A Methodology to Derive Groundwater Clean-up Standards

Volume 1: Main Report

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Volume 1: Main Report

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EXECUTIVE SUMMARY

There are no nationally agreed standards or methodologies to determine the most appropriate extent of groundwater clean-up following pollution incidents or pollution from contaminated land. Targets used in other countries and UK drinking water standards are often referred to, but these are commonly not appropriate and this has led to an *ad-hoc* approach and complaints from site owners liable for sometimes unrealistic clean-up costs. This, together with the introduction of new provisions regarding contaminated land in Section 78 (Part IIA) Environmental Protection Act 1990 as set out in Section 57 of the Environment Act 1995, has created an urgent need for a standardised, practical and reasonable approach to clean-up to be developed which is capable of being applied on a site-by-site basis.

This document develops a common framework for a systematic methodology which will enable Agency staff to achieve consistency when deciding the quality to which contaminated groundwater should be cleaned up. One of the main considerations in the development of the methodology for setting clean-up standards was the need for it to be pragmatic, flexible and, most importantly, to set achievable standards. The risk to a body of groundwater from a pollution incident will obviously vary from site to site, and the merits and feasibility of groundwater clean-up in each case should depend on local circumstances and available funding. The approach that has been taken is based on risk, and effectively sets out to prioritise aquifers for protection and remediation depending on their importance, existing water quality and possible future utilisation.

The focus of the methodology developed for setting groundwater clean-up standards is a quantitative approach based on a risk assessment. The first part of this approach is to prioritise a site in terms of its threat to an identified receptor. This prioritisation is intended to identify those sites that require urgent action to prevent impact of the pollution on the receptor and those sites where the size of the pollution threat to a source or resource makes remediation imperative. The second part sets a quantitative clean-up standard at a compliance borehole located in the pollution plume such that the standard concentration will permit exceedance of agreed water quality standards at a receptor; a borehole, spring or surface water body. The setting of the clean-up standard will be undertaken by mathematical modelling with three stages of complexity:

- a relatively simple analytical approach;
- a refinement of the analytical approach;
- a comprehensive numerical model.

The quantitative approach to the setting of the groundwater clean-up standards will be based on an iterative series of site investigations to provide a detailed hydrogeological data bank of the site and surrounding area. The modelling also is supported by:

1. comparison of methodologies used in other countries;
2. a review of water quality standards for use at all receptors;

1. INTRODUCTION

Groundwaters have a substantial strategic significance in public water supply; they supply approximately thirty-five percent of current demand and in some areas they represent the only available future resource. Groundwater also represents an important source of water for industry, agriculture and private abstractors and it is therefore important that the volume and quality of groundwater is protected against adverse and irreversible changes in the hydrogeological system resulting from man's activities in industry and agriculture. Wherever groundwater is present there is the potential for it to be affected by pollution, since no soil or rock is completely impermeable and no pollutant completely immobile. It is essential that groundwater resources are protected.

The Environment Agency has a statutory duty under the Water Resources Act 1991 to monitor and improve the water environment. Specifically it has a duty to protect the quality of groundwater and take remedial action, when pollution has occurred, to preserve its use as a water resource. The Environment Agency seeks to encourage the implementation of effective remedial measures to prevent pollution of groundwater by existing direct or indirect discharges from contaminated sites. The Environment Agency would always hope to co-operate with landowners to encourage investigation and prevent continuing pollution; if however such co-operation is not forthcoming, the Environment Agency will, if implemented, be able to use its powers of prosecution under Section 85 and Section 161 of the Water Resources Act 1991 as amended by the Environment Act 1995.

Although groundwater quality is generally very good, there are many instances where this is not the case and where water quality has been degraded to a point where it is no longer suitable for potable use without treatment. In some cases groundwater may become grossly polluted and this can present a severe threat to the resource value of the aquifer.

Pollution may occur in both the urban and rural environment and can commonly be attributed to two main factors:

- diffuse pollution caused by widespread activities e.g. agriculture, that commonly impact on wide areas of an aquifer;
- point source or single plume pollution - attributable to discrete activities, although where there are multiple point sources a more diffuse plume may be encountered.

It takes a combination of contamination source, the migration of that contaminant to water and the sensitivity of the receiving water to create a measurable impact from a site on ground and surface waters. A contaminated site can pose a risk to the water environment in a number of ways, however, a source-pathway-target (receptor) scenario must be established before there is a need to take steps to protect groundwater.

Despite the possible requirements for clean-up, there are no nationally agreed standards or methodologies to determine the most appropriate extent of remedial work. Target concentrations used in other countries and UK drinking water standards are often referred to, but these are commonly not appropriate and this has led to an *ad-hoc* approach and complaints from site owners liable for sometimes unrealistic clean-up costs. This, together with the introduction of new provisions regarding contaminated land in Section 78 (Part IIA) Environmental Protection Act 1990 as set out in Section 57 of the Environment Act 1995, and Section 22 of the Environment Act 1985 which adds S161a to S161 of the Water Resources Act 1991 and deals with pollution of controlled waters, has created an urgent need for a standardised, practical and reasonable approach to clean-up to be developed which is capable of being applied on a site-by-site basis.

The Water Act 1989, consolidated in the Water Resources Act 1991, makes provision for a system of classification for controlled waters, and through this system, for setting Water Quality Objectives (WQOs) for specific water bodies. So far WQOs have only been introduced in relation to long-standing EC Directives. The initial WQO scheme concentrated mainly on rivers while systems for other controlled waters would have needed to be developed. Any system for groundwater would need to be based on individual chemical parameters for each aquifer and would, by necessity, be very involved. The scheme is perceived as being highly complex and discussions are likely to continue on the way forward on this issue.

This document develops a common framework for a systematic methodology which will enable Environment Agency staff to achieve consistency when deciding the quality to which contaminated groundwater should be cleaned up. The benefits to be gained from such a system include:

- the Environment Agency will achieve a better level of consistency in the application of clean-up standards for polluted groundwater, and should avoid the majority of complaints received from site owners and developers;

- non-specialist Environment Agency staff will possess a clear methodology to use in cases of clean-up of polluted groundwater, which will improve operational effectiveness.

The physical and chemical processes by which contaminants leach from any given site into the groundwater and the setting of appropriate standards for clean-up is the subject of a concurrent Environment Agency research project (Project Number i714: Guidance on degree of soils clean-up needed to protect water resources). The remit of this project, therefore, effectively begins at the boundary of the site, or at a given distance from the site when contaminants have been identified in the groundwater at a specific concentration. A schematic diagram illustrating the agreed divide between these two projects is given in Figure 1.1.

This report sets out a framework for assessing the potential impact of polluted groundwater on various receptors and the groundwater resource, on the basis of an analysis of risks presented by individual sites. The report outlines the threats which contaminated water may pose to water abstraction users, human health or the environment in general. The scope and methodology for tackling the problem of groundwater clean-up are addressed together with the technical and practical issues that must be evaluated to establish targets for clean-up. The proposed methodology could be influenced by future legislative developments. However in the mean-time it provides the Environment Agency with a system for protecting water resources in the UK from further derogation.

2. BACKGROUND

2.1 General

The key criteria in developing a systematic methodology for setting clean-up standards for groundwater was that it should provide a reasonable and appropriate evaluation of the extent to which groundwater contamination should be cleaned-up, or contained, given the specific circumstances of each site. The methodology proposed in the following section has been designed in such a way that it can be used in widely differing circumstances, without being a prescriptive table of numbers. The methodology takes into account initial work historically undertaken by the NRA and the possibility of naturally occurring contamination and areas of saline intrusion.

To ensure that current best practice was adopted in the new methodology a number of activities was undertaken by WRc prior to developing the proposed approach:

- a thorough review of the advantages and disadvantages of existing clean-up standards and methodologies used in other EU countries and the US (Section 2.2, Appendix A);
- a summary of available groundwater clean-up techniques noting their dependency, limitations and long term cost effectiveness (Appendix D);
- a consideration of the technical issues and implications of the new Environment Act 1995 and current EC Directives relating to clean-up standards with particular reference to cases where site owners are required to clean-up a site (Section 2.3).

The strategy for site specific remediation and standard setting has taken into consideration the risks posed by differing levels and types of contaminants to the environment and human health and the benefits that a given degree of clean-up would provide, whilst at the same time realising that feasibility and cost of clean-up should play an important part in the decision making process.

The dilute and dispersed nature of diffuse sources of pollution may mean that it is virtually impossible to clean-up the groundwater, regardless of cost, even when the source of a problem is removed. As you move away from a source and pollution inevitably becomes more dilute and dispersed, clean-up becomes less easy to justify in terms of the cost of treatment and any benefit of small improvements in water quality. To protect the quality of groundwater it is therefore important to identify pollution incidents as they arise, and prevent widespread dispersion of the pollutants in the aquifer, although clean-up may also be necessary for 'residual' groundwater contamination, resulting from past activities and incidents.

It is obviously important that any decisions relating to the clean-up of contaminated groundwater should not adversely impact on the baseline quality of aquifers, but equally, the baseline quality of naturally contaminated (e.g. saline intrusion) waters should not be ignored. The objective of any clean-up operation will ultimately be a compromise between reaching a level of clean-up that will satisfy the Regulator and the cost of achieving that clean-up by the industrial party. For example an aquifer may not be currently used for abstraction, however, this may change in the future. The standards that are applied to such a site may be more relaxed than those that would be applied at a borehole currently used as a drinking water source, and effectively represent a mid-way point between that and letting the aquifer degrade to a point where it cannot be used in our lifetime as a resource.

One of the main considerations in the development of the methodology for setting clean-up standards was the need for it to be pragmatic, flexible and, most importantly, to set achievable standards. The risk to a body of groundwater from a pollution incident will obviously vary from site to site, and the merits and feasibility of groundwater clean-up in each case should depend on local circumstances and available funding. The approach that has been taken is based on risk, and effectively sets out to prioritise aquifers for protection and remediation depending on their importance, existing water quality and possible future utilisation.

The framework methodology focuses on cleaning-up groundwater that is already contaminated rather than directly addressing the issues of contaminated land *per se*. This project specifically addresses the problems caused either by spillage of chemicals or by leakage from a site over a period of time.

The methodology seeks:

(a) to determine, by risk assessment, the necessity of remediating a given body of contaminated groundwater;

(b) when remediation is shown to be necessary, to determine the priority of the remediation;

(c) to establish a framework for setting the standards to which any remedial work is carried out.

2.2 Existing clean-up standards and methodologies

Two Specific Objectives in the Terms of Reference from the Environment Agency were:

- to review thoroughly the existing clean-up standards and methodologies used by other countries, particularly the USA, Netherlands, and Germany, and summarise their advantages and disadvantages;

- to take into account the views of interested groups such as the water companies, industry, the Department of the Environment, and lobby groups.

In order to fulfil these objectives, details of the standards and methodologies in other countries have been obtained from the following organisations:

- **USA**. State of Colorado, Department of Labor and Employment, Oil Inspection Unit;

- **Denmark**. Ministry of Environment and Energy;

- **France.** Bureau de Recherches Geologiques et Minieres (BRGM);

- **Netherlands.** National Institute of Public Health and Environmental Protection. (RIVM);
- **Italy.** Provincial Government of Milan;
- **Germany.** Länderarbeitsgemeinschaft Wasser;

A review of these standards and methodologies for groundwater is given in Appendix A of this Report.

In order to assess the situation with regard to standards and methodologies in the United Kingdom, views of members of staff active in the field of contaminated land evaluation in England and Wales have been sought from each of the Environment Agency Regions, the DoE Contaminated Land and Liabilities Division, and from hydrogeologists in four Water Utilities with major groundwater interests. In addition views have been sought from four well known consulting firms involved in contaminated land work and from the British Geological Survey. The views of these organisations also are summarised in Appendix A.

