pathways depends on both the hazard-specific and plant-specific properties. For some contaminants, concentrations in roots and leaves may even exceed the concentrations in soil, which among other things is due to the water content in roots (up to 95%) being higher than in soils (about 30%). The crop type determines which uptake processes are more likely to be dominant. For example, the accumulation of contaminants from soil will be higher for root crops than for tree fruits, while the accumulation by uptake from air is likely to be higher for fruits. The degree to which physiological plant-specific parameters such as leaf area, transpiration rate, water and lipid contents as well as growth rate affect the uptake is also highly dependent on the properties of the hazard of interest (Trapp and Legind, 2011).

The uptake of microbial hazards by plants presents a particular challenge. It is well-known that bacteria can colonize on the external tissue of fresh produce plants, but bacteria have also been detected within plant tissue, where they may be protected from postharvest sanitation processes, posing a potential health risk (Wright et al., 2013). However, the extent of internalization and the governing processes behind this are complex and not fully understood. A brief review of recent research into the uptake of pathogens is presented in Appendix I.

The uptake of hazards by plants can be estimated in different ways. A simple and commonly applied approach for doing this is through Bio Concentration Factors (BCFs), which express the ratio of hazard concentration in an organism (here, the crop plant) to contaminant concentration in the surrounding medium. Measurements of concentrations in plant tissues and concentrations in soil will yield a plant-to-soil BCF, given by:

$$BCF = \frac{C_{plant}}{C_{soil}} \qquad (5.5)$$

where Cplant is the concentration in plant tissues (mg kg-1 fresh weight) and Csoil is the total concentration in soil (ideally at steady state, but practically at harvest), which is estimated as described in section 5.1 and Appendix I. BCFs (or regression equations relating BCF to contaminant-specific properties) are usually determined through controlled experiments in the laboratory or in the field. It is important to note that BCFs will only be valid for the exact conditions under which they are estimated, i.e. for the specific plant, hazard and soil type used for the determination. Appendix II presents BCF factors for the selected hazards.

Where (reliable) BCFs are not available, the uptake of different hazards can be modelled as described in Appendix I. Due to the limited data available on uptake of pathogens, a fixed BCF of 0.0005 has been assumed for all pathogens and plant types, based on the work by Islam et al. (2004a, b) and Wright et al. (2013). Note that the BCFs used and reported here are relating total concentrations in soil (dry weight) to concentrations in fresh weight plant tissue. In the literature, BCFs are often reported in terms of dry weight plant tissue (e.g. Novotna et al., 2015) and these therefore need to be corrected for the water content of the plant.

The use of BCFs is a very simple approach. A range of more sophisticated mechanistic and empirical models capable of simulating plant uptake of different organic and inorganic contaminants exist. These models vary in complexity and usually aim at determining the uptake for specific crop types (for more details, see Appendix I). However, for simplicity and for the sake of consistency across all hazards, we opt for the BCF approach here.

5.2.2 Results

For all of the considered hazards, the most significant exposure route is through ingestion of vegetables and fruit grown in the soil treated with effluent, while direct ingestion of soil and dermal contact with soil are found to be much smaller. This is particularly the case for the metals and the organic contaminants, where the dose through vegetable consumption is at least 2 orders of magnitude higher than through soil ingestion and dermal contact.

Table 5.6 shows the concentration levels in effluent for selected hazards that are estimated to be safe for the purpose of urban agriculture. Similar to the previous scenario, it is found that the concentration in the final effluent of all of the selected pathogens, except *L. monocytogenes*, needs to be below 1 CFU/l to avoid an unacceptable risk from ingesting produce that have been irrigated with wastewater effluent. The predicted safe concentration levels in effluent for the microbial hazards are found to be less strict compared to the previous scenario, because inadvertent direct water ingestion is not accounted for. Very low concentrations of *S. Enteritidis* are again required due to its high probability of causing infection once ingested. Because of *L. monocytogenes*' high N_{50} value (Eq. 3.5 and Appendix II), relatively high concentrations of

	Average person	*Vulnerable person*	*Highly exposed infant*
Microbial hazard	**Max allowable concentration level in effluent (CFU/l)**		
E. coli O157:H7	0.22	0.10	0.24
Salmonella Entiritidis	4.0E-3	1.8E-03	4.4E-03
Cryptosporidium	0.42	0.19	0.46
Listeria monocytogenes	507	230	550
PTE	**Max allowable concentration level in effluent (mg/l)**		
Cu	11	2.6	2.2
Cd	0.04	0.009	0.008
Zn	152	34	29
Organic contaminants	**Max allowable concentration level in effluent (mg/l)**		
Naphthalene (NAP)	16	4.0	1.9
benzo-a-anthracene (B[a]A)	4.5	1.0	0.8
Chrysene	0.5	0.1	0.08
PCBs	0.06	0.01	0.01
Polychlorinated dioxins (e.g. 1,2,3,7,8-PeCDD)	74E-06	1.7E-06	1.3E-06
Polychlorinated dibenzofurans (e.g. 1,2,3,4,7,8-HxCDF)	7.3E-05	1.7E-05	1.3E-06
Clopyralid	0.22	0.05	0.07
Fenoxycarb	23	5.26	4.0
Imazalil	1.75	0.37	0.50

Table 5.6: Estimated maximum allowable concentration levels of the selected hazards in the final effluent for the urban irrigation (edible) scenario.

this pathogen in final effluent would be required to experience an unacceptable exposure. However, the results for the pathogens must be considered with caution, as they are based on a number of uncertain assumptions. Assumptions with major impact on the risk assessment results include: to what extent pathogens will be retained and survive in soils and plants post irrigation, uptake and internalization in plants, and the dose-response relationship. None of these elements are fully understood and subject to ongoing research.

Overall it is found unlikely that PTEs entering the food chain due to effluent being used for irrigating crops will present a risk to human health. Cadmium is found to be the PTE with the lowest allowable concentration level, with the calculations here suggesting that a Cd concentration above around 10 ug/l in the final effluent could pose an unacceptable risk to vulnerable receptor groups. The estimated allowable concentration levels for PTEs are far greater than what is normally measured in treated wastewater effluent (see Table 3.3), even for Cd, hence suggesting that PTEs are unlikely to pose a risk for this scenario. This is in agreement with the findings of Smith et al. (1996) that monitored two sites for heavy metals after they had been subject to long-term irrigation with secondary effluent. They found that the heavy metal concentrations were within normal background levels and concluded that it may take up to 100 years for heavy metal levels in effluent-irrigated soil to reach threshold values for environmental concern.

A large variation in the predicted acceptable concentration levels of organic contaminants in effluent is found (Table 5.6). The results suggest that the polychlorinated biphenyls (PCBs) and most of the polycyclic aromatic hydrocarbons (PAHs) and pesticides are unlikely to pose a serious human health risk for the considered scenario, as these chemicals need to be present in quite high concentrations in the effluent before the HQ (Eq. 3.7) exceeds 1. As the PCBs and most of the PAHs are very lipophilic ($\log K_{ow}$ ranging from 5 to 7) and have a low water-solubility, the uptake into plants with transpiration water is considered low (Mikes et al., 2009). Contamination of plants is mainly via attached soil particles or from air, while uptake from soil into the peel of some root crops may occur (Trapp and Legind, 2011). Passuello et al. (2010) assessed the risk of accumulating persistent organic pollutants (incl. PCB180, benzo(a)pyrene and dibenzo(a,h)anthracene) in the food chain following application of sewage sludge to agricultural land and found low risk to humans through oral intake (i.e. HQ << 1), which agree well with the modelling results in Table 5.6. The polychlorinated dioxins/furans (PCDD/F) can only be tolerated at very low concentrations. For many of the dioxins a threshold concentration level of around 1 - 10 ng/l might be of concern. This is mainly due to their high toxicity (i.e. very low RfD values). PCDD/Fs are considered persistent, semi-volatile and lipophilic and hence unlikely to enter plants with transpiration water. Plant uptake of these chemicals is again more likely to occur via air and/or from attached soil particles (Muller et al., 1993; 1994; Trapp and Legind, 2011). Of the contaminants considered by Passuello et al. (2010), 2,3,7,8-TCDD was found to pose the greatest risk to the food chain, but still with HQ-values well below 1.

The estimated maximum allowable hazard concentrations in effluent in this specific scenario depend on the toxicity (i.e. acceptable dose) and the extent to which the hazard is taken up by plants (expressed as the BCF). Figure 5.5 plots different maximum acceptable concentration limits in effluent as a function of different BCF values and acceptable dose values. This graph shows that the higher the BCF and the lower the acceptable dose value are for a given hazard, the lower the acceptable concentration limit has to be in order to avoid unacceptable health risk. When the BCF is below 0.0001, the acceptable concentration levels remain constant, because, at this stage, the daily dose via vegetable consumption is almost entirely due to the amount of soil attached to the plants.

5.2.3 Summary and other considerations

Although the results are associated with considerable uncertainty, the risk assessment suggests that the human health risks associated with using reclaimed water for irrigating urban agriculture are low. As was the case for the previous scenario, some of the considered pathogens (especially *S. Enteritidis*) may have potential to cause unacceptable risks of infections, which again suggest a disinfection treatment is required before effluent is used for irrigation. Overall, the findings here agree with other published studies on using reclaimed water for irrigating agricultural land, which is discussed in more detail in Section 5.3.

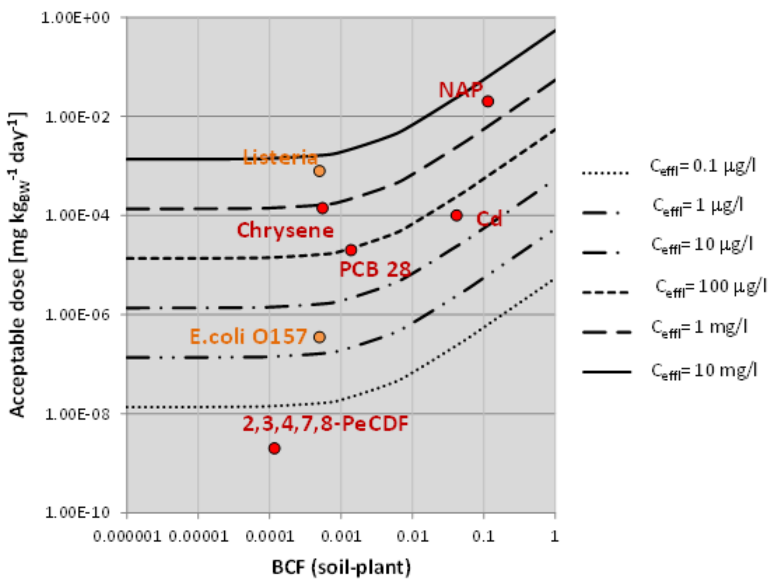

Figure 5.5: Scenario 2 – Urban irrigation (edible): Maximum allowable concentration limits in treated effluent as a function of BCF and acceptable dose. All other parameters are assumed constant. Calculated concentration limits are for the "Average person" receptor using inputs from Table 5.3 and do not consider degradation and volatilization. Seven hazards have been added to the plot for illustration. For pathogens, 1 mg/l correspond to 1 CFU/l in the plot. Refer to Tables 4.2-4.4 for information on specific parameters and concentrations in effluent.

5.3 Scenario 3: Agricultural irrigation

In this scenario, final effluent is used to as a source of irrigation for agriculture. Effluent can be applied to arable land or to pasture grazed by livestock. Receptors are therefore humans consuming food products and the livestock themselves (Figure 5.6). Specific assumptions are presented in Table 5.7.

In the UK, the water demand in the agricultural sector is relatively minor compared to industrial and household demand (Brown et al., 2012). According to Weatherhead et al. (2008), less than 2% of total abstractions are taken for direct use (mainly spray irrigation) by agriculture and horticulture in England and Wales, while less than 0.2% is abstracted for agriculture in Scotland. A smaller quantity is probably additionally taken via the public water companies and the drinking water supply (Weatherhead, 2008). Although these percentages are low compared to other parts of the world, the agricultural irrigation demand is increasing in many areas due to the increasing requirements for high quality produce from agriculture and horticulture (Dunn et al., 2004). In addition, the demand for agricultural irrigation is concentrated during the summer months when resources are scarcest. The demand for irrigation is predicted to increase in the future due to changes in climate and land use. Brown et al. (2012) estimated an additional irrigation demand of around 100 mm/yr in 2050 for Scotland.

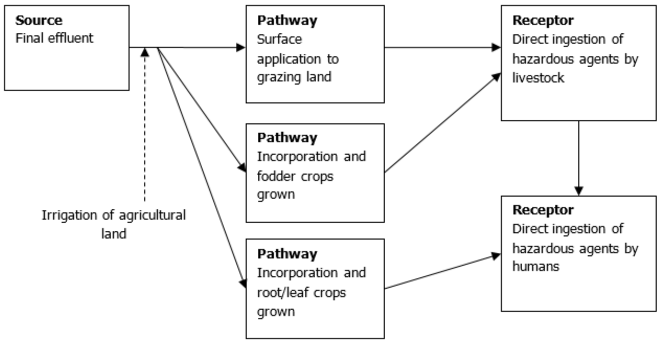

Figure 5.6: Source-pathway-receptor schematic of the agricultural irrigation scenario.