The information on the groundwater clean-up standards and methodologies abroad suggest that there is a widespread uniformity of approach.

The methodology for setting clean-up standards for general protection of groundwater is seen as an integral part of setting standards for clean-up of contaminated soils. This can be seen in the standards set in New Jersey, New York and Pennsylvania where there are standards for soil clean-up specifically aimed at 'groundwater protection' but no standards specifically for polluted groundwater. Similarly in the Netherlands (VROM, 1994), the setting of soil target values treats soil and groundwater as one environmental compartment. This approach may be perfectly adequate in areas with shallow water tables but in areas with deep groundwater, as in parts of the UK, it may be more appropriate, as in this project, to deal with soil and groundwaters as two separate parts of one continuum.

The basis of setting soil and groundwater standards is now most commonly a risk assessment of the potential impact of pollution on a vulnerable target. The basis of the groundwater standards tends to be the natural background qualities or statutory quality standards, for example, toxicologically based drinking water standards. The draft methodology from Milan, Italy, (Appendix A) is a good example of this. The Dutch and Danes publish clean-up target standards for groundwater for selected compounds but there is an increasing consensus that standards should be site-specific based on risk assessment taking into account the size of the contamination incident, vulnerability of aquifers and ultimate targets (boreholes or surface waters), and the end-use of the water. The Dutch standards do allow the flexibility to include this risk-based approach.

The use of a risk-based methodology will tend to require more intensive site investigations on which to base the risk assessments than would be required if purely tabular numerical standards were set. The cost of site investigations has been cited as a criticism of the US approach. It could be argued, however, that a detailed site investigation would be needed in any case to define, control, and monitor a clean-up programme.

The responses from the agencies in the UK demonstrate two features relating to groundwater clean-up:

- there are no national standards for groundwater clean-up so the standards applied at any site depend on negotiations between site owner and regulator. There is no consistency. In the absence of UK standards the Dutch numerical standards are commonly used;

- most clean-up operations in UK are subject to 'commercial confidentiality' agreements so the clean-up standards imposed are often not widely publicised in other Environment Agency regions or available to other site owners with similar problems.

This situation is recognised to be unsatisfactory and the present project (with the complementary one on soil clean-up) is widely welcomed.

2.3 Implications of the new Environment Act

The Environment Act 1995 supplements the existing Water Resources Act 1991 and the Environmental Protection Act 1990 and addresses a number of issues that were not covered in the original Act. The provisions rely on a number of subsidiary regulations and guidance to give them effect. Many of its provisions have not come into force and will be brought in stages. Of particular interest are:

- the establishment of the new Environment Agency in England and Wales (Environment Agency) (which came into force on 1 April 1996) and the Scottish Environmental Protection Agency in Scotland;

- the introduction of new rules governing the identification and remediation of contaminated sites.

The new rules relating to contaminated land form a new Part IIA of the Environmental Protection Act 1990. The definition of several terms used in this new Part IIA will be subject to guidance to be issued by the Secretary of State for the Environment to the Local Authorities following consultation with the Agency.

In the UK, traditional practices associated with redeveloping contaminated sites have focused on the protection of the ultimate end user of the site. The wider issues relating to the environmental impact of contamination remaining in the ground were not widely explored. The influential Interdepartmental Committee on the Redevelopment of Contaminated Land (ICRCL) has published a number of guidance documents relating to the redevelopment of contaminated sites which focuses on identified end uses. The most comprehensive of these documents was Guidance Note 59/83 (2nd Edition 1987) - Guidance on the Assessment and Redevelopment of Contaminated Land. This document failed to address the possible impact of contaminated land on water resources. The ICRCL Guidelines included 'target levels' of contaminants in soils based on the hazard posed by soils according to a given end use. The levels were intended to give guidance only, however over time and without recourse to other advice, the lists have tended to be used on a more prescriptive basis. The ICRCL guidance on acceptable contaminant levels did not extend to safeguarding the aquatic environment.

The Environmental Protection Act 1990, did propose, in Clause 143, setting up a register of contaminated land, but this idea was abandoned following pressure from the business sector and landowners who claimed the measures would lead to 'land blight'. Importantly the Act also ignored the water cycle in relation to contaminated land.

The Environment Act 1995 now recognises the importance of both surface water and groundwater in the assessment of the impact of contaminated land.

In preparation for the Environment Act, the Department of the Environment issued, "[Draft] Guidance on determination of whether land is contaminated land under the provisions of [Part IIA of the Environmental Protection Act 1990]". The Draft Guidance was updated as a 'Final Working Draft' in June 1996. This document, when issued in its final form, will provide the basis of the guidance to be expected from the Secretary of State in implementing the Act.

The impact of contaminated land is defined specifically in terms of a continuum of :

SOURCE - PATHWAY- TARGET

Where: The source is the contaminated soil.
The pathway is the route whereby pollution may move from the source to the target.
The target is any person, organism or structure at risk under the conditions defined by the Act[1]

In Figure 1.1 the **source** is contaminated land at shallow depth beneath an industrial area . The **pathway** is the groundwater transport system. Recharge through the contaminated soils will percolate downwards to the water table or to perched water tables. This movement will be controlled by the physics of groundwater movement, the physico/chemical nature of the soils and polluting chemicals, and the biological activity within the soils. The water table may be considered a primary target of the pollution pathway but it is not a **target** as used in the Guidance Documents. At certain contaminated sites, intense pollution may lead to the infiltration of a chemical as free phase, particularly if the chemical has a low solubility in water. The most common free phase pollution is by hydrocarbons which, being lighter than water, will float on the water table creating what may be considered a secondary **source** of pollution. The pollution arriving at the water table will dissolve in the groundwater and then move with the

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[1] The point at which a groundwater pollution incident impacts on the surface either at a spring, through a borehole or as baseflow to a river or lake is specifically defined as a "Target" in the first DoE Guidelines (Section 2.3). The generic term "Receptor" is more commonly used in the descriptive text of the final working draft of the DoE Guidelines although the specific terms source-pathway-target are still referenced. In this Report the example of the final working draft is followed and receptor is subsequently used throughout instead of target in the general text. Groundwater itself can also be considered a receptor and in a situation where a receptor cannot readily be identified then a surrogate receptor (borehole) may be installed to define the background quality of a water resource (Figure 3.1).

water body to the most vulnerable **targets** of humans via public supply boreholes (as in Figure 1.1) or surface water ecologies via springs and baseflow to rivers.

The environment is now considered as a whole and each sequence of **source-pathway-target** from a site must be considered under the procedures given in the Draft Guidelines.

The period between the Environmental Protection and Environment Acts has seen a burst of activity in the guidance offered by various bodies interested in the study of contaminated land. The series of Contaminated Land Research Reports from the Department of the Environment (DoE 1994) will be an important source of advice. This series has a serious gap with the lack of advice on the technology of site investigation but this gap is plugged by the admirable British Standards Institute (BSI) Code of Practice for Site Investigations; BS 5930:1981(Draft Revision (95/108483) of 7 December 1995) and their Draft for Development: Code of Practice for the Identification of Potentially Contaminated Land and its Investigation; DD175:1988.

The Environment Agency, formerly the NRA, is the main organisation in charge of the protection of surface and groundwater quality and they have issued two important documents in relation to contaminated land; Policy and Practice for the Protection of Groundwater (1992) and Contaminated Land and the Water Environment (1994A).

The Draft Guidance Documents from the DoE stress throughout the need to be pragmatic; to make sure redevelopment is needed, practicable and cost effective:

> "This approach requires remedial action only where;
>
> - the contamination poses unacceptable actual or potential risks to health or the environment; and
> - there are appropriate and cost effective means available to do so taking into account the actual or intended use of the site."

This policy set out in the DoE and Welsh Office document," Framework for Contaminated Land", means that there is a clear need for site investigations in sufficient detail to demonstrate the need (or not) for remedial action. The judgement of whether there is a 'need' for remedial work action rests on the interpretation of several general, non-specific terms such as 'significant'.

Contaminated land is defined by the Environmental Protection Act 1990 (EPA) (Clause 78A) as:

> "Any land which appears to the Local Authority in whose area it is situated to be in such a condition, by reason of substances in, on or under the land, that
>
> (a) significant harm is being caused or there is a significant possibility of such harm being caused; or
>
> (b) pollution of controlled water is being, or is likely to be, caused."

The Environment Act has also amended a provision dealing with anti-pollution works and operation in Section 161 of the Water Resources Act 1991, for which the Environment Agency is now responsible. At present there is no power for the Agency to require work to be carried out by the polluter or the owner of contaminated land; if it cannot persuade someone to carry out the work voluntarily all it can do is carry out the work itself and then claim back its costs from the person or persons responsible. When the new provisions come into force it will be possible for the Agency to serve a 'works notice' on any person who caused or knowingly permitted pollution of controlled waters or any person undertaking an activity likely to cause pollution to enter controlled waters or be present in controlled waters. The Agency will have a very wide discretion as to whom it serves a notice on, albeit within general policy guidelines set out by the Government

Controlled waters (of relevance to contaminated land) are defined in the Water Resources Act 1991 Part III, Clause 104 as -

> "(c) inland waters, that is to say, the waters of any relevant lake or pond or of so much of any relevant river or water course as is above the fresh water limit.
>
> (d) ground waters, that is to say, any waters contained in underground strata"

Harm is loosely defined as:

> "harm to the health of living organisms or other interference with the ecological systems of which they form part and in the case of man, includes harm to his property."

The definition of 'significant' and 'possibly significant' harm is of key importance, since the Guidance states that harm that does not fall into these categories should be disregarded. The definition of 'significant' harm shall be determined in accordance with guidance issued for the purpose by the Secretary of State (78A(5)) and under Clause 78A(6), there may be a provision for different degrees of importance to be assigned to, or for the disregard of, different descriptions of living organisms, ecological systems, places or harm to health or property. This guidance is given as a series of risk matrices with descriptions of harm and supporting rankings of possibilities of harm occurring.

To establish whether any land appears to be contaminated land, the Local Authority must confirm the presence of substances with the potential to cause relevant harm. The DoE series of Contaminated Land Research Reports give specific guidance on how the necessary site investigation should be undertaken.

The final stage of such a site investigation is a risk assessment. Broad guidance is given on how to set about this including, "carry out an estimation of risk, for as many plausible source-pathway-target (receptor) relationships as appropriate for the different descriptions of harm which may be found in, on or under that land." Factors to be taken into account include:

- how the substances found may give rise to harm;
- the factors affecting the probability of harm occurring;

- the statistical probability of harm occurring;
- the timescale within which harm may occur;
- the vulnerability of the targets;
- the level of protection already afforded.

The Draft Guidelines deal separately with water pollution from contaminated sites. It is for the local authorities to decide whether any of the above are applicable, subject to guidance to be given by the Government. The local authority will also normally be the enforcing body although the new Environment Agency will have some supervisory powers in cases where sites pose a risk of water pollution as well as taking a direct interest in the most seriously contaminated sites 'special sites'. Distinct from studies of the sources, the determination of the likelihood of controlled waters being polluted should take into account multiple factors including:

- whether substances in, on or under the land are polluting substances;
- the mobility of the substances;
- the site conditions;
- the hydrogeology;
- proximity to surface waters;
- proximity to groundwaters;
- the sensitivity of surface waters and groundwaters to pollution;
- the effectiveness of anti-pollution measures.

This list of factors by the DoE clearly demonstrates that all controlled waters should not be considered inviolate just because they are defined as 'controlled' by the Water Resources Act 1991. The Environment Bill states that the Local Authority should contact the Environment Agency when it has designated that a contaminated site poses a risk of water pollution to obtain advice on any remedial actions.