Stage: Variable	**Assumption**
Pathway: Application of final effluent to land	1. The irrigation rate is assumed to be 4 mm/day and takes place 4 months every year. The annual water demand in Scotland for agricultural purposes was around 300 mm; hence the assumed irrigation rate is in the high end. 2. Assume irrigation rate is uniform across all crop types. 3. It is assumed that irrigation takes place early morning or late evening, and that direct ingestion of water or inhalation of water vapour can be neglected. 4. Assume that hazards/contaminants applied to land via effluent irrigation are mixed uniformly and instantaneously in the top 10 cm of the soil according to mass balance and fugacity principles 5. Assume soil bulk density is 1.5 kg/l, soil organic carbon content is 4%, soil porosity is 0.4, and soil water content is equal to a field capacity of 0.3.
Pathway: Direct dermal contact	1. Assumptions regarding dermal uptake are the same as in Scenario 2 (Table 5.5), i.e. only dermal contact with soil is considered and exposed skin areas is 20%.
Pathway: Uptake of hazards into crops and grains	1. Uptake of hazards into produce is modelled in the same way as for Scenario 2 using hazard-specific Bio-Concentration Factors (BCFs) (see section 5.2.1) from literature. Where BCFs are not available, the plant uptake of PTEs is modelled as in Hough et al. (2004), while the method of Trapp & Legind (2011) is used for organics. Uptake of pathogens into produce modelled as in WRAP (2013). 2. Produce for human consumption assumed to be grouped into root vegetables, leafy vegetables, grains and fruit (Hough et al., 2012). 3. Assume all potentially hazardous agents to have reached equilibrium in the soil. 4. Soil and particles deposited on plant surfaces (except grains), which are not removed by washing assumed to be 5 mg/g plant.
Receptor: Exposure	1. Livestock intake rates as in Table 5.2. 2. Direct water ingestion by human receptors excluded. Soil ingestion considered for occupational receptors with soil ingestion rates as in Table 5.2. Human intakes of the different vegetable groups as in Table 5.2. 3. The fractional time of exposure through vegetable consumption is assumed to be 1 (Eq. 2.1). For occupational receptor, the fractional time of exposure of soil ingestion and dermal contact is assumed to be 0.5.

Table 5.7: Specific assumptions used in Scenario 3 - agricultural irrigation.

5.3.1 Pathway and exposure calculation method

The same exposure models used for Scenarios 1 and 2 can be used here to determine i) the hazard concentrations in soil and grass following irrigation to grazing/pasture land, and ii) the hazard concentrations in fodder crops or in crops grown for human consumption.

5.3.2 Results

The results are similar to those from Scenario 2, i.e. the human health risks associated with using reclaimed water for irrigation are low, with only some of the considered pathogens being of potential concern. The most significant exposure route for humans is through ingestion of fruit and vegetables grown on the soil treated with effluent, while direct soil ingestion and dermal contact with soil are much smaller. For grazing livestock, the concentrations in the effluent need to be stricter to avoid risks. This is due to the considerably larger direct intake of soil while grazing, and a greater ingestion of grass and vegetables relative to body mass. For hazards that are strongly bound to soil, the most significant exposure route for livestock is via soil ingestion, while grass ingestion is more significant for those hazards more readily taken up by plants (i.e. with higher BCFs).

Figure 5.7 shows different maximum acceptable concentration limits in effluent as a function of different BCF values and acceptable dose values, as predicted for cattle. Figure 5.7 shows that the higher the BCF and the lower the acceptable dose value are for a given hazard, the lower the acceptable concentration limit has to be to avoid unacceptable health risk. When the BCF is below 0.001, the acceptable concentration levels remain more or less constant, because the dose at this stage then is almost entirely due to soil ingestion. Note that in Figure 5.7, the soil-water distribution coefficient K_d is assumed constant for all hazards. In reality, K_d and BCF are likely to be related for most hazards, i.e. BCF would generally be expected to increase with decreasing K_d values.

Similar to Scenario 1 and 2, the pathogens and the dioxins appear to be of the greatest potential concern, while most of the other considered organic contaminants and the PTEs pose less (essentially negligible) risks. It has been estimated that the acceptable concentration of S. *Enteritidis* in treated effluent needs to be as low as 0.001 CFU/m^3 to avoid risks to grazing sheep and cattle.

Impact of fungicides and herbicides on grain quality and fermentation. A concern related to agricultural irrigation with effluent is how crop quality will be affected. A specific Scottish/UK concern is how the fermentation potential of brewing cereals might be impacted upon. A range of fungicides and herbicides may be present in effluent. Residues of herbicides and fungicides in soil have been demonstrated to negatively affect the yield and malting quality of grain (barley). For example, work by Brinkman et al. (1981) found that while light atrazine carryover had a minor effect on malting quality, heavy carryover reduced the quality considerably in five out of the six barley genotypes grown to maturity on soil treated with atrazine. However, work specifically looking at application of green waste compost to land found an increase in yield and no significant impact on grain quality (Cook et al., 1981).

Whether pesticide residues in effluent could pose a risk to grain quality and fermentation can be assessed by estimating the pesticide uptake into grains (using BCFs; Eq. 5.5) and then comparing the estimated concentrations in grain to published maximum residue levels (e.g. as established by the FAO/WHO). A good correlation with the K_{ow} (i.e. the octanol-water partition coefficient) and level of carry-over has been found (Miyake et al. 2002; 1999). While hydrophobic agrochemicals (i.e. $\log_{Kow} > 4$) generally remain in spent grains, spent hops, and spent yeast, hydrophilic agrochemicals ($\log K_{ow} < 4$) are more readily carried over into sweet wort, cold wort, and young beer (Miyake et al., 1999). Appendix II lists a range of fungicides and herbicides as well as some key properties (including maximum residue levels, BCFs, degradation rates in soil and $\log K_{ow}$).

5.3.3 Summary and other considerations

The assessment suggests that the risks associated with using reclaimed water for agricultural irrigation are low for both humans and livestock. As was the case for the previous scenarios, the main concern appears to be the pathogens considered, which may present an unacceptable risk of infections. This again suggests a disinfection treatment is required before effluent is used for irrigation, especially if the crops are to be eaten raw and/or if livestock are allowed to graze immediately after irrigation. It should be stressed that the results are associated with uncertainty. The assessment is based on several simplifying assumptions; though most of these are conservative and adopted to reflect (realistic) worst-case scenarios (e.g. consumption of raw vegetables).

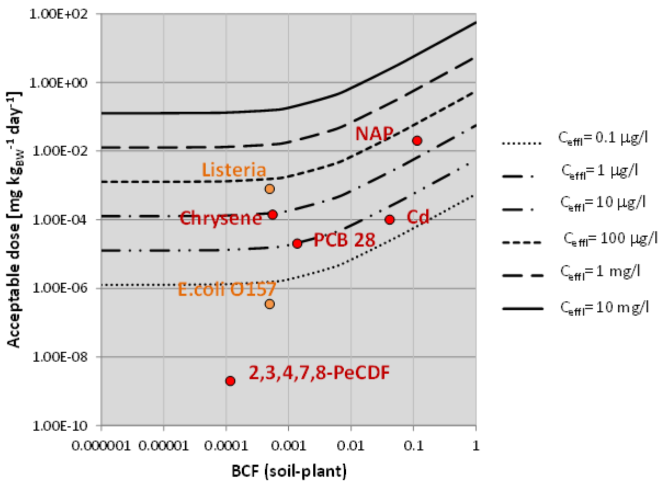

Figure 5.7: Scenario 3 – agricultural irrigation: Different maximum allowable concentration limits in treated effluent as a function of BCF and acceptable dose. All other parameters are assumed constant. The calculated concentration limits are for "Cattle" using inputs from Table 5.5 and do not consider degradation and volatilization. The soil-water distribution constant (K_d) is assumed constant. Seven hazards have been added to the plot for illustration. Note that for pathogens, 1 mg/l correspond to 1 CFU/l in the plot. Refer to Tables 4.2-4.4 for information on specific parameters and concentrations in effluent.

Much of the scientific literature and existing guidelines on reuse of treated wastewater are focussed on agriculture (Becerra-Castro et al., 2015; Norton-Brandao et al., 2013). The findings from this study are consistent with many other published studies, where reclaimed water is found to be safe for agricultural uses (e.g., York et al. 2008; Olivieri et al., 2007; Tanaka et al. 1998; Weber et al., 2006). Based on a quantitative microbial risk assessment, Olivieri et al. (2014) found that the risks of infection in humans from a number of pathogens (incl. enteric viruses, *Cryptosporidium parvum*, and *E. coli O157:H7*) associated with ingesting vegetables irrigated with tertiary treated effluent were generally below the acceptable probability limit of 1 infection per 10000 persons per year. In their risk assessment, they considered concentrations of *E.coli O157:H7* in raw wastewater of up to 5000 cells/l, which after undergoing wastewater treatment was assumed to experience an average log reduction of up to 6.53, hence corresponding to an effluent concentration around 0.001 cells/l. This concentration is also below the safe level found in this study. They concluded that agricultural practices consistent with the Californian water recycling criteria do not measurably increase public health risk. On the other hand, others have found serious public health concerns associated with pathogens in recycled water (LaPara et al, 2006; Lim et al., 2015). Lim et al. (2015) found that norovirus in treated storm water used for food crop irrigation could pose an infection risk to humans well above the 1 infection per 10000 persons per year limit. However, they considered estimated norovirus concentration levels in treated storm water in the range of 0.01-100 genomes/l. Note that the exposure assessment model used in our study is different to those used in Lim et al. (2015) and Olivieri et al. (2014) as well as in the recommendations of the Australian guidelines (EPHC et al., 2006), which consider exposure to microbial hazards by ingesting irrigation water retained in ready-to-eat crops.

The risk assessment here only considered irrigation of agricultural land for grazing livestock or food production. However, not all agricultural production is food crops. The same models applied here could in principle be applied to scenarios considering irrigation of forestry, horticulture or bioenergy crops. The human health risk associated with such scenarios is expected to be much lower and hence require less strict water quality standards. However, detailed risk assessments would be required to assess this and to ensure that other environmental receptors (e.g. wildlife, water courses) are not subject to unacceptable risks.

5.4 Scenario 4: Non-potable domestic - Toilet flushing

In this scenario, final effluent is used as a source of water for the flushing of toilets with potential exposures relating to aerosol inhalation (Figure 5.8).

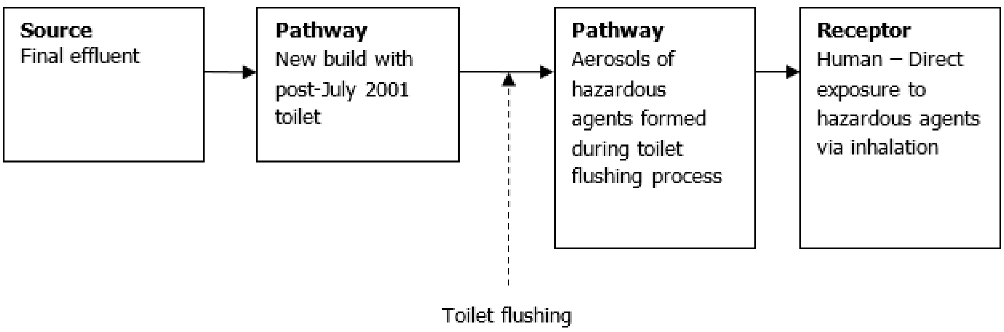

Figure 5.8: Source-pathway-receptor schematic of Scenario 4 – non-potable domestic (toilet flushing).

Stage: Variable	**Assumption**
Pathway: Aerosol formation of hazardous agents during toilet flushing	1. A fixed "transfer factor" of 1x10-5 (in units: ml bowl water/m3 air) of the bacterial/hazard load in the toilet bowl water is assumed to enter the air following the flush, regardless of the toilet type and whether the lid is left up or down (Barker & Jones, 2013; Lim et al., 2015). 2. Aerosol concentration assumed to remain constant over the post-flush exposure duration.
Receptor: Exposure	1. Time spent in the bathroom standing within aerosol range of the toilet post-flushing was assumed as (Tsang & Klepeis, 1996): i. 'Average person' = 25 min d-1 ii. '95%ile vulnerable person' = 90 min d-1 iii. 'Highly exposed infant' = 60 min d-1 2. Human inhalation rate as in Table 5.2.

Table 5.8: Specific assumptions used for Scenario 4 - Non-potable domestic (toilet flushing).

5.4.1 Exposure calculation

The human exposure to hazardous agents via aerosol inhalation is calculated as Eq. 3.2. The main challenge in calculating the exposure for this scenario is to estimate the likely concentration of aerosolised hazardous agents in air (C_{air}) following toilet flushing. It is well known that toilet flushing produces both large droplets and droplet aerosols. Several studies have shown that such droplets and aerosols can contain pathogenic bacteria and viruses (as well as other contaminants) and thereby may cause infection/exposure in humans (e.g. Gerba et al. 1975; Barker and Jones 2005; Johnson et al. 2013a,b; Best et al., 2012; Watkins et al., 2007). While it is well-acknowledged that large droplet contamination of toilet seats and lids, the surrounding floors, and nearby surfaces (Gerba et al. 1975; Barker and Jones 2005) presents a contact transmission risk, the potential for airborne transmission of e.g. infectious diseases is not as widely recognised. Airborne droplet nuclei develop when a fluid of pathogenic droplets evaporates. They are so small and light that they will not settle on surfaces due to gravity, but may remain suspended in air for several hours and hence may have the ability to cause infection, even long after a "contaminated" toilet has been flushed. The proportion of droplets that will either deposit on nearby surfaces or evaporate to form aerosols will depend, amongst other things, on evaporation rate and initial droplet size distribution, which in turn depend on toilet designs (Johnson et al., 2013a).

Barker and Jones (2005) studied the potential spread of infection caused by aerosol contamination of surfaces after flushing a domestic toilet (cistern size of 12 litres) located in a 2.6 m^3 room. They measured a bacterial concentration of 1370 CFU/m^3 in air adjacent to the toilet one minute after flushing. As the bacterial concentration in the bowl before flushing was around 108 CFU/ml, this suggested a "transfer ratio" of 1.37×10^{-5} ml/m^3 of the bacterial load in the toilet bowl entering each m^3 air following the flush. In the same study, the loading of viruses (MS2 bacteriophages) to air was found to be about twice as high as that for the bacteria. Both the bacterial and virus concentrations in air were found to decline with time after the flush (a 100-fold reduction after 60 min). The transfer rates to air were largely unaffected by whether the lid was left up or down.

Johnson et al. (2013) investigated 'bioaerosol' generation (using fluorescence microspheres) from three different toilet types. They found droplet nuclei generation rates of up to about one airborne fluorescence particle per 100 million in the bowl pre-flush. Particle concentration in the bowl pre-flush was 10^{12}/ml, while the particle concentration in air after flush was up to around 500 particles/m^3 air, suggesting a significantly lower "transfer ratio" to air (5×10^{-10}) than in Barker and Jones (2005). However, compared to the Barker and Jones study, the air samplers were located further away from the toilet, in a bigger room (5 m^3), considered toilets with lower flush volumes and the aerosol sampling was started five minutes after the flush and represent average values over a 30 minute period. Johnson et al. (2013) also measured droplet generation rates of up to 25000 droplet/L flushed water, with more droplets being produced with higher energy flushes. The majority of the droplets were smaller than 2 um in size.
It is assumed here that the (microbial) hazard loading to air, as aerosols and small droplets, following a flush will be proportional to the hazard concentration in the bowl water:

$$C_{\mathrm{air}} = \mathrm{TF}_{\mathrm{flush}} \times C_{\mathrm{effluent}} \tag{5.6}$$

where TF_{flush} is a constant "transfer factor" (in units: ml bowl water/m^3 air) of the hazard concentration in the toilet bowl assumed to enter the air following a flush. Based on the studies above, TF_{flush} is assumed to be 1×10^{-5} ml/m^3 regardless of the toilet type, flush volume/energy and whether the lid is left up or down. This "transfer factor" is similar to upper values used in a recent microbial assessment of the public health risks associated with using harvested urban storm water for toilet flushing by Lim et al. (2015), based on values reported in O'Toole et al. (2009).
Another critical determinant of the risks posed by aerosolised hazardous agents following toilet flushing is whether the hazards/pathogens are likely to undergo decay. Once deposited, many pathogens and viruses are known to be able to survive on surfaces for weeks or even months (Johnson et al., 2013b). For the exposure calculations here, it assumed that no decay occurs and that the aerosol concentration remains constant over the post-flush exposure duration.

The average number of times an individual flushes the toilet is reported to be around 4–6 times per day. In an Australian residential water use survey by Athuraliya et al. (2012), the average and the 95%ile flush frequencies were found to be 3.9 and about 8 flushes per person per day, respectively. Lim et al. (2015) assumed that the time spent in the bathroom after flushing the toilet would be between one to five min and that an individual would flush the toilet four times per day. This would correspond to an exposure duration of 4 – 20 min day^{-1}. For the assessment here, the fractional time of exposure is determined based on the time spent in the bathroom as reported in a National Human Activity Pattern Survey for the US EPA (Tsang & Klepeis, 1996). The values for each of the considered receptor groups are presented in Table 5.6. It is assumed that the aerosol concentration is constant throughout the exposure duration.

5.4.2 Results and discussion

Based on the assumptions above, it is possible to carry out calculations of the exposure of different receptors to different hazardous agents present in the final effluent, given that the contaminant concentration levels in the effluent are known. It is also possible to back-calculate what the acceptable concentration levels in the final effluent need to be in order for the receptors not to experience an unacceptable risk. Because it is assumed here that the amount of aerosolisation is not affected by the hazard type (i.e. the "transfer factor" from toilet bowl water concentration to air/aerosol concentration is the same for all hazards), the calculated dose for a given hazard depends entirely on the concentration level in the effluent. This also means that the back-calculated acceptable concentration levels in the final effluent for toilet flushing depend entirely on what the acceptable dose (e.g. RfD) of the given hazard is (for example, how toxic it is). This is illustrated in Figure 5.8, where the acceptable concentration levels in effluent are plotted as a function of the "transfer factor" and the acceptable dose. The values for four selected hazardous agents are also shown for illustration. Figure 5.8 shows the higher the proportion of hazards in the toilet bowl that are being aerosolised is and the more "toxic" the hazard (i.e. the lower the acceptable dose) is, the lower the acceptable concentration level in effluent must be in order to avoid risk to human health.

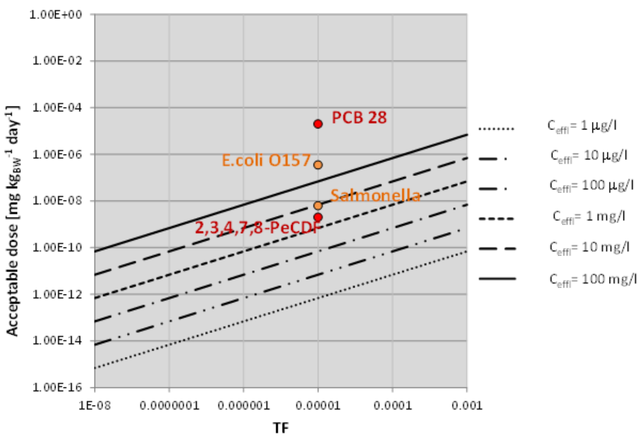

Figure 5.8. Scenario 4 - Non-potable domestic (toilet flushing): Maximum allowable concentration limits in treated effluent as a function of TF (transfer factor) and acceptable dose. The calculated concentration limits are for the "Highly Exposed Infant" receptor using inputs from Table 5.8 and do not consider degradation and volatilization. Four selected hazards have been added to the plot for illustration. Note that a concentration of 1 mg/l corresponds to 1 CFU/l for microbial hazards and a dose of 1 mg kg^{-1} day^{-1} corresponds to 1 CFU kg^{-1} day^{-1}. Refer to Tables 4.2-4.4 for information on specific parameters and concentrations in effluent.

Overall, using effluent for toilet flushing is unlikely to pose significant human health risk. The predicted acceptable concentration levels are far higher than the levels normally expected in treated effluent. For example, given the assumed transfer factor of 1×10^{-5}, a safe concentration for *E. coli O157:H7* of well over 100 CFU/l has been estimated. The results from the risk assessment agree with the findings in Lim et al. (2015), where toilet flushing with treated storm water was deemed as an acceptably safe practice based on a quantitative risk assessment of viral contamination. They considered adenovirus concentrations in storm water following a log-normal distribution (with a median concentration of around 12,000 genomes/l). Lim et al. (2015) also assumed that only a proportion of the inhaled aerosols will be deposited in the lungs and thereby may cause infection. As described in 3.1.2, our assessment is based on the total inhaled dose and hence is more conservative.

Existing guidelines on reclaimed water use from other countries do in some cases specify microbial standards for toilet flushing. For example, in California the microbial standard for toilet flushing (as well as for other uses) with reclaimed water specifies that the median total coliform concentration must not exceed 2.2 MPN/100 ml based on the last seven days' analyses (California Law, Title 22). This is far more stringent than the microbial standard in Japan, which is defined as 1000 total coliforms/100 mL of reclaimed water (Ogoshi et al., 2001).

5.4.3 Summary and other considerations

As is the case for all of the end-use scenarios, the above results are considered uncertain due to the many assumptions. For the toilet flushing scenario, the assumption regarding the degree of aerosolisation is particularly uncertain. The use of Eq. 5.6 to model the aerosolisation as being proportional to the hazard concentration in the bowl water is very simplistic. While this was found to be the case in the work by Watkins et al. (2007), other studies have not observed such a relationship (e.g. Johnson et al., 2013a; Barker & Jones, 2005). The bacterial loading to air will also depend on a range of other factors such as the microbial size, flush intensity and volume, bathroom size, evaporation rate, and air currents. Though Eq. 5.6 is simplistic, most other studies adopt a similar approach when assessing exposure from toilet flushing, e.g. the Australian guidelines also assume exposure to a fixed volume of water post toilet flushing.

The TF_{flush} value used here is conservative as it is based on experimental measurements for a toilet with a large flush volume (12 litres). The current Water Supply (Water Fittings) Regulations 1999 limit toilet cistern sizes to 6 litres. Since a toilet flushing system using effluent would require retrofitting, it is most likely to occur in new build properties and hence a 6/3 litre dual flush cistern is most realistic. The use of such cisterns is likely to generate fewer aerosols and hence pose a smaller risk. Also, most of the existing studies that investigate the risks associated with toilet flushing consider the spreading of microbial contaminants following acute cases of diarrhoea or vomiting, where contaminant loadings of the toilet bowl are many orders of magnitude greater than would be expected in treated wastewater effluent. However, even for such cases it has not been clearly demonstrated whether inhalation of aerosols generated from toilet flushing causes disease transmission so the significance of the risk remains largely uncharacterized (Johnson et al., 2013b).

The risk assessment here only considered human health risk due to inhalation of aerosolised hazardous agents. Other exposure pathways could be considered such as direct contact with hazards deposited on surfaces near the toilet or direct contact with the bowl water (e.g. house pet). However, even if these additional exposure pathways were included, it is still assessed that the use of treated effluent for toilet flushing is unlikely to present a significant public health risk. The main barriers and concerns are more likely to be the costs of installing such systems (which requires dual-plumbing) and the potential of cross-contamination with drinking water due to human error.

5.5 Scenario 5: Industrial (uncontained) - Industrial washing including vehicle washing

In this scenario, final effluent is used for industrial washing applications with potential occupational exposures relating to direct contact with final effluent and inhalation of aerosols during the washing process (Figure 5.9). The example here considers industrial washing of cars, the specific assumptions used for the QRA are presented in Table 5.9.

According to a report for the International Carwash Association by Brown (2002), many industrial car washes already have reclaim systems in place to conserve freshwater use.

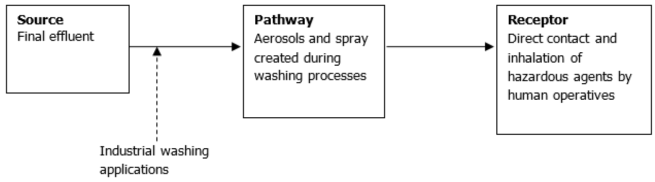

Figure 5.9: Source-pathway-receptor schematic of Scenario 5: Industrial (uncontained) – car washing

Stage: Variable	**Assumption**
Pathway: Aerosols and spray created during industrial car washing application	1. Based on measured particle size distributions of aerosols during car washing in O'Toole et al. (2009), it is assumed that the concentration of aerosols and very fine droplets is 0.01 cm3 per m3 air during and following car washing. 2. It is assumed that exposed skin will be in contact and covered with water during the entire car wash event duration.
Receptor: Human health	1. Exposed skin area is assumed to be 1000 cm2 comprising the face, hands and the lower arms (EA, 2009). 2. Dermal uptake of microbial hazards assumed to be zero.
Receptor: Exposure	1. Time spent in washing cars within aerosol range is assumed to be 7 hours per day, for 250 days per year. 2. It is assumed that a car wash event takes 10 min/car and the operator will wash 40 cars per day. 3. Inhalation rate as in Table 5.2. 4. A total direct water ingestion rate of 10 ml/day is assumed during car washing, which is of similar order to the proposed ingestion rate used for assessing exposure during firefighting in the Australian standards.

Table 5.9: Specific assumptions used in the QRA for Scenario 5: Industrial (car washing).

5.5.1 Exposure calculation

The human exposure to hazardous agents via aerosol inhalation is calculated using Eq. 3.2, and the specific assumptions presented in Table 5.9. The main challenge for this exposure pathway is to determine the likely concentration of aerosolised hazardous agents in air (C_{air}) generated during car washing. Car washing will generate mist droplets and aerosolized particles due to spray and evaporation. Fine droplets and aerosols can pose an airborne transmission risk through inhalation, but can also drift and be transported in air and settle on surfaces downwind. The concentration of hazardous agents in air (mg/m^3 air or CFU/m^3 air) can be estimated as follows:

$$C_{air} = C_{effluent} \sum_i C_{aerosol,i} V_{aerosol,i} \qquad (5.7)$$

where $C_{aerosol}$,i is the concentration of aerosols and fine droplets in air with median diameter size i (number of aerosols per m^3 air) , $V_{aerosol,i}$ is the volume of the aerosol and fine droplet particles with median diameter i (cm^3/aerosol) and $C_{effluent}$ is the concentration of hazardous agent in the effluent water (mg/cm^3 or CFU/cm^3). The term in the summation of Eq. 5.7 is essentially the total volume of aerosolized water per m^3 air.

Limited studies have investigated the generation and drift of droplets and aerosols during car washing. In a study of the water usage in the US professional car wash industry, the water losses due to evaporation and carryout were estimated by examining the volume difference between fresh water consumed and wastewater discharged for car washes of different types (in-bay automatic, self-service car and conveyor) and across three different locations (Brown, 2002). In this study, it was found that car washes used on average between 50 – 275 L vehicle-1 depending on car wash type (lowest water usage for self-service, highest for in-bay) and that on average between 15-30% of this water was lost due to spray, evaporation and carryout. However, it is difficult to convert this water loss to likely concentrations of droplets and aerosols in air.

O'Toole et al. (2009) measured the aerosolisation of particles of different sizes during 5 minutes of cleaning of a car door using, respectively, a high pressure spray unit (7.3 l min^{-1}) and a hand spray nozzle (11.8–15.4 l min-1). When using the high pressure spray, a visible fog formed, which persisted for several minutes after the high pressure device was turned off. A high variability in aerosol concentrations was observed and it was therefore not possible to detect statistically significant differences in emissions associated with the type of device and spray setting. However, the water-efficient device tended to generate more particles smaller than 2 μm in diameter (which are more likely to be inhaled and reach lower parts of the lungs) than the conventional device and hence the use of a water-efficient device may cause a higher exposure to these small particles. Based on the measured size distribution of aerosols during the car washing from O'Toole et al. (2009), the total volume of aerosolized water has been estimated to around 0.01 cm^3 per m^3 air (i.e. 1x10^{-5} L m^{-3}). Given a known effluent concentration (in mg/l or CFU/l), the concentration in air (mg m^{-3} or CFU m^{-3}) following a car washing event is therefore determined as: $C_{air} = 1\times10^{-5} \times C_{effluent}$.

The human exposure to hazardous agents via dermal uptake from water is calculated from Eq. 3.3, while the exposure due to inadvertent direct ingestion of water is calculated from Eq. 3.1. Specific assumptions are presented in Table 5.7.