The implications of the new Environment Act as set out in the DoE Framework and Draft Guidelines are quite significant and have been taken fully into account when developing the methodology described in this report. The remediation methodologies, including the setting of clean-up standards, should be:

- based on a risk assessment in terms of the **SOURCE-PATHWAY-TARGET** scenario;
- pragmatic, practicable and cost effective.

Site investigations will need to be undertaken in sufficient detail to provide the necessary data for the risk assessment, both in terms of soils and groundwater transport within the aquifer system.

3. FRAMEWORK

3.1 Introduction

A simplified diagram of a situation commonly met when contaminated land is impacting on the quality of a groundwater resource and, potentially, of a groundwater source is shown in Figure 3.1.

The contamination, either by leaching from a contaminated site or by infiltration from a point source leak or spill, has reached the water table. At the water table the contaminants have dissolved in the groundwater body and have moved some way down the groundwater gradient as a discrete pollution plume. The pollution plume may be relatively small as in Figure 3.1 or, in the case of an old pollution incident, extensive and possibly reaching a vulnerable receptor.

Organic pollutants commonly are not miscible with water and free product may remain on the water table. Pollutants lighter than water, for example hydrocarbons such as lubricants or fuels, will float on the water table whilst pollutants denser than water, for example, chlorinated solvents, will tend to sink to the base of the saturated aquifer. These immiscible liquids (NAPLs; Non-aqueous phase liquids) will then act as secondary sources of pollution.

In a case where a groundwater clean-up standard is being considered, it may be assumed that the presence of groundwater pollution has been proven. This will have been shown by an initial site investigation of a contaminated site or the appearance of polluted groundwater at a receptor; a borehole, spring or surface water course.

The action to be taken at each contaminated site will be decided by a prioritisation programme following an overall decision-making process discussed in detail in Section 4 (Figures 4.2 - 4.7). This prioritisation will ensure that the clean-up programme chosen will be focused, pragmatic, practicable and cost effective.

The methodology associated with setting clean-up standards at a site follows a logical iterative procedure beginning with the work programme needed to fully understand the hydrogeological setting of the pollution incident and ending with the definition of the optimum remedial programme. The individual stages of this programme are shown in Figure 3.2 and are summarised in the latter part of this Section.

The elements of the proposed methodology for setting groundwater clean-up standards are shown in Figure 3.1. They basically comprise a compliance borehole, a sentinel borehole and background borehole (surrogate borehole) or receptor. The locations of these elements presuppose and will be based on, adequate site investigations of the pollution incident.

The **compliance borehole** may be located on the down-gradient edge of the contaminated site or at some point down-gradient of the site boundary. It has to be located within the pollution plume as, by definition, its function is to provide groundwater samples for analysis to demonstrate a reduction in pollution levels and, ultimately, the achievement of agreed clean-up standards. It is recommended that the compliance borehole should not be located at the edge

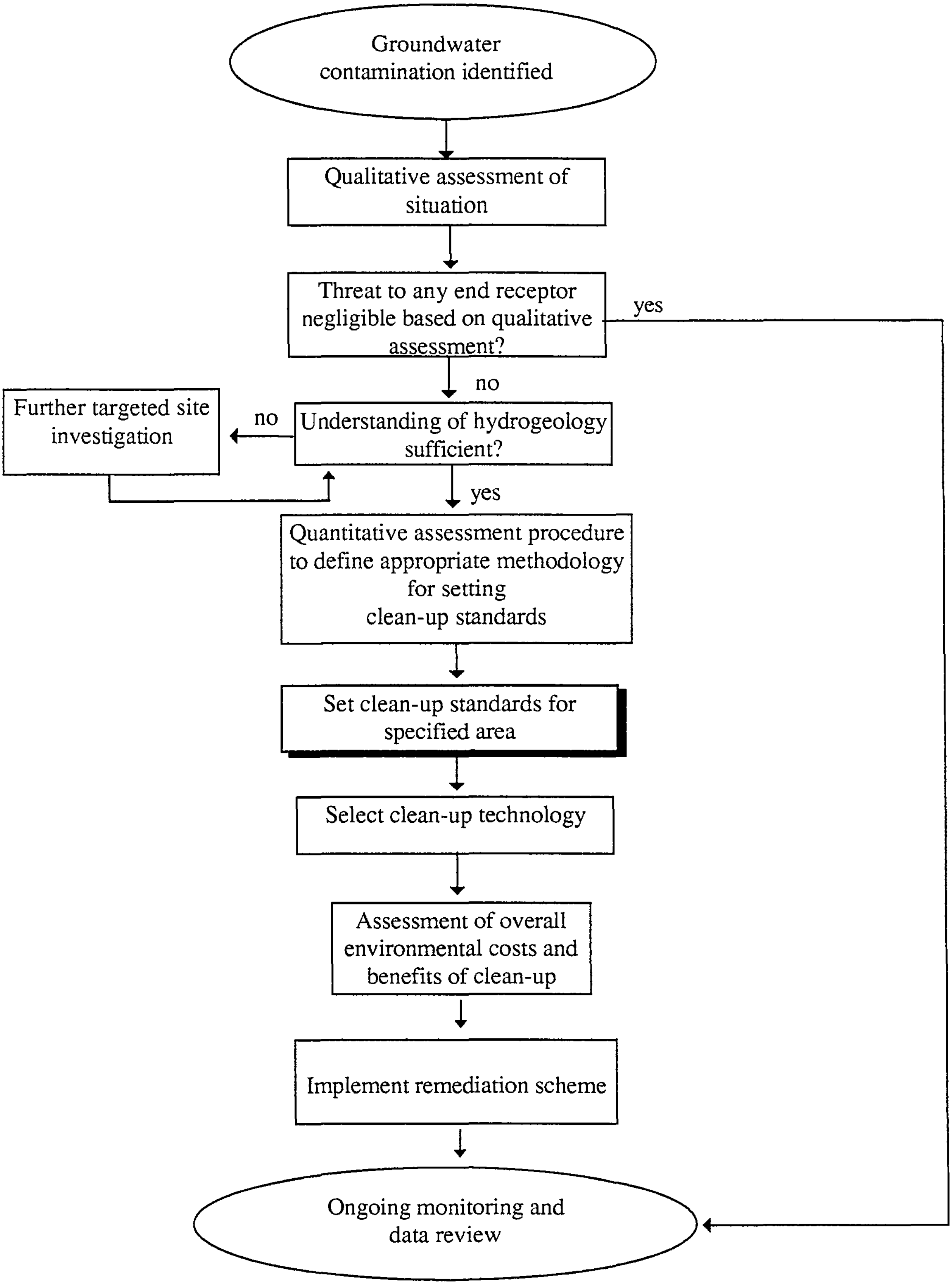

Figure 3.2 **Proposed methodology for setting and implementing groundwater clean-up standards**

of the pollution source, because remediation activities there could induce complex responses in the groundwater and make it very difficult to demonstrate whether or not standards have been achieved. The location of the compliance borehole (and therefore the point of application of any clean-up standards) should be as close to the edge of the site and source of pollution as is practicable, since the greater the distance from the site at which the clean-up standards are set the greater the potential for wider derogation of the aquifer. Whilst taking into account the non-uniformity of any pollution sources in the majority of studies a borehole located a short distance down-gradient of the source of pollution would be preferred, where dispersion processes will have resulted in a greater degree of mixing across the width of flow than if the borehole were located on the edge of the site. Where extensive site investigations have been undertaken to evaluate the geometry and extent of the pollution plume, it may be possible with forethought to use one of these boreholes as the compliance monitoring point. Where a given pollution plume is particularly variable or the width of the plume perpendicular to the direction of groundwater flow is large, it may be appropriate to use more than one borehole to monitor the effectiveness of any remedial measures. The different variables of each individual case make it inadvisable to fix any hard and fast rate or model output to fix the position of the compliance point, but it is thought preferable that this location is fixed within a 200 m radius of the site, perpendicular to the direction of groundwater flow. For monitoring purposes the borehole should be installed in such a way that it fully penetrates the active aquifer.

The **receptor** is the point down-gradient of the pollution which is the actual or surrogate point deemed to be at risk from the pollution. A groundwater quality at this point will be agreed and then will be used as the basis on which the clean-up standard at the compliance borehole will be set. The agreed quality may be dependent on the end-use of the water or other environmental considerations. The quality of water agreed for a surface water threatened by polluted baseflow may be in terms of Environmental Quality Standards. The agreed quality of a public supply borehole or other boreholes and springs that may be used for potable supply will be, at a minimum, the statutory Drinking Water standards. In some circumstances and for certain pollutants, the agreed quality may be the background quality in an aquifer. This would be the case if the background groundwater quality is worse than the Drinking Water standards.

There will be situations where an aquifer is at risk but there are no obvious receptors at risk close to the pollution incident. There may be no point receptor but the groundwater resource itself is at risk. In such a case a surrogate (or background) borehole should be located approximately 1 km or 400 days travel time, whichever is the least, down-gradient of the pollution plume to act as a surrogate receptor and provide samples to define the background groundwater quality from which the clean-up standard can be calculated. In most, if not all, cases the surrogate borehole would be a theoretical point at which any calculations for clean-up could be based. However, there may be a need for a borehole to be installed if background water quality in the aquifer is insufficiently characterised. The distance chosen for the receptor is somewhat subjective but is consistent with the definition of the radius of Protection Zone II around public supply boreholes advocated by the Agency. It is considered undesirable to increase this distance as that would permit pollution of the groundwater resource over unacceptable areas.

The groundwater clean-up standard is set at the compliance borehole and is defined by first calculating or mathematically modelling the groundwater pollution flow and attenuation between the compliance borehole and the surrogate borehole or receptor(s). The pollution attenuation is then back-calculated from the surrogate borehole or receptor using the agreed quality at that point to establish the clean-up standard at the compliance borehole such that the water quality at the receptor or surrogate borehole will not exceed the agreed concentration. At sites where there are several receptors a clean-up standard will be established for each receptor and then the more rigorous of the standards adopted for the site. In areas of past heavy industrial development contaminated sites may be very extensive and complex and in such a situation detailed mathematical modelling of the site should be specified and several compliance boreholes defined in order to monitor a clean-up programme adequately.

In situations where the pollution plume has not reached the receptor, it is recommended that a **sentinel borehole** should be drilled down-gradient of the pollution plume. A clean-up standard may be set for this site using the same back-calculation as above. The sentinel borehole then acts as a warning of the approach of pollution towards the receptor and, should pollution reach it, could act as a calibration point to improve the modelling of the pollution transport.

It is possible that, when remediation begins, the intensity of the pollution plume decreases and the clean-up standard is achieved but the tongue of the plume beyond the compliance borehole may continue flowing towards the receptor. Such a plume may not be remediable except by natural attenuation processes and in such a case the sentinel borehole will be invaluable in warning of the pollution's arrival and its magnitude.

The compliance borehole is the point at which the groundwater clean-up standards are set. It is also the effective end point of the complementary Environment Agency Project on setting soil clean-up standards. It is recognised, therefore, by the Environment Agency that the soil clean-up standards set for a site by the methodology of one project must result in compliance at the compliance borehole with the groundwater clean-up targets set by following the methodology in this Project.

3.2 Programme for setting and implementing groundwater clean-up standards

The different units within the programme are discussed briefly below but are dealt with in more detail in the following Sections and supporting Appendices (Volume 2.).