5.5.2 Results and discussion

This scenario considers three exposure routes: inhalation, direct ingestion and dermal uptake. Given the assumptions above it is found that the exposure through inhalation is significantly lower than through the other two routes. For most of the organic contaminants, dermal uptake is found to be the most significant exposure route with exposures about 100 times higher than for direct ingestion of water and up to 10000 times higher than for inhalation. For most of the pesticides, the main exposure route is the direct ingestion of water, due to their lower K_p^w values and high solubility in water. However, organic contaminants pose a limited risk to human health through any of the exposure routes. The only exception is dioxins such as 2,3,7,8-TeCDD and 1,2,3,7,8-PeCDD, which are toxic and may be taken up easily through the skin (K_p values of around 1 cm/hr; US EPA, 1992). The acceptable concentration limits of dioxins in effluent have been estimated to around 1 ng/l. The PTEs pose an insignificant risk to human health for the car washing scenario. PTEs have relatively low K_p^w values and are hence less likely to be taken up through the skin, and the exposure to PTEs during car washing is mainly through direct ingestion.

Figure 5.10 (top) shows the acceptable concentration levels in effluent as a function of the "transfer factor" (total volume of aerosols (in litres) generated per m3 air) and the acceptable dose. It is observed that the higher the transfer/spray factor (i.e. the higher the proportion of hazards that are being aerosolised) is and the more "toxic" the hazard (i.e. the lower the acceptable dose), the lower the acceptable concentration level in effluent must be in order to avoid risk to human health. Figure 5.10 (bottom) depicts the results of the dermal uptake and shows the acceptable concentration levels in effluent as a function of Kpw and the acceptable dose.

5.5.3 Summary and other considerations

Overall, it is found that using effluent for car washing is unlikely to pose a significant human health risk. The exposure through dermal uptake is high for some hazards. However, the dermal uptake assessment is here based on conservative assumptions (e.g. all of the operator's exposed skin area is wet during and throughout all car wash operations) and the above results are associated with uncertainties. For this scenario, the assumption regarding the degree of aerosolisation and the amount of uptake via the skin are particularly uncertain. The approach used for assessing dermal uptake is furthermore often found to overestimate actual uptake. The inhalation and dermal contact risks to the operator could be reduced substantially by wearing personal protective equipment.

This scenario considered car washing, but there are many other potential industrial uses that could be considered. Reclaimed water is for example being used for cooling purposes. The most frequent water quality problems in cooling water systems are corrosion, biological growth and scaling. These problems arise from contaminants in potable water as well as in reclaimed water, but the concentrations of some contaminants in reclaimed water may be higher

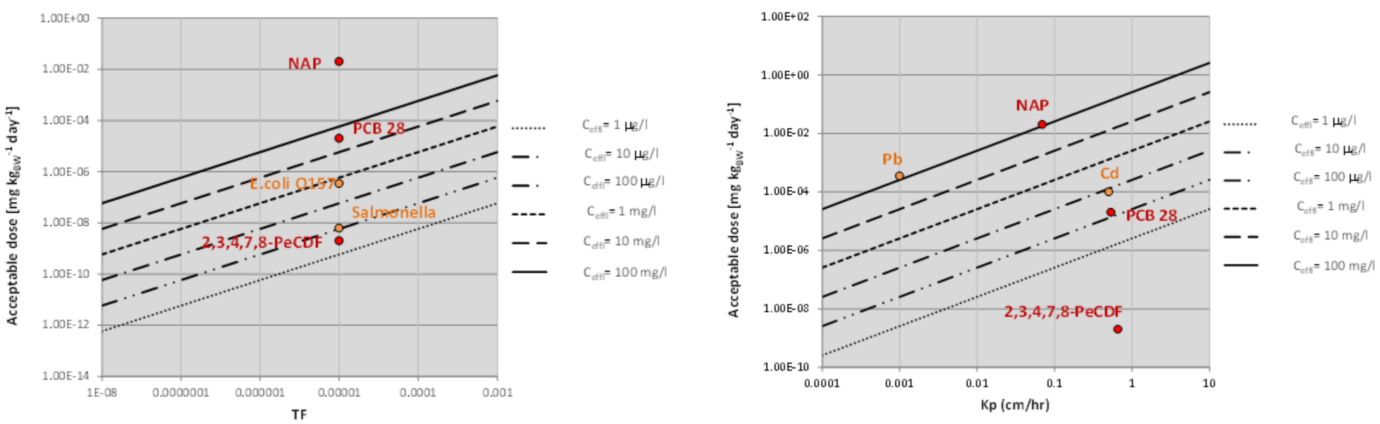

Figure 5.10: Scenario 5: Industrial (car washing). Top: Uptake via inhalation of aerosolised hazards. Maximum allowable concentration limits in treated effluent as a function of TF (transfer factor) and acceptable dose (e.g. RfD). The calculated concentration limits do not account for degradation and volatilization. Five selected hazards have been added to the plot for illustration. Note that a concentration of 1 mg/l corresponds to 1 CFU/l for microbial hazards and a dose of 1 mg kg^{-1} day^{-1} corresponds to 1 CFU kg^{-1} day^{-1}. Bottom: Dermal uptake from water. The values for five selected hazardous agents are also shown for illustration. Note that pathogens are not included as the dermal uptake of these is assumed negligible. Refer to Tables 4.2-4.4 for information on specific parameters and concentrations in effluent.

6 Stage 2: Stakeholder engagement

6.1 Introduction

To underpin the work of the risk assessment, input from Scotland's regulators, managers and commercial customers for water was achieved through stakeholder engagement. A full list of the stakeholders who contributed is provided in Table 6.1. This section of the report details the rationale, design, execution and outcomes of this engagement.

6.2 Rationale

Interest in using reclaimed water can be fostered through engagement with potential user communities (e.g. farmers, local authorities, commercial enterprises) and the consumers of their products (e.g. supermarkets, the general public) to raise awareness of the economic and environmental benefits of reclaimed water. This is seen as particularly important as the implementation of reclaimed water projects is not always universally welcomed. Water reuse projects around the globe have faced opposition from individuals and groups with concerns about disbenefits, primarily adverse health risks. In 2006, for example, the Australian city of Toowoomba rejected the council's plan for an indirect potable reuse project. In a referendum, 62% of the voters opposed the proposed scheme (2011, The Chronicle) supporting opposition claims that the reused water would be unsafe.

In order to mitigate, understand and accommodate potential opposition, stakeholder engagement has been widely embraced. Stakeholder engagement has evolved in a variety of contexts, including socio-technical developments, and has led to the emergence of a general set of principles which have been seen as helpful in promoting reclaimed water.

I. Dialogue needs to be sustained
II. Independent sources of information are available to everyone and not linked to the sponsoring agency
III. The community can ask questions
IV. The community is involved early
V. Decision process is generally accepted to be non-coercive, reasoned and fair
VI. There is a willingness to listen and expand the discussion if necessary
VII. Citizens have some level of control in the process (such as by contributing to the agenda or ground rules)

Significant resources have been devoted to engagement processes where reclaimed water projects have been accepted. California's Water Education Foundation (a nonprofit, nonpolitical, tax-exempt educational organization), established "to create a better understanding of water resources and foster public understanding and resolution of water resource issues through facilitation, education and outreach," reported expenditure of two million dollars in 2011 (the last year for which figures could be obtained)[1]. Australia and Spain also invest significant funds in engaging with consumers and customers. While investing in engagement processes cannot guarantee success, it is generally agreed that stakeholder engagement can facilitate better outcomes and that it requires investment. This is particularly true when the engagement commences in the early stages of a project allowing stakeholders to participate in shaping the outcomes and before a legacy of misunderstandings and failures, leading to distrust, has had the opportunity to accumulate. The following diagram (Fig. 6.1) from left to right, shows how this project aimed to run engagement to avoid 'firefighting' (typical where engagement has not been adopted at an early stage), move beyond the management of stakeholders, and achieve more effective engagement.

6.3 Design

The approach taken to stakeholder engagement in this project was early stage. Neither Scottish Water nor Scottish Government currently has any plans to develop reclaimed water in Scotland. The engagement component here was very much a scoping exercise to identify potential uses, explore barriers and opportunities to reclaimed water reuse and consider the shape of future guidelines. The approach therefore was designed to capture a broad range of inputs from interested parties through two stakeholder workshops at which the participants had a wide remit to influence proceedings.

6.4 Stakeholders

The research team worked with Scottish Water to draw up a list of stakeholders and to select a chairperson. A steering committee of Scottish Water managers was established to help guide the process. The selected stakeholders and chair were invited to become a fledgling National Stakeholder Panel and to attend two workshops facilitated by the researchers. The workshops were designed to achieve the following objectives:

- Explore the idea of a national-level stakeholder group (that would potentially outlive the project)
- Get feedback on the risk assessment developed in Stage 1
- Identify benefits and savings that reclaimed water can deliver
- Identify the perceived risks and barriers to use of reclaimed water in Scotland
- Identify specific sectors where reuse might be developed

[1] http://www.watereducation.org/sites/main/files/file-attachments/wef_annualreport_2011.pdf

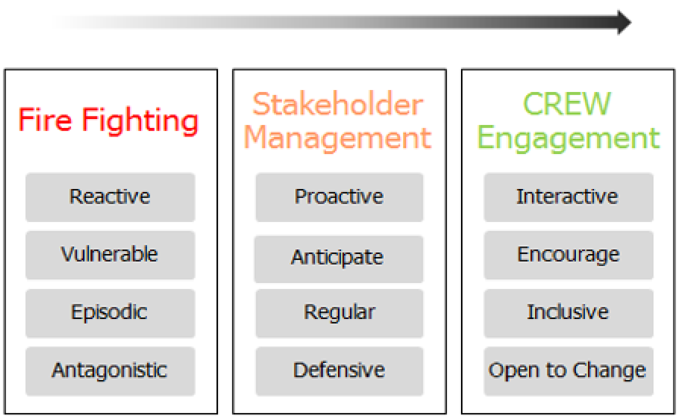

Figure 6.2: Benefits of early engagement Based on 'How to do Corporate Responsibility' - Doughty Centre Series

The original research plan specified the stakeholder engagement as a precursor to the technical risk evaluation; however, this was altered (to become Stage 2 of the research) to allow the stakeholders to contribute to an evaluation of the risk model developed in Stage 1. A consequence of this was that they were neither able to influence the initial selection of scenarios nor input into the design of the risk assessment. They were, however, encouraged to challenge the approach but did not register strong reservations or offer alternative scenarios.

Given the nature of the technical risk assessment being presented, stakeholders with some knowledge of the water industry were chosen. Choosing stakeholders who have knowledge relevant to the issue at hand for effective problem solving is an established principle in stakeholder involvement (Futaki 2010), although it is not without limitations. Other groups with a legitimate interest may have been excluded. Any follow-up engagement will need to consider whether wider composition, including the general public, is appropriate. In order to make a selection, researchers performed a stakeholder analysis resulting in a power/influence matrix (see Grimble and Wellard 1997) and identified a number of organisations to be approached. More participants were recruited for workshop 2 by the snowball technique whereby existing participants propose additional people. The organisations that the researchers were asked to consider were: Scottish Natural Heritage; forestry organisations; Health Protection Scotland; Confederation of British Industries (specifically in relation to the chemical industry); the Institute of Civil Engineering and The Chartered Institution of Water and Environmental Management (both for technical input); water stakeholder groups; the National Health Service; the Diffuse Pollution Management Advisory Group; Business Stream; Royal Environmental Health Institute of Scotland; and Planning Aid for Scotland. For logistical reasons it was only possible to accommodate a few of these nominations at the second workshop. Care was taken not to over-represent any particular organisation.

6.5 Engagement workshops

Two workshops were conducted; the first in October 2015 and the second in February 2016. The original project design incorporated an opportunity to carry out additional tasks depending on what the stakeholder group requested. There was a provision for up to 6 discreet activities described as sub-groups that would report back to the second stakeholder workshop. These subgroups had been conceived of as sectoral groups reporting back to the main group on specific questions directed by the stakeholder group. However, there was also an intention to allow the workshops to be shaped by the stakeholders following the principles of meaningful engagement (see Fig. 6.2). At the first workshop the stakeholders made requests for information reflecting a consensus that we are at an early stage in the process of building a foundation for water reuse in Scotland and that more general questions ought to take precedence at this point. They did not interact sufficiently with the risk assessment to specify sub groups but were highly engaged and able to set new tasks for the research team.

Workshop 1 was planned as an exploratory first step towards establishing a national stakeholder panel, bringing together interested parties to discuss how best to build an effective forum and to use this project to provide initial support. All present responded positively to being involved in the future and there was general assent to the proposal that both the forum and guidelines were needed.

The agenda comprised the following items:

I. Discuss the end-use, scenario-based risk assessment for using reclaimed water. The project team sought input from the stakeholders about the usefulness and the execution of the approach taken.

II. Identify the risks and barriers in the Scottish context. Discussion with Scottish Water had already outlined some of the 'benefits and limitations' but an exercise with stakeholders canvassed opinion more widely.

III. Setting the foundations for future policy/legislation. This item allowed a wide-ranging discussion about the direction water reuse in Scotland might take.

Affiliation	Name	Workshops Attended
Scottish Government (chair)	Jon Rathjen	1,2
Amec Foster Wheeler	Sandra Ryan	2
Business Stream	John Morgan	2
Citizens Advice Scotland	Gail Walker	1,2
COSLA[1]	Lorna Richardson	2
DWQR[2]	Sue Petch	1,2
Food Standards Scotland	Will Munro	1,2
Mining Institute of Scotland	Mark Friel	1,2
NFUS[3]	Andrew Bauer	1
RSPB[4]	Sheila George	1,2
Scottish Golf Union	Carolyn Hedley	1,2
Scottish Water	Adam Zyndul	1,2
Scottish Water	Roi Oterio	2
SEPA[5]	Janine Young	1,2
SEPA	Marcia Banks	2
Soil Association	Lillian Kelly	1,2
WICS[6]	Tom Sharples	1,2

[1] Convention of Scottish Local Authorities, [2] Drinking Water Quality Regulator, [3] National Farmers Union Scotland
[4] Royal Society for the Protection of Birds, [5] Scottish Environment Protection Agency, [6] Water Industry Commission for Scotland

Table 6.1: Workshop participants

6.6 End-use, scenario-based risk assessment

The risk assessment based on five scenarios for reuse using a source-pathway-receptor approach was presented, drawing particular attention to the assumptions made when assessing the risks. The decision to consider human health risks (a safety-first model) was supported by the group. There was some argument that economics plays a primary role and that an economic analysis ought to have been the basis of the project. There were also calls for more environmental risk assessments to underpin any future guidelines. Both these additional dimensions fell outside of the scope of this project but could form important elements of any follow-up work.

6.7 Savings, benefits, risks and barriers

The attendees were encouraged to think about the possible savings, benefits, risks and barriers to the use of reclaimed water. Workshop exercises captured a number of ideas and suggestions recorded in the word clouds below. These were fed back to the stakeholders in a summary report giving them the opportunity for further comment.

Other comments and questions focussed on the drivers, or lack of drivers, for reuse in Scotland. For example:

- *"There's plenty water in Scotland, it's different from water pressed regions in Europe"*
- *"Is this all about finding a use for water that SW has to deal with?"*
- *"What is the capacity (now and in the future)?"*
- *"If it happens it's going to be driven by necessity and opportunity"*
- *"Why not wait to see what comes out of Europe?"*

The benefits discussed in part mirror the benefits described in the introduction confirming what was already known. The discussion on risks and barriers elicited several concerns around governance and accountability which were discussed further in the second workshop.

6.8 Requested inputs for workshop two

I. Work on the so-called 'Yuck Factor' (i.e. public acceptability) of reused water came up repeatedly during both workshops and at other discussions with Scottish Water. A small study on 'Yuck Factor' in relation to water reuse projects was undertaken and reported back both to the steering committee and to the participants at the second workshop.

II. Stakeholders were aware of water reuse projects around the world and requested that the researchers provide additional information for the second workshop to discuss potential parallels to the Scottish context. The material requested focused on existing guidelines at a high level. The Californian context was specifically requested owing to the robust guidelines that appear to be in operation. (See Section 2).

III. The group identified a number of stakeholders missing from the table, some of whom were recruited for the second workshop.

6.9 Yuck Factor

The research team investigated the 'Yuck Factor' in relation to water reuse projects and reported back to the participants. The following summarises the findings:

Various projects, notably in Australia, the US (California), and Singapore, have encountered barriers to community acceptance of using recycled water, identified in the literature as far back as the 1970's, as 'yuck factor' experienced by potential users of recycled water. 'Yuck factor', refers to instinctive negative responses to new technology. Recycled water is thought to attract technophobic sentiments expressed in terms of disgust or 'yuck' caused by a psychological association that people make with raw sewage (Po, Kaercher et al. 2003). 'Yuck factor' is said to have been a contributory factor when Toowoomba, Australia, voted down a reclaimed water plan (2011).

6.9.1 What's in a name?

Research has been published to support the view that acceptance of water products can be adversely affected by the label used to describe the product and that the potential for eliciting disgust on the part of consumers can vary with the way the product is framed. Menengaki et al (2009) argue for a modest increase in willingness to both pay for and use artificially cleaned water when it is labelled recycled water. This description led to better responses than treated wastewater in their survey results. However, an earlier study found the term recycled water problematic (Leovy 1997) recommending the term repurified water (a term that was adopted in a San Diego project). A Singaporean project branded the product NEWater (Po, Kaercher et al. 2003).

The success or failure of branding strategies is difficult to assess given that confounding variables are difficult to control for. For example, in the context of a Scottish project, experience in other cultures may have limited relevance because of the variability in connotation of terms across different cultures. This issue of branding could be further explored by a national stakeholder group if this is developed. Notably, the stakeholders were sympathetic to the view that the product ought to be simply labelled as 'water'.

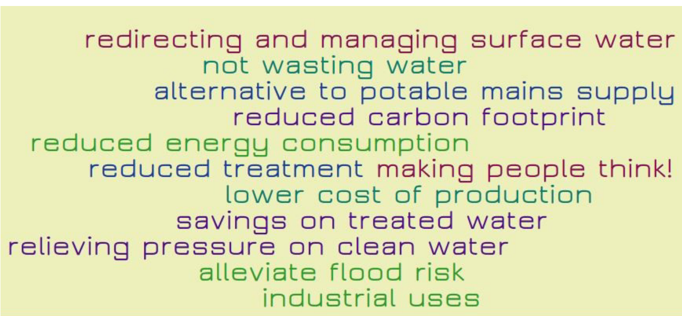

Figure 6.3: Savings/Benefits word cloud.

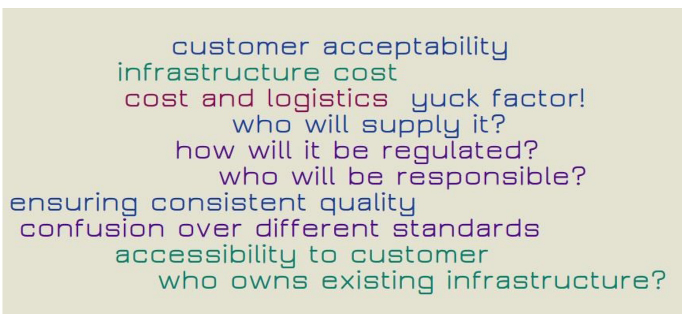

Figure 6.4: Risks/Barriers word cloud.

Alternatively, Russell and Lux (2009) downplay the importance of 'naming' and argue that sociological approaches can be applied to investigate cultural practices surrounding water use. The reasoning is that people's disgust originates in the way they view water use and that it's framing in language is an effect not a cause. Ching (2010) also looks beyond individual psychological responses and approaches 'yuck' as a social construct manifest through newspaper representations rather than a matter of public acceptability per se. The implication is that a more holistic view of what drives adverse attitudes is needed. Engagement is regularly promoted as a mechanism to explore some of the complexity surrounding social attitudes. The consensus in the literature advises involving the public in meaningful engagement and not trying to sell or persuade them about decisions that have already been made (Po, Kaercher et al. 2003). Research into the international contexts where reuse has been successfully implemented confirmed that all three cases examined (California (US), Australia and Spain)) had put significant resources into citizen engagement programmes. One element of engagement could be to conduct some national research on 'yuck' factor with the Scottish public.

6.10 International guidelines

Following a direct request from the first workshop the researchers developed three high-level case studies (see Section 2, Table 2.1). Material was presented to the second workshop in the form of tables listing the approaches in each international case study. An exercise was facilitated where the participants were divided into three groups to focus on California (CA), Australia and Spain.

6.10.1 Australia

Australia is a country with huge challenges caused by water scarcity and has a well-developed reclaimed water sector backed-up by guidelines. Participants identified similarities between Australia and Scotland. Both have large areas with little or no planning potential. In addition, the UK may be like Australia in terms of its regional outlook with very different water scenarios across the country. Given the similarities, Australia's project-specific approach was thought potentially useful for non-potable projects in Scotland.

Turning to dissimilarities, 'Yuck' was thought to be less of an issue for Scotland over Australia given that potable applications are not being proposed. Additionally, a central economic driver was held to be missing with no severe national water scarcity in Scotland and it was generally felt that the economics and practicalities do not currently add-up. The golf industry, in particular, was said to need a price incentive to use recycled water. Storage, transport and supply present barriers that would need to be overcome. The quantities needed for irrigating golf courses are massive (about 150 m3/day per golf course), and there were concerns that it seems neither practical nor financially viable to transport large quantities of effluent water over long distances where tap or groundwater supplies already exist. In addition, many golf courses already collect storm and rainwater for irrigation exemplifying how retreated effluent is only one source of recycled water. The subgroup concluded that small projects, following the Australian model, might be the best way forward for Scotland.

6.10.2 Spain

Spain is the biggest water recycler in Europe and many of its reuses match the potential reuses in Scotland identified by the project, including golf course irrigation and vehicle washing. Thinking about Spain's regional approach, the group were curious about whether different guidelines/standards might apply to different regions of Scotland. Another interesting parallel identified was the avoidance of drinking water applications in Spain which are likely to be mirrored if reuse is extended in Scotland. There appears to have been less public opposition in Spain as there are no potable applications, compared to countries where drinking water is affected. Research indicates that drinking water is the most sensitive area in terms of public acceptability (BIO by Deloitte 2015).

In terms of dissimilarities, it was recognised (as in all three case studies) that the central scarcity driver for Spain's reuse is not present in Scotland. It was also said that the governance culture is different between the two countries with the UK following regulations more strictly than Spain. This perception may create a barrier to developing guidelines in Scotland although the belief that other European countries apply European rules less strictly than the UK permeates other policy areas associated with European-level governance and has not stopped the adoption of regulation and guidance more generally.

6.10.3 California (CA)

California is seen by many as a benchmark for water reuse with a mature and substantial supply having been in operation for some decades. Knowledge of the Californian picture within the group led to criticism that the 'one-size-fits-all' operation there leads to overtreatment (gold plating) of reused water that would not suit Scotland where reuse would need to include lower quality water in certain circumstances. Instead, the idea of an integrated and holistic approach was favoured for the Scottish context.

6.10.4 Scottish water reuse

International comparisons spawned a wide-ranging discussion about Scotland's reuse potential. The stakeholders were concerned about predictable water shortages in Scotland with the example of Aviemore, where there are supply issues in the summer when population surges place demands on the water supply. Other examples that the participants highlighted included the west coast of Scotland where supplies are periodically under severe pressure in particular communities. There was also a recognition that reuse happens already. Full processing is bypassed in certain cases; for example, with Edinburgh street-cleaning and grey water schemes including toilet flushing (assessed in Section 5.4) in some new developments. This is a potential route forward through the promotion of good practice by identifying the uses already occurring and providing a consolidated source of information. There was a concern that current reuse contexts where little or no guidance exists ought not to be 'regulated for the sake of it.' Scottish Government could adopt a 'light touch' regarding any new regulation.

The discussion turned to drivers. Reuse in Scotland is not incentivised through water metering as it is in England. Metering was not thought to be a good option for Scotland. The idea of a 'green conscience' was identified as a potential driver; a notion echoed by the research findings that showed Australian outreach platforms directly appealing to environmental values. Other potential drivers for the Scottish context that can be followed-up included:

I. Remoteness: When the existing network is too far away making local treatment/reuse alternatives economically attractive
II. Capacity: When existing infrastructure has capacity issues requiring significant investment
III. Local initiatives: When local stakeholders have a strong appetite to test different approaches

Summing-up the plenary discussion, the Chair concluded that Scotland is not ready to discuss potable options for water reuse, so this should not feature in any current deliberations. Rather, the reuse context going forward is three- fold and should be investigated when there is/are:

I. Known supply problems where a solution is actively being sought
II. Specific new developments, which might range from housing developments to golf courses
III. Serendipity – when opportunities arise through chance circumstances

6.11 Recommendations from the workshops

The final plenary session comprised a round table to determine the next steps and is summarized by the following:

I. **Opportunity mapping:** There is a requirement for a needs analysis in Scotland. What does the supply and demand look like and where are the biggest opportunities (hotspots)? This is key to understanding whether guidelines/standards on effluent reuse are even needed (see also Brown et al. 2012). Particularly given the strong message from stakeholders that 'Scotland is not a water-pressed region', opportunity mapping was thought key to taking any reuse agenda forward.

II. **Identify Current Reuse Practices**: Water recycling activities are already taking place in Scotland. Can we learn from these and use them to define good practices?

III. **Future proofing** is an important consideration in relation to a needs analysis. Should Scotland include effluent reuse as part of a more flexible framework to deal with an unknown future? Although there is generally an abundance of water in Scotland (though not in all cases), it can still be questioned whether we will be able to meet the water demand in the future with the current water supply infrastructure (given e.g. future population growth and climate change).

IV. **Mapping and review of existing legislation** and policies relevant to effluent reuse aid; understanding what is already legislated for. What is already allowed? A layman's guide to existing 'do's and don'ts' might be valuable (see Section 2).

V. **National Stakeholder Panel:** The participants considered it too early for a nationally accountable panel and suggested that their contact details be held for a later stage in the development of this issue.

Final thoughts focussed on economics. A pessimistic notion about a forthcoming industrial decline over the coming 10-20 years citing the case of steelmaking was challenged by more optimistic speculation that climate change might drive the heavy industrial users of water (bottlers, brewers) to relocate in Scotland. Presently, it was acknowledged, there is no compelling reason to reuse water unless it becomes economically viable or legally required. For suppliers it was recognised that while there is harmonised pricing for customers there is not harmonised cost of supply. Therefore small or remote communities and other customers benefit by not paying the actual cost of supply. This structure diminishes customer incentives to seek alternative supplies. Against this, 'energy use' (that had been seen as a driver for reuse from the inception of this project) was viewed as a potential driver of water reuse in Scotland. With energy costs rising and carbon targets influencing policy, water reuse will become more urgent. From the water customers' perspectives, companies may already have individual sustainability targets (including energy). Industry drivers and incentives for effluent reuse might include corporate strategies to advertise and profile themselves on how they have substituted a certain amount of potable water with reclaimed water, thereby saving energy and emissions of greenhouse gases (echoing ideas of having a 'green conscience').

Other potential opportunities for using reclaimed water in Scotland can be summarized as follows:

I. Augmentation of water bodies under pressure. SEPA has done some mapping of this.
II. Big industries that use a lot of water (e.g. cooling, firefighting, car washing)
III. Irrigation
IV. Localised reuse schemes and/or new developments, i.e. if a WWTP happens to be located close to a golf course or an industry with high water demands.
V. Huge quantities of mining water (with a very different composition to effluent from municipal WWTPs) are currently stored in quarries and potentially represent a big resource.

7 Conclusions and perspectives

This project has focussed on developing the foundation for reclaimed water use in Scotland with the main objective of supporting the development of national guidelines. The research has been conducted in two stages. Stage 1 focussed on the development of national guidelines for reclaimed water use in Scotland and was completed via the following tasks:

I. *A review of existing guidelines for reclaimed water in other countries, focussing on the well-established standards in California, Spain and Australia*. It was found that the existing standards focus on ensuring that effluent reuse is safe to human health and the environment. However, the existing guidelines in other countries differ substantially, for example, in terms of the end uses considered, the approach taken, and the number/types of parameters considered and their associated limit values.

II. *A literature review of potentially hazardous agents present in final effluent*. A vast number of hazards may be present in final effluent at a range of concentration levels depending on the type and size of the wastewater catchment and the degree of treatment. Three groups of hazards were considered in this project: pathogenic microorganisms, potential toxic elements and organic contaminants.