3.2.1 Preliminary site investigation

There is a number of possible scenarios from which this whole assessment procedure could arise, for example:

1. a geotechnical site investigation during which groundwater contamination is provisionally identified;
2. a contaminated site investigation which identifies that groundwater contamination may have occurred;

3. the detection of contamination in groundwater off site, for example in a spring, a groundwater-fed river or a borehole;

4. following a chemical spill, a leak from a pipe or underground storage tank (UST), or the identification of a practice which may have led to groundwater contamination.

In any case, the purpose of the preliminary site investigation is to use any existing data to make an assessment of whether or not contamination is present, and if contamination is shown to be present, to establish if there is a need to carry out a more detailed study. The key to this assessment is establishing the presence or absence of the source-pathway-target relationship. The presence of an apparent hazard (source), in the form of a spillage, for example, may not constitute a significant risk to groundwater if a migration pathway to a sensitive receptor is absent, or if a pathway exists but no receptor is threatened.

To quantify the source, any investigation should determine the magnitude, vertical and lateral distribution, and nature of the pollution. Other information should include details of the toxicity of the pollutant, the quantity lost and the period of time over which that loss occurred, as well as any other background data which are available.

Establishing the absence or presence of a pathway requires a conceptual model of the hydrogeological situation to be developed. At a general level, inquiries should be made to establish whether the site is located on a major, minor or non-aquifer, but any other background information on the geology, hydrogeology and hydrology of the area should also be sought at this stage.

The general categories of receptor which may be at risk from contaminated groundwater can be summarised as follows:

- abstraction borehole;
- spring;
- river or other surface water feature which is groundwater-fed - this could include wetlands, ponds etc.;
- the aquifer itself and its overall groundwater resource value.

Searches for abstraction licences should be carried out in the study area. The presence of other types of receptor may require more detailed inquiries to be made, for example, by field visits, studying topographical, geological and hydrogeological maps, discussion with Environment Agency staff, or liaison with Local Authorities, conservation groups etc.

Appendix B (Vol. 2) details the basic data that should be determined in this initial investigation. Without a reasonable knowledge of the contaminants and site specific characteristics it is difficult, if not impossible, to make any qualified decisions about clean-up and remediation.

The importance of carrying out this initial part of the study and any iterative further site investigations (3.2.2, 3.2.4) to an accepted minimum standard must be stressed. A basic check list of data should be presented to the company or consultant that the Environment Agency would require at this stage of the investigation. The ability to set quantitative clean-up standards (Section 4) is dependent on having access to adequate site database. The site investigations needed to provide the site database should be carried out according to recommended procedures for sampling and analysis, and a brief discussion of Quality Assurance (QA) programmes and Quality Control (QC) measures are discussed in Appendix B.

3.2.2 Qualitative assessment

Having completed the preliminary site investigation, a qualitative assessment of the available data can be carried out with the objective of deciding whether or not the site poses a potential threat to a receptor. If it is clear from the available data that there is no real risk associated with the groundwater contamination, then no further action may be necessary.

If, however, this qualitative assessment indicates that there is a potential risk associated with the groundwater contamination, then a more detailed study of the situation will be required, and it may be necessary to undertake further investigations to ensure that the behaviour of the groundwater system is fully understood. Failure to establish a full understanding of the hydrogeology may result in the setting of inappropriate clean-up standards and/or the choice of inappropriate remediation techniques; such mistakes could be very costly.

In practice, the full characterisation of a groundwater system may be an iterative process, particularly in areas where little background data are available. An initial conceptual model will be proposed by the hydrogeologist which will gradually be refined during the progress of investigations. It is perhaps, therefore, slightly artificial to compartmentalise the process as has been done in Figure 3.2; however, it is necessary to do this in order to develop the overall assessment methodology.

If, although a source and receptor are identified, a route for migration of the contaminant is apparently absent, then whilst there should be no immediate need for remedial action, the site should be monitored to ensure that conditions do not change, that the system remains one that poses no risk to groundwater and that there is no direct threat to surface waters, for example by interflow or via underground services.

3.2.3 Costs and Benefits

Clause 37 of the Environment Act 1995 imposes a general duty on the new Agencies (Environment Agency and Scottish Environment Protection Agency) to have regard to costs and benefits in exercising powers (See Section 2.3). Clause 37 of the Act states that:

> "Each new Agency in considering whether or not to exercise any power conferred upon it by or under any enactment, or in deciding the manner in which to exercise any such power, shall, unless and to the extent it is unreasonable for it to do so in view of the nature or purpose of the power or in the circumstances of the particular case, take into account the likely costs and benefits of the exercise or non-exercise of the power or its exercise in the manner in question" (HMSO 1995)

The Statutory Guidance (DoE 1995b) reiterates that this general duty does not apply if it would be unreasonable in the circumstances of a particular case and does not affect the Agency's mandatory obligations to discharge specific duties and legal requirements remain unaffected by the duty. The duty will apply, however in those instances where there is more than one way of achieving the legal requirements and if the Agency retains discretion as to how they should be achieved.

Within this framework of considering costs and benefits Ministers have acknowledged that not all costs and benefits need to be precisely quantified (particularly due to the inherent methodological difficulties in valuing the environment).

The Statutory Guidance recommends that the following methodologies and procedures may be considered:

- principles, procedures and techniques for giving proper consideration to non-market impacts;
- the precautionary principle;
- reliance on sound science;
- likely impact on carrying capacity of the environment;
- likely longer term implications and effects;
- likely costs and benefits of its actions for society as a whole;
- evidence with the UK and internationally proven and likely impacts on the environment;
- impacts on the economy (including affected business sectors and individual companies);
- the distribution of costs and benefits across the economy.

The methodology developed in Appendix C, Volume 2 addresses these issues and helps the Environment Agency to develop practical procedures to ensure that it meets the requirements of the general duty contained in Clause 37.

A prime consideration of any clean-up operation is what level of clean-up can a developer or polluter be realistically expected to fund. A key consideration in defining the scale and level of clean-up has therefore been an assessment of the benefits that could be expected by any remedial programme. Where the benefits are not easily quantifiable in monetary terms, the approach adopted in this report is to identify the benefits in qualitative terms through a structured environmental impact assessment and select the most cost effective solution.

As the remediation of polluted groundwater is associated with risk and uncertainty a structured Decision Analysis Framework has been developed to assist the Environment Agency to identify the best remediation strategy at a site level and justify its actions in accordance with the general duty imposed by Clause 37 of the Environment Act 1995.

3.2.4 Quantitative assessment and definition of clean-up standards

The preliminary site assessment will have established the hydrogeological setting of the pollution incident. A second phase of site investigation may be needed to characterise more closely the hydrogeology and pollution transport mechanisms at the site as well as the physical and chemical characteristics of the pollutants. A grid of boreholes may be required to define areal and vertical variations in the hydrogeological characteristics which may influence groundwater and pollutant migration routes. This more detailed quantitative survey in some cases will involve modelling techniques to establish the severity of the contamination.

These hydrogeological studies should be sufficiently detailed to provide the information on which the remediation programme can be designed and the clean-up standards defined.

The technique for assessing the necessary remediation programme and the methodology for setting clean-up targets is discussed in detail in Section 4.3. The assessment procedure (Figure 4.2) establishes a prioritisation for sites in terms of the importance of the receptor at risk and the imminence of harm to the receptor. Major aquifers currently used for drinking water supply have been awarded the highest priority for action and clean-up would be required to the tightest standards i.e. at least drinking water quality. Similarly in a minor aquifer where there is no current utilisation of groundwater the given contamination incident should not be seen to elevate existing background levels of the pollutant in the aquifer. A key element of this scenario is to establish 'current' background water quality in the aquifer. In this way, those aquifers that are already polluted, clean-up would not be required over and above existing background contaminant levels. Non- aquifers have been awarded the lowest priority for action although a restricted monitoring programme may be required to confirm that the contamination incident does indeed pose no future risk to groundwater.

The key to the procedure is the identification of five risk scenarios:

1. Groundwater abstractions at risk;
2. Surface water features at risk;
3. Major aquifer at risk, groundwater not exploited at present;
4. Minor aquifer at risk, groundwater not exploited at present;
5. Site located on a non-aquifer.

Each of these five has been assigned a priority level, based on a broad assessment of the threat posed by each, from Priority 1 for the potential pollution of an abstraction, to Priority 5 for pollution which has occurred on a non-aquifer. For each of the priority levels a flow chart points to a more detailed assessment procedure.

The assessment and prioritisation of a site will be completed by the selection of a suitable mechanism for actually setting the groundwater clean-up standard. The proposed methodology involving back-calculation from an agreed water quality at a receptor or surrogate background borehole has been referred to in Section 2.3 and is discussed in detail in Section 4.3. The mechanism involved in the simplest circumstances may be merely an analytical solution or dilution calculation but in complex situations it could involve a groundwater pollution transport model. The simple analytical solution to calculate the effective pollution load at the receptor assumes equilibrium conditions exist in the aquifer, i.e. all the pollution from the source is entering the borehole. This obviously represents a worst case scenario and ultimately provides a safety margin to the calculated clean-up value. Similarly, because the compliance borehole by definition must be located within the centre of the pollution plume, this provides an added safety factor than if the compliance borehole were located on the edge of the plume, where concentrations of the pollutant would probably be lower. It should be noted that when using the analytical solution, the clean-up target set for the compliance borehole is independent of the distance between the source and receptor. It is not proposed to select a specific model since model specifications improve with time and the demands may be site specific. Guidance is provided on the choice of model to ensure the optimum model is chosen for each remediation project. Both the selection of the model structure (i.e. the type of model) and the assembly of the descriptive data are important in obtaining the *correct* model for any situation. The aims of the modelling are also important and must be considered prior to selecting the model to be used. Unrealistic aims will result in selection of an over-complex, data-hungry model which is unnecessarily sophisticated.

3.2.5 Select clean-up technology

The selection of the most valid method of aquifer clean-up will largely depend on the aquifer characteristics, and the physical and chemical properties of the contaminant. To a large extent these factors will influence the ultimate success of the remedial works i.e. the benefit observed, and will directly influence the likely timescales and costs of that work. A summary of *ex situ* and *in situ* groundwater remediation techniques is given in Appendix D. Whilst assigning specific costs to individual clean-up techniques is difficult, an attempt has been made to provide guidance figures from historic clean-up operations although many of these have been carried out in the US. A summary is also provided of the usefulness of application of the major techniques and the compounds for which they are likely to prove the most successful.

3.2.6 Implementation

Having decided that a particular contamination incident requires action, there is a number of ways in which the problem can be approached which include:

1. actively remediate the pollutant at source;

2. actively remediate the plume of pollution to contain or reduce the concentration of pollution in the groundwater;

3. allow natural attenuation to remediate the pollution (with strict monitoring to show that concentrations will not exceed a given maximum and degrade).

The groundwater clean-up operations could include:

- free product recovery;

- *in situ* or *ex situ* remediation;
- on-going monitoring of groundwater until the selection of the most appropriate course of action.

The level of clean-up will be dependent on the value of the resource and hence the targets assigned to any remedial action and ultimately the cost of selecting and implementing appropriate measures and technology. Any decisions on remedial options preferably should be made with Agency approval.

3.2.7 Compliance monitoring

Once clean-up standards for a given site have been defined, a programme of monitoring should be established to assess with adequate confidence whether or not compliance with the set conditions has been achieved. The monitoring point to measure groundwater quality and assess when the clean-up standard has been achieved will be the compliance borehole (Figure 3.1).

The definition of a compliance monitoring programme is essential. It is a commonly observed fact that when a remediation programme is stopped, having achieved a clean-up standard, the quality of the groundwater deteriorates because of desorption of pollutants from the aquifer matrix. The compliance monitoring should continue after the clean-up standard has been met and arrangements made for the remedial action to be reinstated, repeatedly if necessary, until the clean-up target is met and the groundwater quality remains better than that target for a prolonged monitoring period.