III. *Identification of a series of potential end-use scenarios for reclaimed water relevant to Scotland.* It was found that many specific end uses of final effluent are covered in the guidelines in other countries. The typical application areas include agricultural, urban, domestic, industrial, recreational and environmental uses. Five specific end uses were selected in this project: irrigation of public green spaces and sports turfs, irrigation of allotments, agricultural irrigation, toilet flushing (domestic) and industrial car washing.

IV. *Development of exposure and risk assessment models for each of the considered end-use scenarios, focussing on assessing potential human health impacts*. The developed risk assessment models are based on the standard source-pathway-receptor principle and consider different exposure pathways and routes within each scenario. The models have been implemented into Excel and can be run in both forward and backward mode. The latter is useful for estimating what safe concentration levels in effluent need to be.

V. *Use developed risk assessment models to infer the levels of selected harmful agents in final effluent that will not result in unacceptable risks to vulnerable receptors and hence can be deemed safe.* The irrigation scenarios present a higher risk (especially those involving consumption of raw vegetables) compared to the domestic and industrial scenarios considered. Pathogenic microorganism was in all cases found to be the main human health concern, suggesting that a disinfection treatment would be required before use of effluent. Most of the other hazards considered were found to pose limited risks to human health. Our results are generally consistent with published literature. However, the risk assessments are associated with considerable uncertainty and based on risk-averse assumptions to ensure that risks are not underestimated.

Stage 2 aimed to form a national-level stakeholder group to evaluate the models developed in Stage 1 and to identify benefits and savings, as well as perceived risks and barriers to use of reclaimed water in Scotland. The stakeholders engaged in a constructive manner, however, were sceptical about the readiness of Scotland to embark on significant development of water reuse projects at this time and consequently thought that the development of guidelines was premature. The justification put forward pointed to the absence of effective drivers in terms of economic incentives, legislative pressures and the acute resource pressure caused by water scarcity affecting countries that are reusing substantial amounts of wastewater. Despite their overall doubts about the current prospects for reuse, they had a number of positive suggestions for taking the issue forward which are consolidated in our main recommendations.

7.1 Main findings and recommendations

To develop guidelines for effluent reuse in Scotland, the following aspects should be addressed:

i. Decide what underlying approach Scotland should follow when adopting national guidelines surrounding water reuse. For example, should the standards be similar to the Spanish and Californian models (where legally binding water quality and/or treatment criteria tailored to specific end uses are specified), or should they follow the more flexible Australian risk management approach? Although quantitative water quality standards are important, many experts agree that a broader risk-based approach is needed in order to provide sufficient reassurance of safety (BIO by Deloitte, 2015).

ii. Clearly identify which end uses of effluent should be covered in the guidelines and which end uses should be strictly forbidden. We have considered five specific and common end uses, which are covered in the guidelines in many other countries, but other end uses might be relevant to Scotland. The guidelines should cover (but not necessarily be limited to) any end use of reclaimed water for which there are potential markets. Future research should investigate and map such markets and opportunities in Scotland.

iii. The types of wastewater to be covered in the guidelines need to be clearly defined. In this project, only final effluent from WWTPs has been considered, but other sources of wastewater could be included, such as grey water and storm water. There are also large volumes of mining water stored in quarries across Scotland, which may present a valuable resource for reuse, however its composition is very different to effluent and specific standards may be required.

iv. Decide which and how many water quality parameters and indicators to monitor and focus on, their associated limit values and whether/how this can be tailored to specific end uses and WWTPs. Because wastewater effluent can contain a large mix of hazards and at broad concentration levels, it is infeasible to monitor them all. Therefore a suitable subset of water quality indicators needs to be identified. A good understanding and characterisation of the source water is also important, as this will potentially reduce the number, types and amounts of hazards to consider.

v. Identify and define relevant application controls for all considered end uses as a means of reducing risks. The review and the risk assessment work carried out here have identified a number of application controls relevant to the considered scenarios, many of which are also adopted in existing guidelines in other countries. For example, potential human health risk in the irrigation scenarios could be reduced by ensuring public access is restricted during and immediately after irrigation, as this will greatly reduce the likelihood of direct contact and/or ingestion of effluent water. Other application controls include setting up warning signs when and wherever use of reclaimed water is taking place and ensuring that operators/users that are likely to come in direct contact with effluent water (e.g. car wash operators) wear appropriate protective equipment.

vi. The current research has focussed primarily on the human health risks associated with effluent reuse. National guidelines should also ensure that effluent reuse is safe for the environment. The risk approach used here for human health assessment could be adopted to consider environmental receptors and should be considered in future research.

vii. There will be public opposition to water reuse judging by the experience of other countries although this is likely to be mitigated, the evidence shows, if the end use does not include drinking water. The so-called 'Yuck Factor' will need to be confronted with a range of public engagement initiatives that will require adequate resources. The engagement ought to be commenced in the early phases of preparation of any new policy in order to ensure maximum effectiveness.

The above points specifically concern the key requirements for establishing guidelines on effluent reuse. However, a key finding of this study is the seeming lack of any imperatives for reuse of treated effluent in Scotland, which is seen as a significant barrier for progress. There is therefore a clear need to identify the main drivers and circumstances under which water reuse could be the right answer and the following steps are recommended in relation to this:

viii. Opportunity mapping needs to be undertaken to continue the foundation building and to capitalise on any favourable conditions that may already be present. Pinch-points in the current system need to be identified. This is key to understanding whether and in what form guidelines on effluent reuse are needed.

ix. A cost benefit analysis is needed. The absence of a persuasive driver for Scotland's' water users, as reported by the stakeholders who participated in the study, may present a significant barrier for progress. In particular, there was thought to be no imminently foreseeable economic driver because of the relative abundance of water in Scotland. While it was conceded that this may change over time, currently the appetite for action is not strong. A cost benefit analysis will identify potential end uses and lead to a more focussed investigation on specificities.

x. Review of existing legislation, policies and practices relevant to effluent reuse. Compiling a register of current good practice might act as a spur for further development of reuse applications that may not require regulation. Mapping the existing legislative and policy landscape, a task that this report has contributed to, will also be an important part of the foundation for reclaimed water reuse.

The authors would like to thank all the stakeholders who participated in our workshops and particularly Scottish Water for contributing the time and ideas that made this study possible.
It is hoped that this report will stimulate debate and encourage additional work to be commissioned to further develop water reuse strategy in Scotland. We will continue to monitor the situation and will look for opportunities to contribute further should a suitable opportunity arise.

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9 Appendix I: Pathway and exposure models

9.1 Modelling concentration levels in soil following irrigation with effluent

In the following, the procedure used for calculating the resulting concentrations in the soil is presented. The calculations are based on a simple mass balance approach, as illustrated below.

It is assumed that effluent with solute pollutant concentration C_0 (ug/l or CFU/l) is applied to land using an irrigation rate of I (mm/day). From this the daily amount of pollutant (in ug day^{-1}) added with irrigation can be calculated as $m_p = C_0$ x I x A, where A is the area being irrigated. It is assumed that added pollutant mass is mixed uniformly and instantaneously in the soil over a pre-defined depth d_r, i.e. the pollutant mass is mixed into a soil volume of V_T = A x dr.

To determine how the pollutant concentrations in the soil develop over time following multiple irrigation events, a general water and pollutant mass balance can be set up, as follows:

change in pollutant mass in soil = pollutant added with irrigation (I) – pollutant removed by plant transpiration (Q) –pollutant removed by leaching/recharge (N) – pollutant removed by degradation

$$\frac{dm_p}{dt} = IAC_0 - QAC_W - NAC_W - \lambda V_T \theta_W C_W \tag{1}$$

where C_w is the pollutant concentration in soil water and 0w is the soil water content. Here it is assumed that a proportion of the water and pollutant added to the soil via irrigation is removed from the system through plant transpiration and leaching as well as through first-order degradation (with rate constant λ), and it is assumed that only the soluble fraction of the pollutant can be removed by these processes. Both the plant transpiration rate Q and the water content within the considered volume are assumed constant (e.g. water content can be assumed to be at field capacity), and to ensure water balance any excess water is assumed to leach, i.e. N = I – Q, when I > Q. In reality, the water content will increase in the soil following an irrigation event and gradually decrease through transpiration and leaching, and both the plant transpiration and leaching will depend on the water content, e.g. transpiration will cease when water content is at wilting point and leaching will only occur if the water content exceeds the field capacity following irrigation. For the purpose of determining the pollutant concentrations in soil, the assumptions above are considered appropriate and the fluctuations in the water content should have little impact on these concentrations.

The change in pollutant mass in the soil (Eq. 1) can be expressed as the sum of changes in pollutant concentrations in soil water C_w (mg/l), in soil pore air C_a (mg/l) and adsorbed to the soil C_s (mg/kg soil) as follows:

$$\frac{dm_p}{dt} = V_T \rho_b \frac{dC_T}{dt} = V_T\left(\theta_w \frac{dC_w}{dt} + \theta_a \frac{dC_a}{dt} + \rho_b \frac{dC_s}{dt}\right) \tag{2}$$

where C_T is the total concentration (mg/kg soil dry weight, 0w and 0a are the water and air content in the soil volume after irrigation and p_b is the dry soil bulk density. Combining Eq. 1 and 2 gives:

$$d_r\left(\theta_w \frac{dC_w}{dt} + \theta_a \frac{dC_a}{dt} + \rho_b \frac{dC_s}{dt}\right) = IC_0 - (I + \lambda d_r \theta_w) C_W \tag{3}$$

To solve Eq. 3, expressions describing the link between the concentrations in the different phases (soil, water, air) are needed. Different options for doing this exist depending on the type of pollutant considered, as described in the following.

Organic contaminants

For many organic contaminants, a common approach is to assume that soil-water-air equilibrium partitioning of the pollutant will occur instantaneously and is reversible. The partitioning between soil and soil pore water are in this case often described through a linear isotherm as $C_s = K_dC_w$, where K_d (l/kg) is an equilibrium distribution coefficient between soil and water. The higher the K_d is, the more the chemical will be bound to the soil. For volatile organic chemicals the distribution between the air and the water phases at equilibrium can be described by Henry's law as $C_a = K_HC_w$, where K_H is the contaminant-specific Henry law's constant. By inserting these two expressions into Equation 3, it is possible to solve the mass balance equation. The solution is:

$$C_w(t) = \frac{IC_0}{I + \lambda d_r \theta_w}\left(1 - exp(-kt)\right) \tag{4}$$

$$\text{where } k = \frac{I + \lambda d_r \theta_w}{d_r(\theta_w + \theta_a K_H + K_d \rho_b)}$$

From Equation 4, it is seen that when the system is in steady state (e.g. when t is very large), the soil pore water concentration will be equal to:

$$C_{w,steady} = \frac{IC_0}{I + \lambda d_r \theta_w} \tag{5a}$$

and the resulting total concentration in soil will be:

$$C_{T,steady} = \frac{I(\theta_w + \theta_a K_H + K_d \rho_b)}{\rho_b (I + \lambda d_r \theta_w)} C_0 \tag{5b}$$

For non-volatile chemicals, the term 0_aK_H can be removed, while λ = 0 for non-degradable pollutants.

Note that Eq. 4 expresses the concentration development with time following consecutive effluent irrigation events assuming that the only water input to the soil is through effluent irrigation and hence does not account for dilution due to rainfall (i.e. 'irrigation' with clean water). A simple way of accounting for such dilution effects is to assume that irrigation repeatedly takes places for t_1 consecutive days followed by a period of t_2 days where irrigation does not take place, but where it rains with rate I. In this case, the concentrations in the soil will eventually reach and fluctuate around a steady level equal to the steady-state concentration in Eq. 5 multiplied by $t_1/(t_1+t_2)$.

PTEs/inorganic compounds

For metals, the same equations as for the organic chemicals can be used, but neglecting the effects of volatilization (0_aK_H =0) and degradation (λ = 0) and the steady-state concentration in pore water is hence equal to C_0, while the total soil concentration becomes:

$$C_{T,steady} = \frac{(\theta_w + K_d \rho_b)}{\rho_b} C_0 \tag{6}$$

However, for metals the distribution coefficient K_d often depends on pH (as well as other soil parameters). In general, the sorption of metals increases with increasing pH, while at acidic conditions metals tend to be in a more soluble form and hence be more available for uptake into plants. Models based on a pH-dependent Freundlich relationship can be used to describe metal solubility in soils and sorption (Jopony and Young 1994). This approach can be used to predict free metal ion activity in the soil pore water [M^{2+}] from total soil metal content and soil pH, as seen in Equation 7. Soil metal content, [M_C], is assumed to be adsorbed on humus and has units of mg of a specific metal per kg of soil organic carbon.

$$p[M^{2+}] = \frac{p[M_C] + k_1 + k_2 pH}{n_F} \tag{7}$$

where nF is the Freundlich power term (equal to 1 for a linear sorption) and k_1 and k_2 are empirical constants. For a linear sorption, the distribution coefficient K_d for metals can hence be expressed as:

$$\log(K_d) = k_1 + k_2 pH \tag{8}$$

We are here assuming a pH of 7. Table A1.1 presents the Kd values used for the selected metals.

Pathogens
The transport and retention of microbes through soil is more complex. The transport is similar to that of colloids, but because microbes are living organisms that are influenced by a range of biological and external environmental factors, their behavior and transport in soils is much more complex. In general, the retention of microbes in soil is mainly due to adsorption to phase interfaces (e.g. soil-water and water-air interfaces). The retention will, among other things, depend on the size of the microbes and the soil pore-size distribution.