4. QUANTITATIVE ASSESSMENT TO DEFINE CLEAN-UP STANDARDS

4.1 Introduction

The framework of the work programme necessary to define a clean-up standard for polluted groundwater has been described in Section 3. The standard setting for a specific site will comprise several steps or phases that may be summarised:

- establish the pollution and hydrogeological parameters of the site by site investigation;
- prioritise the site in terms of the importance of its potential impact on receptors;
- set a clean-up standard for the site such that the impact on receptors will meet agreed criteria;
- monitor the site to demonstrate that the clean-up standard is being met.

The core of this programme is the quantitative phase which comprises two stages:

- setting scenarios for risk assessment in order to prioritise a site;
- mathematical modelling to set a clean-up standard for the site.

This quantitative assessment phase **must** be based on site investigations, possibly of several phases, which establish the database necessary for quantifying both the site prioritisation and the mathematical modelling. The site investigation phases are summarised in Section 3 and discussed in detail in Appendix B. A summary of the factors that should be determined in a comprehensive site investigation suitable for the quantitative assessment phase is given in Figure 4.1.

The quantitative assessment phase also will need support in terms of:

- a cost benefit assessment methodology to be used to demonstrate the relative merits of clean-up strategies or standards;
- the various clean-up methods and techniques available for the clean-up of a specific site;
- the Water Quality Standards to be agreed for the receptor to be used to set the clean-up standard at the compliance borehole.

These support functions are discussed in detail in Appendices C, D and E respectively.

A case history of a hypothetical site is set out in Section 4.3.5 to demonstrate clearly how a programme of setting a clean-up standard for a site may be applied.

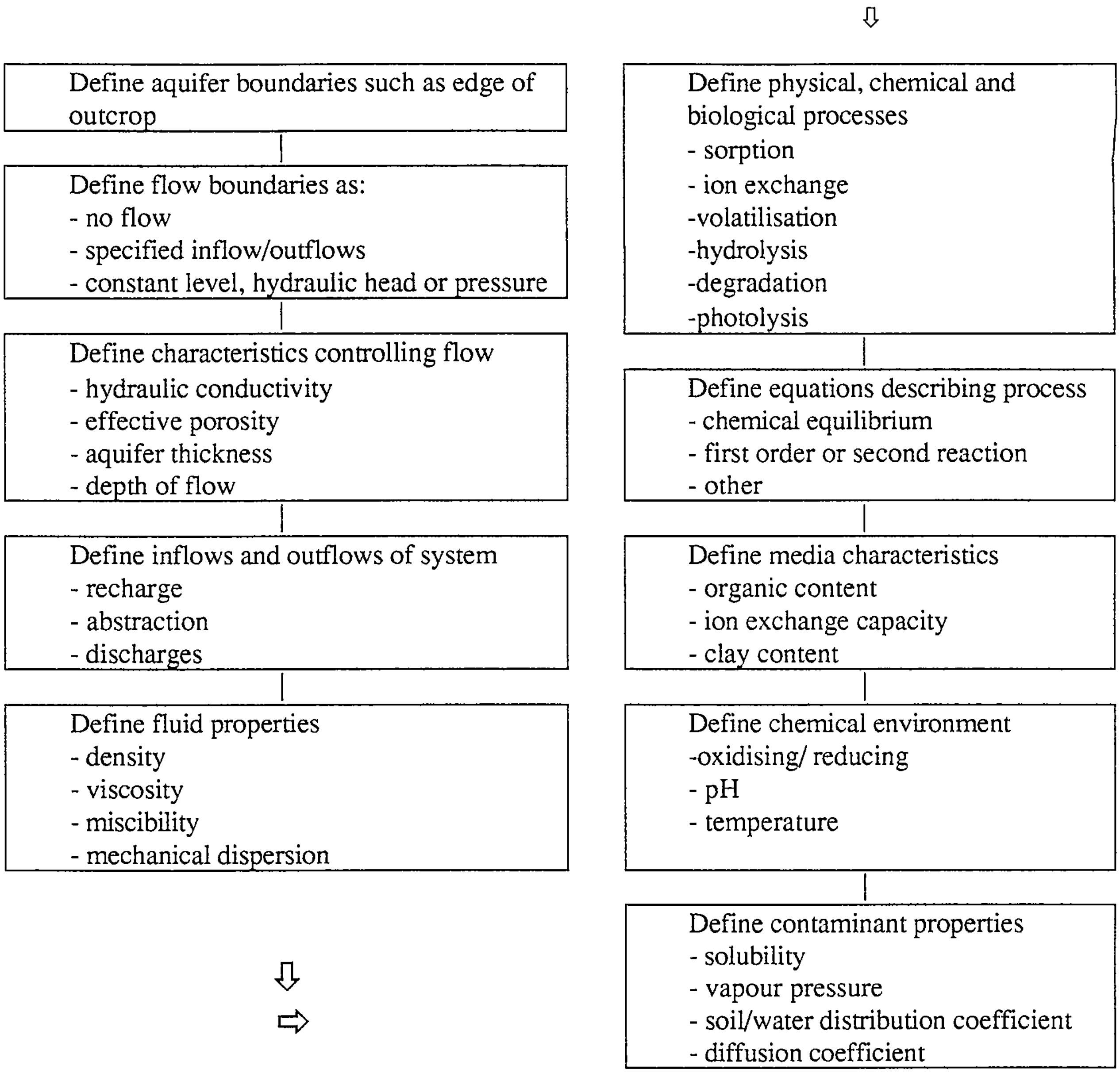

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Figure 4.1 Factors to be determined in detailed site evaluation

4.2 Risk assessment scenarios and site prioritisation

A full understanding of the hydrogeology of the contaminated site and of the properties of the pollutant(s) enable the first stage of a risk assessment and site prioritisation procedure shown in Figure 4.2 to be followed. The reason for this prioritisation of sites is to ensure that the appropriate remediation standards can be calculated on the basis of risk, rather than the application of a blanket standard for groundwater quality which would be unduly stringent in many cases. The methodology aims to identify the type of receptor at risk, and hence allow the application of appropriate quality standards; the distance of the pollution source from the receptor and hence the natural attenuation available; the existing background quality of water in the aquifer and the likely future utilisation of the aquifer.

Figure 4.3 provides a summary of international health-based or aesthetically-based guideline values and standards that exist for individual parameters, such as in the UK drinking water regulations, WHO water quality guideline values and the USEPA drinking water standards and regulations. Similarly, for discharge to surface waters where Environmental Quality Standards exist they and the various other directives regarding the various uses of surface waters should be consulted. A more detailed examination of water quality standards are presented in Appendix E (Volume 2). Where no such standards or guidelines exist, an appropriate assessment of the specific contaminant(s) would be needed in order to determine whether the concentration(s) present posed a threat to the intended use of that groundwater. A discussion of the water quality standards that are relevant to each of the five scenarios identified above is provided in Sections 4.2.1 to 4.2.5. These broad priorities may be subject to subdivisions based on local conditions; for example, borehole abstractions may be classified by end-use as potable, industrial or agricultural or surface waters may be subdivided (Figure 4.3).

The procedure has been designed to ensure that all potential risk scenarios are considered for any site. That is to say that if there is an obvious threat to a public supply borehole, the procedure should still force the user to consider the presence of other, lower priority, risk scenarios.

For each of the priority levels, the flow chart points to another, more detailed assessment procedure (Figure 4.4 to Figure 4.7), except Priority 5 which is sufficiently simple to be contained in one box of the flow chart of Figure 4.2.

4.2.1 Priority 1 - Groundwater abstractions (Figure 4.4)

For the scenario in which the quality of a groundwater abstraction is potentially threatened by groundwater pollution, classification has been based on the Environment Agency's groundwater source protection zones (GPZs), which are defined for all major licensed potable or similar abstractions. The abstraction could be a spring as well as a borehole. Three GPZs are recognised:

- Zone I (Inner source Protection), which is defined by the 50 day travel time from any point below the water table to the abstraction point, or a 50 m radius from the abstraction, whichever is the greater;
- Zone II (Outer Source Protection), which is defined by the 400 day travel time from any point below the water table to the abstraction point, or 25% of the total abstraction catchment, based on the long-term licensed abstraction rate;
- Zone III (Source Catchment), which is the total abstraction catchment based on the long-term licensed abstraction rate.

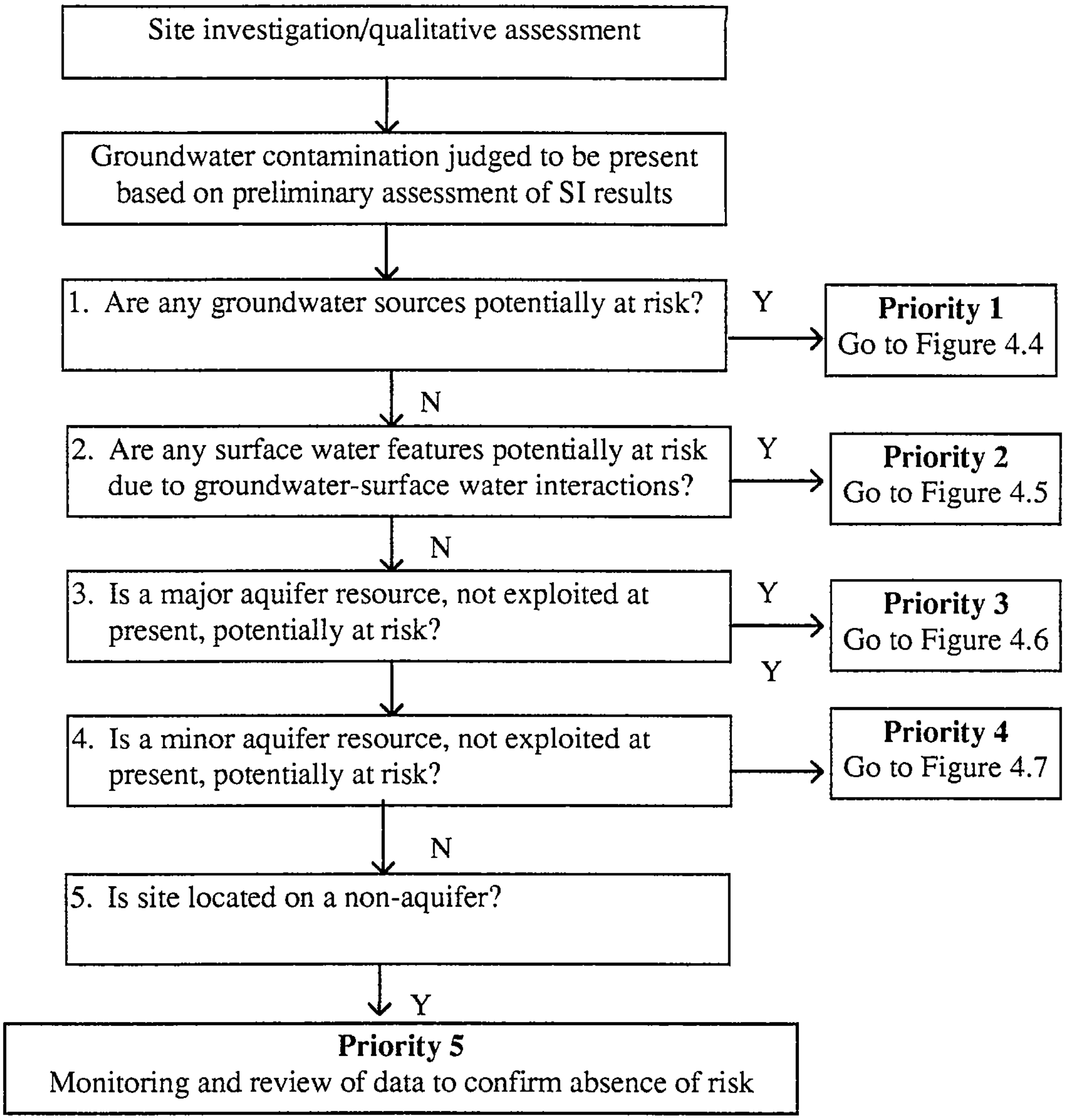

Figure 4.2 Risk assessment procedure for sites where groundwater contamination has been identified: Stage 1