The equilibrium assumption used for the organic chemicals and metals above is often found to provide a poor description of the adsorption of microbes in soil. Instead, the removal of microbes present in the infiltrating water by retention in soil is commonly described as a first-order process:

$$\rho_b \frac{dC_s}{dt} = k_{att}\theta_w C_w - k_{det}\rho_b C_s \tag{9}$$

where k_{att} and k_{det} are first-order adsorption and desorption constants. If k_{det} is assumed to be 0, the attachment of microbes to the soil surfaces is considered irreversible, in which case Equation 6 reduces to the classical colloid filtration equation, which is commonly used for evaluating microbial transport behavior in laboratory and field-scale studies (Tufenkji, 2007). Inserting Eq. 9 into Eq. 3 and assuming that the air phase can be neglected gives:

$$\frac{dC_w}{dt} = \frac{IC_0}{\theta_w d_r} - \left(\frac{I}{\theta_w d_r} + \lambda + k_{att}\right) C_w + k_{det}\frac{\rho_b}{\theta_w} C_s \tag{10}$$

It is possible, but more difficult, to solve the coupled Eq. 9 and 10 to get expressions for how the bacterial concentrations in soil and soil water change with time. At steady-state, these are:

$$C_{w,steady} = \frac{IC_0}{I + \lambda d_r \theta_w} \tag{11}$$

$$C_{s,steady} = \frac{k_{att}}{k_{det}} \frac{\theta_w I C_0}{\rho_b (I + \lambda d_r \theta_w)} \tag{12}$$

In the literature, the first-order rate coefficients k_{att} and k_{det} have been estimated for a number of microorganisms based on controlled laboratory experiments (e.g., Jiang et al., 2007; Engstrom et al. 2015; Bradford et al. 2006). Furthermore, various quasi-empirical expressions exist for estimating the adsorption rate constant k_{att} based on the flow velocity, soil water content and the sizes of the microbe/colloid and of the soil particles (Tufenkji 2007). In this context, the opportunity for and frequency of the transported particles to collide with immobilized soil particles is usually expressed as the collector efficiency (n), while the percentage of the microbes that finally attached to particles is indicated by collision (sticking) efficiency (a). The desorption of microbes from the soil-water interface is determined by the hydrodynamic shear force and attachment strength. Higher water flow velocity can remobilize attached microbes into the flowing water. Also, a change of water pH or ionic strength could result in detachment of adsorbed microbes (Jiang et al., 2007).

As shown in Eq. 12, the amount of microorganism being retained in the soil will depend strongly on the microbe-specific ratio k_{att}/k_{det}. The higher this ratio is, the larger the number of microbes retained in the soil will be. Values for this ratio are however associated with considerable uncertainty. Schijven and Hassanizadeh (2000) review and discuss variations in reported k_{att}/k_{det} ratios for a number of viruses and find that the ratio can vary by orders of magnitude (from 0.1 to over 1000). For E.coli, the adsorption coefficient k_{att} is usually found to be in the range between 10-4 to 10-1 min-1 (Engstrom et al., 2015), while the desorption coefficient k_{det} is often found to be about 2 - 100 times lower than k_{att} (e.g. Jiang et al., 2007; Bradford et al., 2006). We will here assume that k_{att}/k_{det} is 50 for all of the considered microbial hazards.

9.2 Modelling uptake of hazards into plants

Uptake of organic contaminants by plants
Organic contaminants may enter crop plants through several pathways, the main ones being: i) uptake with soil water, ii) diffusion from soil or air, and iii) deposition of soil or airborne particles. The importance of the different pathways depends on both the contaminant-specific and plant-specific properties (Trapp and Legind, 2011). Experiments and model simulations have shown that that persistent, polar (log K_{ow} < 3) and non-volatile (dimensionless Henry's constant < 10–6) contaminants generally have the highest potential for accumulation from soil into plants. Concentrations in roots and leaves may even exceed the concentrations in soil (in some cases by several orders of magnitude), which is partially due to the water content in roots (up to 95%) usually being higher than in soils (about 30%). Volatile contaminants generally have a low potential for accumulation, because they quickly escape to air (Trapp and Legind, 2011).

The crop type is decisive for which uptake processes are more likely to be dominant. For example, the accumulation of contaminants from soil will be higher for root crops that for tree fruits, while the uptake from air is higher for fruits. The degree to which physiological plant-specific parameters such as leaf area, transpiration rate, water and lipid contents as well as growth rate affect the uptake is highly dependent on the properties of the contaminant of interest. Water soluble contaminants will usually be rapidly translocated from soil to leaves, and the accumulation in leaves will in this case almost entirely be decided by transpiration rate (Trapp and Legind, 2011).

The uptake of contaminants by plants can be estimated in different ways. A simple way of doing this is through bioconcentration factors (BCFs), which express the ratio of contaminant concentration in an organism (here, the crop

PTE	Log(Kd) [kg/l]	Comment
Copper (Cu)	2.7 (0.1-3.6)	
Chromium (Cr)	1.0 (-0.7-3.6)	
Cadmium (Cd)	2.9 (0.1-5.0)	Regression log(K-d)=0.64*pH-1.53
Lead (Pb)	4.0 (0.7-5.0)	Regression log(K-d)=0.42*pH+1.99
Nickel (Ni)	3.0 (1.0-3.8)	Regression log(Kd)= 0.6*pH-1.59
Zinc (Zn)	3.0 (-1.0-5.0)	Regression log(Kd)= 0.89*pH-3.16

Table A1.1: K_d values for the selected PTEs (Allison and Allison, 2005; Anderson & Christensen, 1988; Christensen, 1985).

plant) to contaminant concentration in the surrounding medium. Measurements of concentrations in plant tissues and concentrations in soil will yield a BCF plant to soil, given by:

$$BCF = \frac{C_{plant}}{C_{soil}} \quad (13)$$

where C_{plant} is the concentration in plant tissues and C_{soil} is the concentration in soil (ideally at steady state, but practically at harvest). BCFs (or regression equations relating BCF to contaminant-specific properties) are usually determined through controlled experiments in the laboratory or in the field. It is important to note that BCFs will only be valid for the exact conditions under which they are estimated, i.e. for the specific contaminant and soil type used for the determination.

A range of mechanistic models cable of simulating plant uptake of organic contaminants furthermore exists (e.g., Fujisawa, 2002; Hung and Mackay, 1997; Passuello et al., 2010; Paterson et al., 1994; Rein et al., 2011; Trapp, 2004; Trapp and Matthies, 1995). These models vary in complexity and usually aim at determining the uptake for specific crop types. Many of these models are based on a multimedia modelling principle, where mass balances are set up and combined for the different compartments considered (e.g., soil, roots, and leafs). Assuming equilibrium partitioning, this leads to relatively simple ordinary differential equations that can be solved analytically. These models are then used to simulate the partitioning, transfer, and fate of chemical pollutants within and between the different plant compartments. The processes and their parameterization depend on the type of crop and the contaminant properties.

For organic contaminants for which reliable BCF factors are not available, the standard plant uptake model described in Trapp and Legind (2011) is applied instead. The model includes the soil, roots, and leaves (or grains) compartments and is cable of accounting for: i) continuous and/or pulse input to all compartments, ii) uptake into roots with the transpiration water, iii) translocation from roots to leaves/grains with the transpiration stream, iv) loss from leaves to air, v) deposition from air to leaves, vi) transport to leaves with attached soil, vii) growth dilution, degradation and metabolism in roots and viii) loss from soil due to degradation, leaching, run-off and plant uptake. To maintain the precautionary approach, only the steady-state solution for a continuous source concentration is applied here. Finally, deposition of particles on the surfaces of leaves or grains is neglected and uptake from air is assumed solely by diffusive exchange in the gas phase. The steady-state expressions are given by:

$$C_{roots} = \frac{Q}{\frac{Q}{K_{rw}} + k_r M_r} C_{w,soil} \quad (14)$$

$$C_{plant} = \frac{\frac{Q_p}{K_{pw}} C_{roots} + A_p g_p C_{air}}{\frac{K_H}{K_{pw}} A_p g_p + k_p M_p} \quad (15)$$

where C_{roots} and C_{plant} are respectively the concentrations in the roots and plant (here: leaves or grains), $C_{w,soil}$ and C_{air} are the concentrations in soil water and air, respectively, K_H is the dimensionless Henry's constant, and k_r and k_p are first-order growth rates of the roots and leaves/grains, respectively. K_{rw} and K_{pw} are the equilibrium partition coefficients between roots and water and between leaves/grains and water, respectively. These can be determined through the following empirical expressions:

$$K_{xw} = W_x + 1.22\, L_x (K_{ow})^b \quad (16)$$

where W_x and L_x are the water and lipid content of either roots, leaves or grains and *b* is a correction factor for differences between solubility in octanol and sorption to plant lipids. Based on previous studies, *b* can be assumed to be 0.77 for roots and 0.95 for leaves/grains (Trapp and Legind, 2011).
Plant-specific parameters and inputs used for the calculation are shown in Table A1.2.

As seen from the above equation, the concentrations in soil water and air are needed to estimate the accumulated concentrations in roots and leaves/grains. These can be estimated using fugacity modelling (Mackay, 2001) as described in Section 1 of this appendix.

Although more sophisticated plant uptake models capable of simulating the dynamic behaviour of the soil-plant system exist, the simpler approach described above for estimating the uptake of organic contaminants into crop plants is considered appropriate for risk assessment purposes. The steady-state solution is likely to overestimate the concentrations in the crops by orders of magnitude, which is in line with the precautionary approach used throughout this project.

Symbol	Input [unit]	Value
Roots		
W_r	Water content of roots [L/kg]	0.89
L_r	Lipid content of roots [L/kg ww]	0.025
Q	Transpiration stream [L/d]	1
M_r	Root mass [kg ww]	1
k_r	First-order growth rate [1/d]	0.1
Leaves/ grains		
A_p	Area of leaves [m2]	5
	Area of grains [m2]	1
W_p	Water content of leaves [L/kg]	0.8
	Water content of grains [L/kg]	0.15
L_p	Lipid content of leaves/grains [L/kg ww]	0.02
M_p	Mass of leaves/grains [kg ww]	1
p_p	Density of leaves/grains [kg ww/L]	1
g_p	Conductance of leaves/grains [m/d]	86.4
k_p	First-order growth rate for leaves/grains [1/d]	0.035
Q_p	Transpiration stream for leaves [L/d]	1
	Transpiration stream for grains [L/d]	0.2

Table A1.2: Default input data set for the standard model for the calculation of plant uptake (normalised to 1 m2 of soil). Values from Trapp and Legind (2011).

Uptake of Potentially Toxic Elements (PTEs)
Plant uptake of inorganic agents (i.e. PTEs) is estimated using models previously developed by the authors (Hough et al., 2004; Hough et al., 2003; Hough, 2002). The uptake of PTEs by plants is highly dependent on soil chemical properties such as pH. Uptake also varies according to crop type, even varying by cultivar. Metal uptake by vegetables is often modelled using regressions of the following form:

$$\log[M_{plant}] = C + \beta_1[pH] + \beta_2 \log[M_C] \quad (17)$$

where $[M_{plant}]$ is the metal concentration in the plant (mg kg^{-1}), $[M_C]$ is the total soil metal content, which is assumed to be adsorbed on humus (mg of a specific metal per kg of soil organic carbon) and C, *B*1 and *B*2 are empirical metal- and vegetable-specific coefficients. The use of $[M_C]$ in Equation 17 requires values for organic carbon content (% C). Equation 17 has been parameterised to estimate uptake of PTEs by 18 different fruit, vegetable, and cereal crops with relatively good results. The regressions from Hough et al. (2004) will here be used for quantifying the uptake of PTEs into plants, as a supplement to published BCFs.

Uptake of microbial hazards by plants
It is well-known that bacteria can colonize on the external tissue of fresh produce plants, but bacteria have also been detected within plant tissue, where they may be protected from postharvest sanitation processes, posing a potential health risk (Wright et al., 2013). However, the extent of internalization and the governing processes behind this are complex and not fully understood.

The reviews by Deering et al. (2012) and Hirneisen et al. (2012) present an overview of recent research and highlight the ability of foodborne pathogens to be taken up and internalized into a wide range of plant hosts in both roots and leafy tissue. Solomon and Matthews (2005) surface-irrigated mature lettuce plants with *E. coli O157:H7* or with FluoSpheres (fluorescent microspheres used as a bacterial surrogate) and harvested them 1, 3, and 5 days after exposure to analyse the internalization of *E. coli O157:H7* in the lettuce plants. They observed presence of FluoSpheres and *E. coli O157:H7* in the internal portions of the plant tissues following surface sterilization (with *E. coli* concentrations of up to about 700 particles/g tissue) and suggested that the entry of *E. coli O157:H7* into lettuce plants may be a passive event as the concentration the pathogen was similar to that of FluoSpheres. Wright et al. (2013) studied the internalisation of *E. coli O157:H7* in spinach and lettuce. After surface sterilization with gentamicin, they found that the number of individual spinach plants supporting gentamicin-protected bacteria was higher than lettuce: 81% compared with 23%, respectively, for the leaves and 91% compared with 31%, respectively, for the roots. The average number of internalized E. coli recovered from lettuce and spinach leaves was 2.66 and 3.24 CFU/g (log10), respectively, and from lettuce and spinach roots was 3.70 and 3.60 CFU/g (log10), respectively, corresponding to about 0.5% of the total population for each tissue type tested.

Islam et al. (2004a, 2004b) studied the persistence and fate of *E. coli O157:H7* in soil and on various vegetables (carrots, onions, lettuce and parsley) following amendment with composts or with contaminated irrigation water. They showed that *E.coli O157:H7* could persist in the soil and be detected on the vegetables for several months after application of compost or irrigation water. Their results generally suggested that the total vegetable-associated *E. coli O157:H7* cell numbers decreased by about 2 log CFU/g over a two-month period (post application), although differences were observed for the different plants grown (e.g. *E. coli* seemed to persist for longer on parsley and carrots compared to lettuce and onions). Their results also suggest that the total plant associated pathogen populations (in CFU/g) were about 1 to 100 times lower than in the surrounding soil.

For estimating the uptake of pathogens into plants/crops, we will here assume a fixed uptake factor (i.e. bio-concentration factor; Eq. 13) of 0.0005 for all of the considered pathogens and plant parts. This factor is based on the observations from Islam et al. (2004a, b) and Wright et al. (2013), but is obviously associated with large uncertainty.

10 Appendix II: Input values for selected hazards

Table A2.1: Selected hazards for the risk assessment scenarios.

Hazard group	Hazard	$logK_{ow}$	K_d	K_p^w [cm/hr]	ABS_d [-]	Decay coefficient [day^{-1}]	Safe exposure limits	BCF [25-27]		
								Roots	Leaf	Fruit/grain
Microbial hazards (pathogens)	*E. coli O157:H7*	n.a.	10 *	n.a.	n.a.	Soil: 0.11 (0.05-3.5) [1-4]	2.58×10^{-5} CFU/day [5]	5×10^{-4}	5×10^{-4}	5×10^{-4}
	Campylobacter	n.a.		n.a.	n.a.	Soil: 0.10 [4]	1.44×10^{-5} CFU/day [5]	5×10^{-4}	5×10^{-4}	5×10^{-4}
	Salmonella	n.a.		n.a.	n.a.	Soil: 0.11 [4]	4.63×10^{-7} CFU/day [5]	5×10^{-4}	5×10^{-4}	5×10^{-4}
	Cryptosporidium	n.a.		n.a.	n.a.	Soil: 0.05 [4]	6.54×10^{-5} CFU/day [5]	5×10^{-4}	5×10^{-4}	5×10^{-4}
	Listeria	n.a.		n.a.	n.a.	Soil: 0.11 [4]	5.84×10^{-2} CFU/day [5]	5×10^{-4}	5×10^{-4}	5×10^{-4}
Potentially Toxic Elements (PTEs)	Cu	n.a.	500 [24]	0.001 [7]	0.001 [7,22,23]	n.a.	RfD = 4.00×10^{-2} mg kg^{-1} day^{-1} [6]	0.067	0.067	0.067
	Cd	n.a.	800 [24]			n.a.	1.00×10^{-4} mg kg^{-1} day^{-1} [8]	0.042	0.042	0.042
	Pb	n.a.	10000 [24]			n.a.	3.50×10^{-4} mg kg^{-1} day^{-1} [9]	0.003	0.003	0.003
	Ni	n.a.	1000 [24]			n.a.	0.02 mg kg^{-1} day^{-1} [10]	0.033	0.033	0.033
	Zn	n.a.	1000 [24]			n.a.	0.3 mg kg^{-1} day^{-1} [11]	0.033	0.033	0.033
Polycyclic Aromatic Hydrocarbons (PAHs)	Naphthalene (NAP)	3.3	4 (est.)	0.07 [7]	0.13 [7,22,23]		2.00×10^{-2} mg kg^{-1} day^{-1} [12]	3.38E-01	3.40E-03	3.32E-03
	benzo-a-anthracene (B[a]A)	5.76	1414 (est.)	0.95 [7]			1.40×10^{-3} mg kg^{-1} day^{-1} [13]	1.75E-03	1.12E-04	2.31E-05
	Benzo-a-pyrene (B[a]P)	6.13	3429 (est.)	1.24 [7]			1.4×10^{-2} mg kg^{-1} day^{-1} [13]	7.24E-04	1.69E-05	3.39E-06
	benzo-b-fluoranthene (B[b]f)	6.57	9836 (est.)	2.54 [7]			1.40×10^{-3} mg kg^{-1} day^{-1} [13]	2.53E-04	1.67E-05	3.39E-06
	chrysene (CHR)	5.81	1594 (est.)	1.03 [7]			1.40×10^{-4} mg kg^{-1} day^{-1} [14]	1.55E-03	5.55E-05	1.11E-05
Polychlorinated Biphenyls (PCBs)	PCB 28	5.66	1113 (est.)	0.53 [7]	0.14 [7,22,23]		2.0×10^{-5} mg kg^{-1} day^{-1} [15-19]	2.21E-03	9.64E-04	4.53E-04
	PCB 101	6.33	5536 (est.)	0.61 [7]				4.50E-04	3.65E-04	9.80E-05
	PCB 153	6.87	20174 (est.)	0.90 [7]				1.24E-04	1.07E-04	2.33E-05
	PCB 180	7.16	40402 (est.)	0.90 [7]				6.18E-05	2.41E-05	4.91E-06
Polychlorinated dibenzo-dioxins and –furans (PCDD/Fs)	2,3,7,8-TeCDD	6.42	6868 (est.)	0.75 [7]	0.03 [7, 22,23]		2.00×10^{-9} mg kg^{-1} day^{-1} [20]	3.63E-04	1.35E-04	2.99E-05
	1,2,3,4,6,7,8-HpCDD	8.37	732487 (est.)	4.26 [7]			2.00×10^{-7} mg kg^{-1} day^{-1} [20]	3.41E-06	2.94E-06	5.90E-07
	1,2,3,4,7,8-HxCDF	7	27542 (est.)	0.92 [7]			2.00×10^{-8} mg kg^{-1} day^{-1} [20]	9.06E-05	1.20E-05	2.42E-06
Pesticides and herbicides	Clopyralid	-2.63	2.7E-06 (est.)	1.7×10^{-6} [7]	0.25 [7, 22,23]		2.00×10^{3} mg kg^{-1} day^{-1} [21]	4.09E+00	1.31E+02	2.62E+01
	Fenoxycarb	4.07	24.7 (est.)	0.02 [7]	0.25 [7, 22,23]		5.60×10^{-2} mg kg^{-1} day^{-1} [21]	8.17E-02	5.51E-02	1.10E-02
	Imazalil	2.56	0.66 (est.)	1.9×10^{-3} [7]	0.25 [7, 22,23]		1.30×10^{-2} mg kg^{-1} day^{-1} [21]	9.53E-01	7.15E+00	1.45E+00
	Pentachlorophenol	5.12	305 (est.)	0.20 [7]	0.25 [7, 22,23]		3.00×10^{-2} mg kg^{-1} day^{-1} [21]	7.89E-03	8.85E-04	1.88E-04

*K_d for pathogens is estimated from k_{att}/k_{det} as: $K_d = 0w(k_{att}/k_{det})/pb$

References

[1] Avery et al.,2005; [2] Oliver et al., 2006; [3] Hutchison et al., 2004;[4] Nicholson et al. 2005; [5] Based on an acceptable probability of 1.0 x 10-4 infections/pers/year and parameters in Table A2.2 [6] Hérbert, 1993 [7] US EPA, 1992 [8] USEPA, 1995 [9] Mushak et al., 1989; [10] Ambrose et al. 1976; [11] Yadrick et al. 1989; [12] Shopp et al., 1984 [13] Hoogenboom et al., 2003; [14] USEPA, 2002; [15] Arnold et al., 1993a; [16] Arnold et al., 1993b; [17] Tryphonas et al., 1989; [18] Tryphonas et al.,1991a; [19] Tryphonas et al.,1991b; [20] COT; [21] WSDOT, 2006; [22] Environment Agency 2009; [23] USEPA 2004a; [24] Allison and Allison, 2005; [25] Trapp & Legind (2011); [26] Novotna et al. (2015); [27] Islam et al. (2004)

Table A2.2: Dose-response model parameters for selected microbial hazards (WRAP, 2013).

Name	Dose-response model			Receptor and exposure route	References
	Beta-Poisson		Exponential		
	N50	*A*	*r*		
E. coli O157:H7	1130	0.16		Human (based on Shigella)	Strachan et al., (2001), Crockett et al. (1996)
			2.18×10^{-4}	Swine (oral)	Cornick & Helgerson (2004)
	5.97×10^{5}	0.487		Rabbit (intragastric)	Pai et al. (1986), Hass et al. (2000)
	1.9×10^{5}	0.22		Human (Shigella and EPEC data)	Powell et al. (2000); Teunis et al. (2004)
	2.6×10^{5}	0.06		Human (est.)	Strachan et al. (2005)
Salmonellae Enteritidis			0.1	Human (based on ID50 from outbreak data of *S.Enteritidis* and *Typhimurium*)	Teunis et al., 2010
Listeria monocytogenes	2.1×10^{6}	0.17			Haas et al. (1999)
			1.15E-05	Mice (oral)	Golnazarian et al. (1989)
	1.78×10^{9}	0.0422			Smith et al. (2007); Williams et al. (2007);
Campylobacter jejuni	795	0.15			Teunis et al., (1999)
	890	0.144		Human (oral)	Black et al. (1988)
Cryptosporidium parvum			0.00419	Human	Haas et al. (1996); DuPont et al. (1995)
			5.72E-02	Human	Messner et al. (2001)
Rotavirus	6.1	0.253		Human	Teunis and Havelaar (2000), Ward et al. (1986)
Adenovirus			0.4172	Human	Lim et al. (2015)

Table A2.3: Additional pesticides considered for risk to fermentation and brewing potential of barley. MRL – Maximum residue level.

Pesticide	$LogK_{ow}$ (pH 7, 20°C)	BCF (soil-grain), est. [2]	MRL barley (mg/kg) [3,4,5]	Carryover to young beer (%) [1], est.	Alcohol reduction (%)	Soil degradation (DT_{50} field, d)
Azaconazole	2.36	0.473	1.5	51	ND	ND
Azoxystrobin	2.50	0.330	0.3	48	ND	80.2
Bitertanol	4.10	0.002	0.05	18	ND	23.0
Cyproconazole	3.09	0.064	0.02	37	0.00375	36.0
Cyprodinil	4.00	0.003	3	20	ND	45.0
Difenoconazole	4.20	0.001	0.02	16	0.0093	85.0
Dimethomorph	2.68	0.205	0.05	45	ND	44.0
Dodemorph	4.60	0.000	0.02	8	ND	41.0
Epoxiconazole	3.30	0.033	1	33	0.00833	120
Etaconazole	3.10	0.062	0.02	37	ND	ND
Fenbuconazole	3.79	0.006	0.02	24	ND	61.0
Fenhexamid	3.51	0.017	0.05	29	ND	25.0
Fenpropimorph	4.50	0.000	0.5	10	ND	25.5
Flusilazole	3.87	0.005	0.02	22	ND	94.0
Flutolanil	3.70	0.009	0.05	25	0.00125	234
Imazalil	2.56	0.282	0.02	47	ND	6.40
Myclobutanil	2.89	0.114	0.02	41	0.005	35.0
Oxadixyl	0.65	13.606	0.02	84	ND	75.0
Propiconazole	3.72	0.008	0.2	25	0.041	214
Pyrifenox	3.40	0.024	0.02	31	ND	66.0
Tebuconazole	3.70	0.009	2	25	0.00429	55.8
Thiabendazole	2.39	0.438	0.05	50	ND	724
Thiophanate-methyl	1.45	3.801	0.3	68	ND	5.00
Triadimenol	3.18	0.048	0.2	35	ND	64.9

[1] Miyake et al. (1999); [2] Trapp & Legind (2011); [3] WRAP (2013), [4] http://www.codexalimentarius.net/pestres/data/pesticides/index.html

11 Appendix III: Existing standards for effluent reuse for agricultural irrigation

Overview of existing standards for effluent reuse for agricultural irrigation. From Becerra-Castro et al. (2015).

Country or organism (year)	Irrigation categories	pH	EC (µS/cm)	SAR	NTU	SS (mg/L)	BOD (mg/L)	COD (mg/L)	DO (mg/L)	TNK or TN (mg/L)	$N-NO_3$ (mg/L)	P (mg/L)	Sulphate (mg/L)	Total coliform (CFU/100 mL)	Faecal coliform (CFU/100 mL)	*Escherichia coli* (CFU/100 mL)	Nematode eggs (no./L)
US-EPA (2012) [1]	UR	6–9			2	30	10								Absent		
	R	6–9			–	–	10								2×10^2		
WHO (2006a) [2]	UR															10^3	≤1
	R															–	≤1
California (1978) [1]	ND	6–9			2				Present						2.2×10^2		
Italy (2003) [3]	ND	6–9.5	3000	10		10	20	100		15		2	500			10^2	
France (2010) [4]	UR					15		60							4°	2.5×10^2	
	R					–		–							$2–3^{\circ}$	$10^4–10^5$	
Spain (2007) [5]	UR				10	20										10^2	0.1
	R				–	35										$10^3–10^4$	0.1
Portugal (2006) [6]	UR	6.5–8.4	1000	8		60					50		575		10^2		
	R	6.5–8.4	1000	8		60					50		575		$2 \times 10^2–10^4$		
Australia (2000) [7]	UR	6.5–8.5			2									10			
	R	6.5–8.5			–									$10^2–10^4$		$10^2–10^4$	
Israel (1999) [8]	ND				5	10	20								10		
Tunisia (1989) [9]	ND	6.5–8.5	7000			30	30	90									≤1
Jordan (2002) [9]	UR	6–9		9	10	50	30	100	>2	45	30	30	500			10^{2*}	<1
	R	6–9		9	–	150	200–300	500	–	70	45	30	500			10^3–n.d.*	<1
Kuwait (2001) [9]	ND	6.5–8.5				15	200	100	≥2.0	35		30	0.1	$4 \times 10^{2*}$	20^{*}		<1
Oman (1993) [9]	UR	6–9	2000	10		15	15	150			50	30	400		2×10^2		<1
	R	6–9	2700	10		30	20	200			50	30	400		10^3		<1
Saudi Arabia (2000) [9]	UR	6.0–8.5			5	10	10				10		600	2.2^{*}			1
	R	–			–	40	40				–		–	10^{3*}			1
China (2007) [10]	UR	5.5–8.5				60	40		≥0.5						2×10^4		
	R	5.5–8.5				80–100	60–100		≥0.5						4×10^4		
Mexico (1987) [11]	UR					20	20								240^{*}		
	R					30	30								10^{3*}		

UR, unrestricted irrigation; R, restricted irrigation; EC, electrical conductivity; SAR, sodium absorption rate; NTU, Nephelometric Turbidity Unit; TSS, total suspended solids; BOD, biological oxygen demand; COD, chemical oxygen demand; DO, dissolved oxygen; TNK, total nitrogen Kjeldahl; TN, total nitrogen; $N–NO_3$, nitrate nitrogen; P, phosphate; °log reduction value; *MPN/100 mL; n.d. no determination.
References: 1. EPA (2012); 2. WHO (2006a); 3. Decreto Ministeriale (15/2003); 4. NOR-SASP1013629A (2010); 5. Real Decreto (1620/2007); 6. Marecos do Monte (2007); 7. NWQMS (2000); 8. Arlosoroff (2007); 9. WHO (2006b); 10. Yi et al. (2011); 11. NOM-001-ECOL-1996.

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