	Receptor Category		Details of Receptor	Key considerations
1.	Groundwater resource, not exploited at present	i	'Good' background groundwater quality	Groundwater Directive (80/68/EEC) Is the aquifer likely to be utilised in the future (i.e. location land-use etc.). Used for what?
		ii	'Poor' background groundwater quality	Is the aquifer quality improving naturally; would it be considered as a resource in the foreseeable future
2.	Groundwater abstraction - spring or borehole	i	Utilised for public supply	The Drinking Water Directive (80/778/EEC) Water Supply (Water Quality) Regulations 1989
		ii	Industrial potable supply	Water Supply (Water Quality) Regulations 1989
		iii	Industrial supply for food grade products	Water Supply (Water Quality) Regulations 1989
		iv	Industrial supply - other uses (cooling, washing etc.)	Specific quality requirements e.g. low Fe
		v	Domestic supply well, spring or borehole	The Private Water Supply Regulations 1991
		vi	Agricultural supply well, spring or borehole	Some non-statutory standards suggested by ADAS.
3.	Discharge of groundwater to fresh surface water (springs), not directly used for supply	i	Providing baseflow to stream or river etc.	Surface Waters Directive (75/440/EEC), Freshwater Fish Directive (78/659/EEC) The Surface waters (Classification) Regulations (DoE Circular 7/89) Dangerous Substances Directive (76/464/EEC) (and Daughter Directives)
		ii	Feeding to lake, pond or reservoir	Use dependent e.g. fishing, cooling water etc.
		iii	Feeding wetland	Hydrogeology critical (dilution), and sensitive ecology
		iv	Canal	
4.	Discharge of groundwater to saline spring or brackish surface water (springs)	i	Direct to estuary, sea	Dangerous Substances Directive (76/464/EEC) (and Daughter Directives) The Surface Waters (Classification) Regulations (DoE Circular 7/89) Shellfish Waters Directive (79/923/EEC) Bathing Waters Directive (76/160/EEC)
		ii	To salt marsh	Hydrogeology critical (dilution), and sensitive ecology

Figure 4.3 Potential receptors of contaminated groundwater

The most urgent action is required in the case of groundwater pollution being identified in GPZ I (Urgency Level 1 in Figure 4.4). Action may include the closure of the abstraction or installation of temporary treatment pending physical removal of the source of pollution, for example by excavation. In this case, it is proposed that the same standard should be applied at the abstraction and at the compliance borehole. The water quality standard applied to the abstraction point will be agreed by applying the appropriate regulations (Appendix E); for example, if the borehole is a potable source then the Drinking Water Regulations would apply.

An Urgency Level of two has been set for GPZ II and one of three for GPZ III since the latter has a greater travel time and so provides a greater time for clean-up measures to be effective whilst still protecting the resource. In the case of groundwater pollution in GPZ II and GPZ III, the standard at the abstraction point should be the same as for GPZ I except where background water quality in the aquifer is of a worse standard than drinking water quality and in that case this would be the value used to calculate the clean-up standard. The methodology used for the back-calculation to relate the water quality standard agreed for the receptor (abstraction borehole) to the clean-up standard to be applied at the compliance borehole is described in detail in the next section (4.3).

The outcome of any of the choices in Figure 4.4 is that the user is taken back to Figure 4.2, box 2 to check for the presence of any other risk scenarios.

4.2.2 Priority 2 - Surface waters (Figure 4.5)

Groundwaters provide baseflow to many surface waters, which are subsequently used for potable, industrial and agricultural supply, as well as providing amenity and recreational value (Figure 4.3). The quality of the baseflow is therefore critical and it is important that groundwater pollution is controlled to limit derogation of surface water sources and sustain quality targets.

Surface waters have a number of important characteristics which may affect their response to groundwater pollution, and the kind of remedial actions which may be appropriate:

- The impacts of pollution may be acute, and clearly visible to the general population;
- There is a more general risk of exposure for human and animals once a pollutant reaches a surface water, and this can be via a number of pathways (e.g. potable abstraction, swimming, fishing);
- The ecological impacts of the pollution will need to be considered;
- Rivers and streams have a greater capacity for self-remediation than aquifers, but this does not apply to all categories of pollutant.

Figure 4.5 presents the assessment procedure for sites which pose a potential threat to surface water features; these sites have been assigned a Priority of 2.

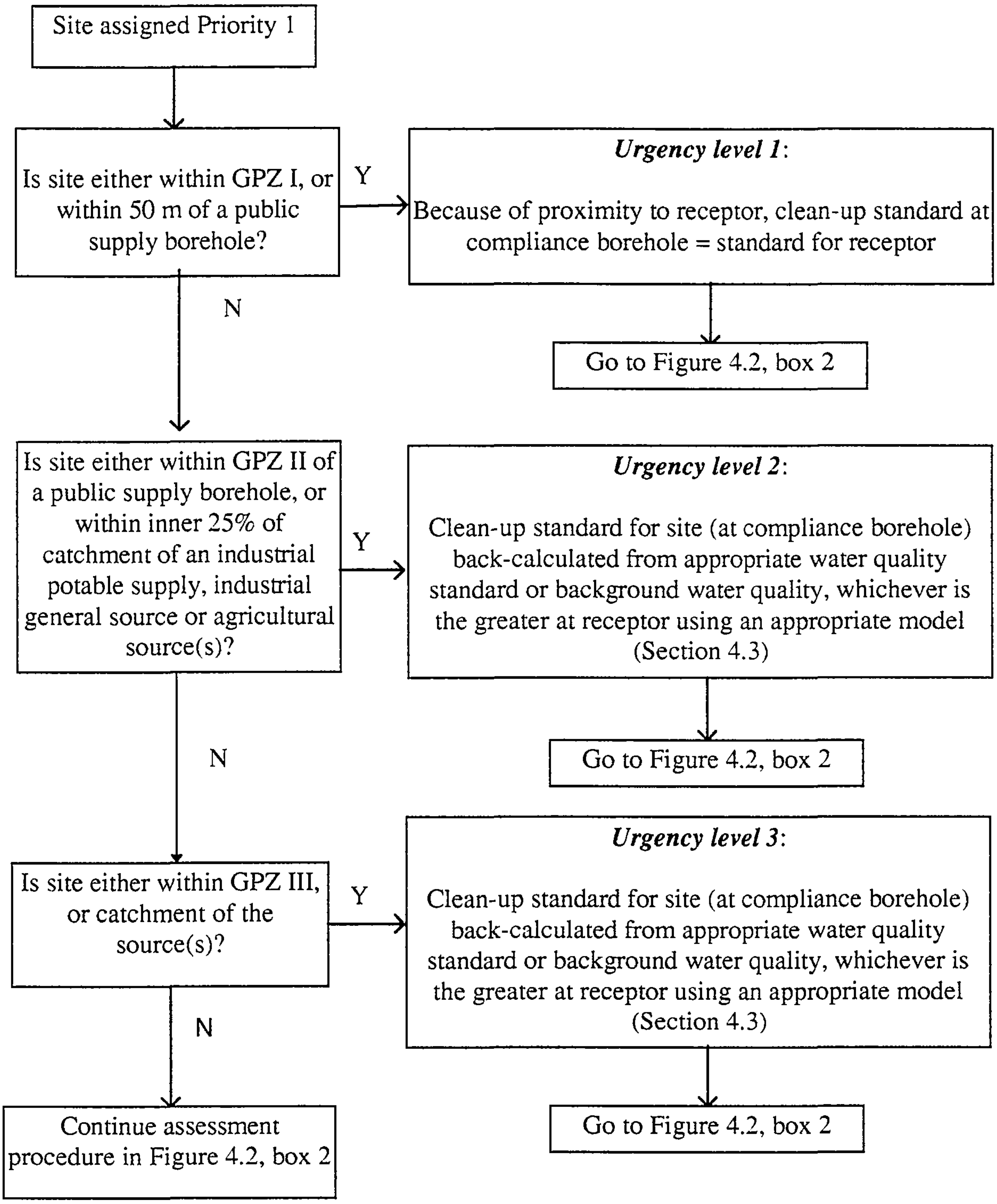

Figure 4.4 **Assessment procedure for "Priority 1" sites where a groundwater source or sources may be at risk**

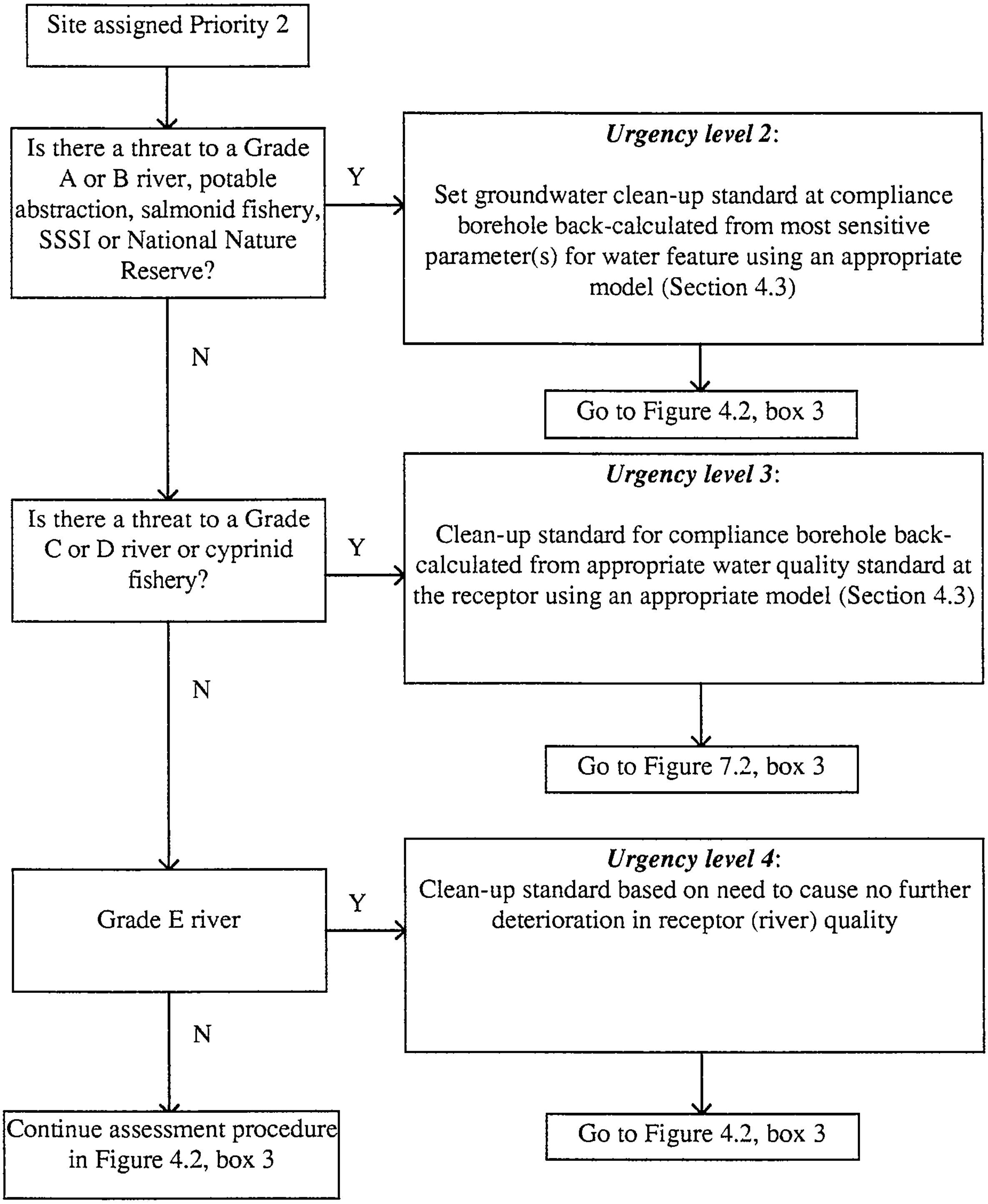

Figure 4.5 **Assessment procedure for "Priority 2" sites where surface water features may be at risk**

Three sub-divisions of surface waters based on the most recent General Quality Assessment (GQA) chemical grading have been proposed in terms of the urgency for remedial action and the methodologies for setting the groundwater clean-up standards:

1. Good quality rivers (Grades A and B); rivers with a potable abstraction; salmonid fisheries; SSSIs or other important conservation feature, other high quality surface water features.
2. Fair quality rivers (Grades C and D); cyprinid fishery; other moderate quality surface water features.
3. Poor quality rivers (Grade E); other low quality water features.

The classification of rivers under the GQA scheme (NRA 1994B) is intended to replace the old National Water Classification classes with a system which is more formalised and should allow absolute comparison between rivers in different area. The bases for the GQA assessment are given in Table 4.1 taken from NRA 1994B.

Table 4.1 GQA chemical grading for rivers and canals

Water Quality	Grade	Dissolved Oxygen (% saturation) 10-percentile	Biochemical Oxygen demand (ATU[1]) ($mg\ l^{-1}$) 90-percentile	Ammonia
Good	A	80	2.5	0.25
	B	70	4	0.6
Fair	C	60	6	1.3
	D	50	8	2.5
Poor	E	20	15	9.0
Bad	F[2]	-	-	-

[1] as suppressed by adding allyl thio-urea

[2] i.e. quality which does not meet the requirements of Grade E in respect of one or more determinands

In the first two Urgency Levels, the end use of the water should be taken into account when choosing the appropriate standard to apply for the back-calculation of groundwater clean-up standards; in many cases, reference may be made to the statutory Water Quality Objectives (WQOs). For Urgency Level 4, the clean-up target is to be based around the background river quality, possibly using data from the GQA classification.

In the case of surface waters assigned Urgency Levels 2 and 3 significant effort will need to be directed into investigation works, both of the aquifer and surface water. Investigations will include some form of contaminant transport modelling in order to back-calculate clean-up standard at the compliance borehole. The model will, in some cases, need to include surface water in the transport calculations, although it may be sufficient to work on the basis of a target concentration at the point of discharge (e.g. spring), taking into account a low-flow dilution factor in the surface water feature.

4.2.3 Priority 3 - Major aquifer, not exploited (Figure 4. 6)

There will be some cases where a groundwater pollution incident is identified on a major aquifer, but is remote from, and does not impact upon, an abstraction borehole or surface water feature. This scenario has been assigned Priority 3 and the assessment procedure is illustrated by Figure 4.6.

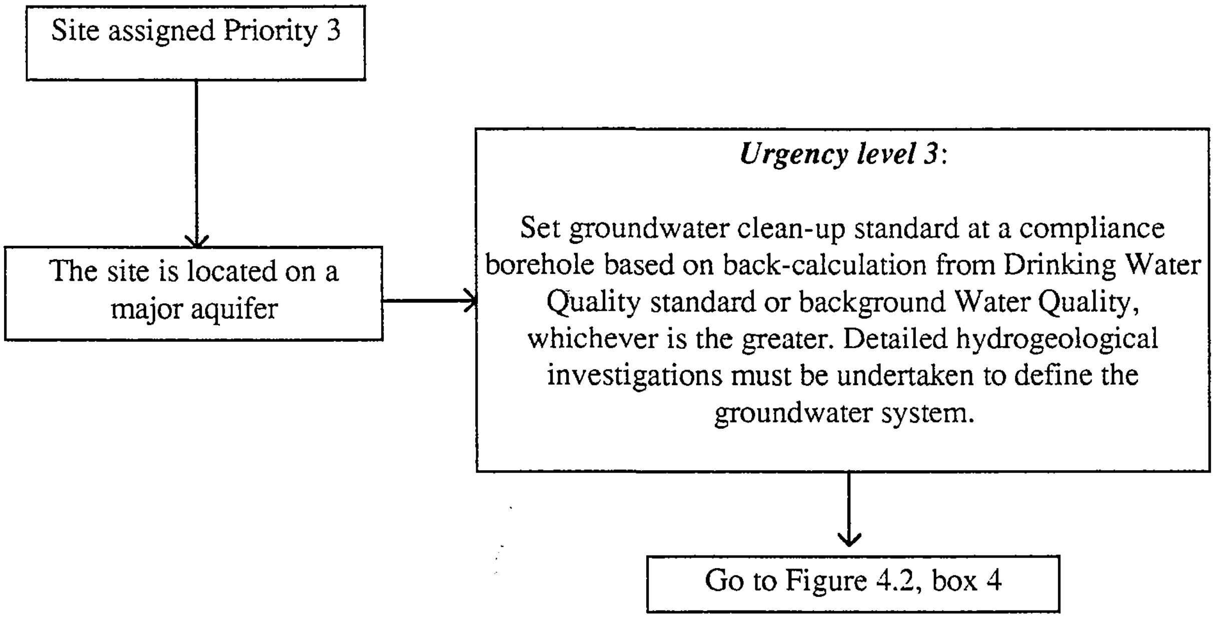

Figure 4.6 **Assessment procedure for "Priority 3" sites where the overall quality of the groundwater resource in a major aquifer may be at risk**

In this situation, the principal objective should be to preserve the overall quality of groundwater in the aquifer so that it remains available for future exploration. The unexploited nature of the aquifer means that there are no readily identifiable receptors within 1 kilometre of the pollution site. In such a case a surrogate 'receptor' should be used in any calculations (Section 4.3) set at 1 kilometre or 400 days travel time down gradient of the pollution incident (Figure 3.1). The agreed water quality standard at the surrogate receptor will be based on drinking water standards or background concentrations whichever is the greater.

This scenario is of a lower priority than "Priority 1" (Figure 4.4) because of the lack of a target at immediate risk. The methodology for setting of the clean-up standard, however, is similar to that for Figure 4.4.

4.2.4 Priority 4 - Minor aquifer, not exploited (Figure 4.7)

If groundwater contamination is identified in a minor aquifer currently not utilised as a resource (Priority 4, Figure 4.7), then the approach taken in assessing the risk and choosing the methodology for setting the clean-up standards will be the same as that for a major aquifer.

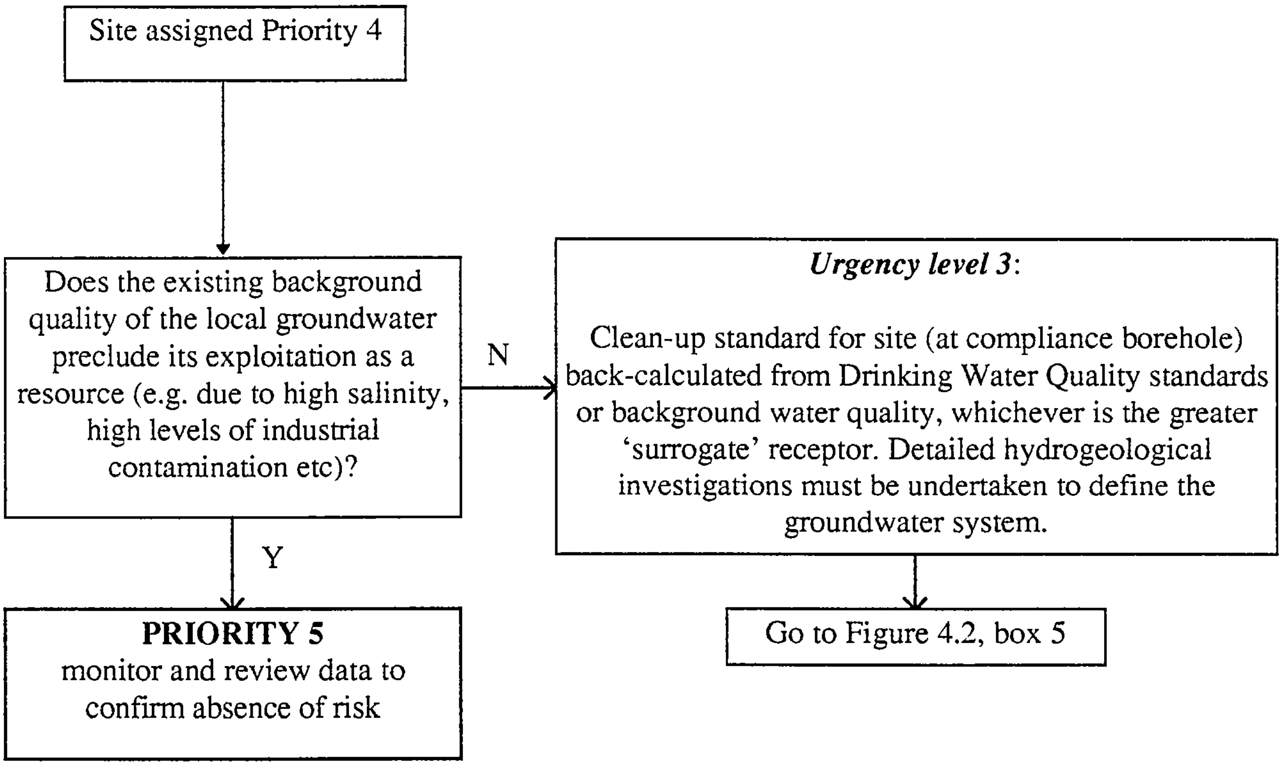

Figure 4.7 Assessment procedure for "Priority 4" sites where the overall quality of the groundwater resource in a minor aquifer may be at risk.

The approach taken in Priority 3 and 4 case situations, should take into account any existing deterioration in the background quality of the aquifer. This is because the aquifer (especially in the case of minor aquifers) may already be of low resource value due to the natural groundwater chemistry (e.g. high salinity in coastal areas, high iron levels etc.) or because of widespread anthropogenic inputs of pollutants (e.g. pesticides, chlorinated solvents).

4.2.5 Priority 5 - Non-aquifer

The final stage of assessment (Figure 4.2) is to confirm whether or not the site is located on a Non-Aquifer. If the site is located on a Non-Aquifer, then a final assessment should be made to establish that there really is no pathway and end receptor for the contamination. It is possible that groundwater contamination could be identified at a site located on a Non-Aquifer (as designated by the Agency), for instance in made-ground or small deposits of permeable drift overlying a non-aquifer. Points which should be considered include:

(i) some small licensed groundwater abstractions take place from small drift deposits which may not have been classified as an aquifer;

(ii) contamination which is present in a shallow, permeable drift deposit overlying an impermeable bedrock may well pose an immediate threat to surface waters;

(iii) Non-aquifers may allow significant migration of pollutants, for example in clays due to the presence of desiccation or settlement cracks, because they form only a thin deposit or because the formation has a higher permeability to the pollutant than to water (due to a higher density or lower dynamic viscosity).

(iv) the pollutant may accumulate in made ground over an impermeable layer and subsequent development works may disturb this pollution, for example, where building piles penetrate boulder clay over an aquifer, or site dewatering works may result in the contaminated water being discharged to a surface water or a sewer.

4.3 Mathematical modelling to set clean-up standards

Mathematical models can be used to simulate the behaviour of water and contaminants in aquifers, and provide a rational basis for predicting the effects of various strategies, including groundwater clean-up, on groundwater contamination levels. The groundwater model in this project has to be able to model the contaminant transport within the groundwater plume to an appropriate receptor. The model is then used to back-calculate a clean-up standard at a compliance borehole from an agreed water quality standard at a receptor or surrogate borehole (Figure 3.1). The application of the model is described in Section 4.3.5.

Groundwater models comprise equations to represent the movement of water and contaminants and data to describe the aquifer and associated inputs. Two different types of model can be applied to most groundwater problems;

- an analytical model, in which the equations describing the movement of water and contaminants can be solved exactly, or analytically, to give an equation relating outputs to inputs;

- a discretised numerical model, in which the equations and boundary conditions are solved by finite difference, or finite element, techniques, to provide good approximations to the exact solution at model nodes spaced across the aquifer domain, at discrete times.

It is important in any clean-up situation that the correct type of model is applied to the problem. In general terms the analytical solution may be applied to the less complex situations while the discretised numerical model may be more suitable for complex situations with multiple sources or multiple receptors. Alternatively the analytical model can be used as a first approach to a problem and then a discretised numerical model developed as more information is assembled in the hydrogeological database.

Analytical models have the advantage of representing the full solution in one equation, but cannot be applied to all situations. The Theis equation is the analytical solution for a fully penetrating well pumping at a constant rate in an infinite, confined, homogeneous aquifer. For any particular combination of aquifer properties and pumping rate, an almost identical solution can be obtained using a discretised numerical model, but the model must be re-run for different aquifer properties or different pumping rate.

Analytical does not necessarily mean simple or approximate, and in the case of contaminant movement from the edge of a pollution plume to a receptor some very useful solutions can be obtained for application to the present study. The four categories of receptor previously described need to be considered, because each will give rise to a different model.

The four categories of receptor are:

- an abstraction well;
- a spring;
- a river or other surface water feature which is groundwater fed;
- the aquifer itself.

4.3.1 Analytical solution for abstraction wells

If the compliance borehole is within the source catchment area of an abstraction well, then the groundwater flow regime is defined with sufficient certainty to make the model requirements quite modest. The best approach in this case is to calculate a dilution factor based on the load from the contaminant source and the rate of abstraction from the well. It should be noted that the load is the determining factor, rather than the pollution concentration in the plume. Concentration will vary spatially within the plume, possibly from very high values down to background levels or to the limits of detection. It will be impossible to characterise the plume with sufficient accuracy to permit a sound calculation of concentration at the receptor well.

If the load in groundwater at the compliance borehole is $L\ g\ a^{-1}$, and the rate of abstraction is $Q\ m^3\ a^{-1}$, then the long term concentration in the abstraction well will be:

$$c = L/Q\ mg\ l^{-1}$$

The concentration c, can be set as the agreed Water Quality Standard at the abstraction borehole (receptor) and the equation used to calculate the maximum permissible load at the compliance borehole. The method for converting the maximum permissible load L, to the clean-up standard concentration, Cs, is described in Section 4.3.5. The maximum permissible load at the compliance borehole is useful in that it can be used as direct input to the model of the complementary project to calculate the clean-up standards for contaminated land.

This equation comprises the best model when it may be assumed that the contaminant is conservative and that the input load remains constant. If the input load or the abstraction rate vary, then average values may be used to give reasonable predictions. If the contaminant degrades in groundwater a more sophisticated assessment is required. In most cases degradation will be, or can be assumed to be, zero and the above model may be used without being overly pessimistic.

It must be emphasised that the simple mass balance model will give the same long term results as a sophisticated, distributed groundwater flow and transport model for conditions of constant inputs and outputs, and providing the contaminant source is located within the capture zone of the well. This is illustrated in Figure 4.8 which shows predictions from detailed groundwater models in comparison with the results from the mass balance model. Two simulations are shown in Figure 4.8, one for a 3-d, dual porosity model of a Limestone aquifer, and the other for a 2-d, single porosity model of a thin sand and gravel aquifer. It is assumed in the detailed simulations that a constant contaminant load is applied to the aquifer 1200 m upstream of the abstraction well at time zero. The total transmissivity and porosity of the aquifer are the same in the two cases.

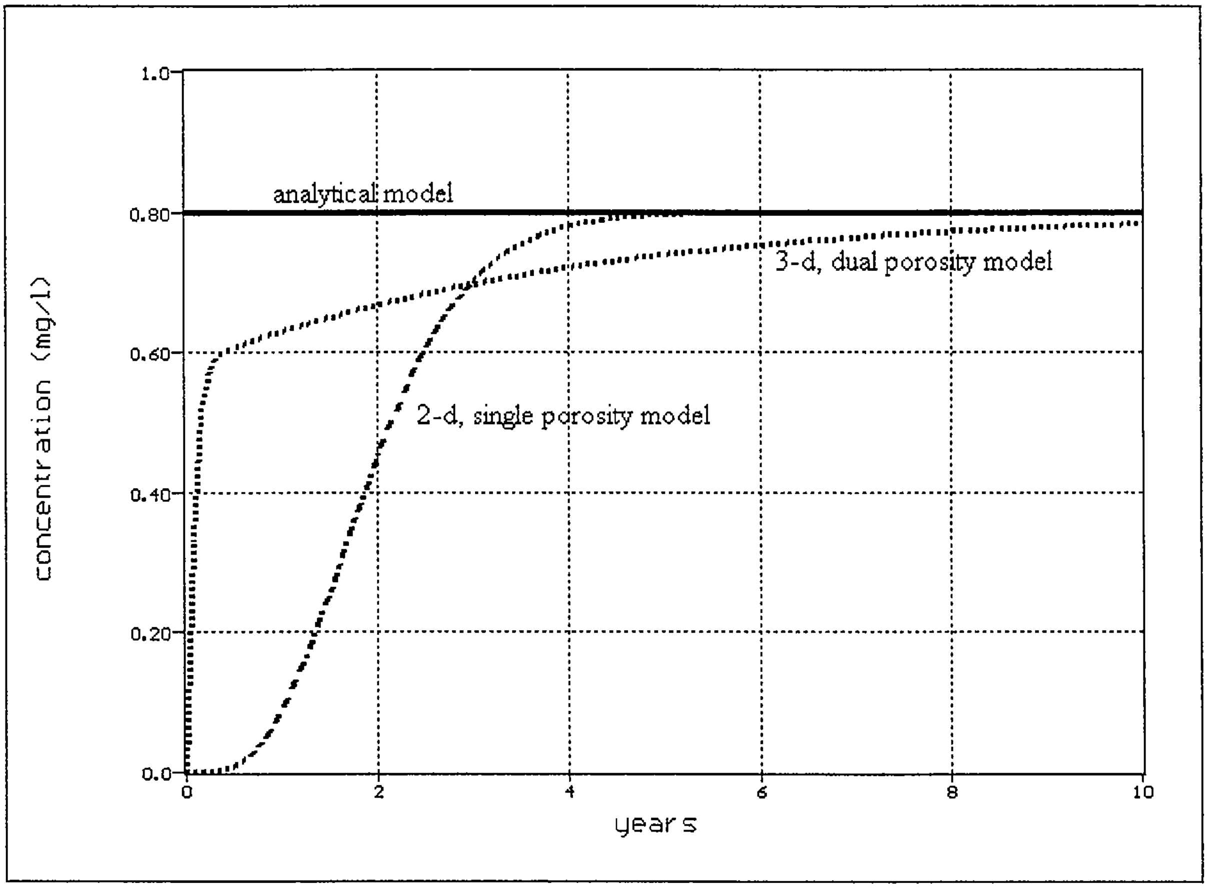

Figure 4.8 Comparison of analytical and detailed numerical model solutions for a constant load

It is apparent that the simple mass balance model can provide the correct long term answer, but not the time-dependent behaviour between the start of contamination and the establishment of equilibrium conditions. This may be important for determining the time scale of remediation impact.

The mass balance model can be used when there is more than one contaminant source within the catchment to an abstraction well, by simply adding the loads. It can also be used when a source is on the edge of a catchment area by taking into account only that part of the contaminant load which falls within the source catchment area. In this case any uncertainties in the contaminant plume and catchment boundary definitions can be accommodated by taking the worst case assumption in which all of the source falls within the capture zone of the well.

The simple mass balance model has been applied to the Helpston/ Etton contamination incident (Dames and Moore, 1993). Application of the above equation gives a concentration in the Etton well of 25 $\mu g\ l^{-1}$ Mecoprop for the long term average. This is very similar to that predicted by Dames and Moore with a sophisticated model requiring a significant database of groundwater levels and aquifer properties.

4.3.2 Analytical solution for springs

Springs behave in a similar way to abstraction boreholes, except that the rate of discharge is determined by local hydrogeological conditions rather than being imposed. The mass balance model used in Section 4.3.1 can be applied to a spring discharge. The long term average rate of discharge will replace the imposed abstraction rate in the equation.

A further problem arises if the contamination from the source penetrates to some depth before reaching the vicinity of a spring directly down gradient. In this case only some of the contamination will discharge to the spring, with the remainder continuing to migrate further down-gradient. This occurs because the catchment to the spring has a 3-dimensional character, with flows near the water table making the greatest contribution. If the spring is very large, or occurs at the feather edge of an aquifer outcrop then all of the contamination can be assumed to discharge to the spring, and the mass balance equation is valid. In other cases a more detailed model study may be required.

It should be noted that application of the mass balance model will always provide a worst case result, but may ignore the impact on other receptors down-gradient.

4.3.3 Analytical solution for rivers

Flows in rivers are, of course, very variable and will usually comprise both surface run-off and baseflow elements in groundwater areas. The specification of a receptor is difficult to define; at the most stringent level it could be applied to baseflow over a very limited length of river, and at the other extreme it could be applied to the total discharge from a groundwater catchment. These extremes could be covered by detailed, spatially discretised groundwater models with a coupled river flow calculation. However, for the purpose of this study a simple approach has also been investigated, comparable to the mass balance equation discussed above. If this is considered to be inappropriate then recourse must be made to a detailed model study.

The receptor will normally be located at some point on the river downstream of the zone into which the contaminant source will discharge. Upstream of this zone contamination from the source is expected to be zero. The total river flow at the receptor site will have an average value given by:

$$F = A * ER$$

where: A is the catchment area upstream of the site (m^2)

ER is the average annual effective rainfall ($m\ a^{-1}$)

F is the flow ($m^3\ a^{-1}$).

An alternative value of F to be used in such an analytical approach in a well defined catchment is the low flow value calculated by the Environment Agency. This will reflect a worse case scenario compared with that calculated from the average annual effective rainfall.

The average long term concentration of contaminant in the river at the receptor site is then given by:

$$c = L/F \text{ mg } l^{-1}$$

where L is as defined previously.

At first sight this seems to be a very simple, perhaps very approximate solution. However, it will provide a good approximation in all cases because;

- the model preserves the contaminant mass balance;
- groundwater systems are very dispersive, so that fluctuations in baseflow will not give rise to significant fluctuations in concentrations.

4.3.4 Analytical solution for the aquifer

The problem with defining standards for an entire aquifer is the variability in contaminant concentration which will arise from a source of finite extent. If all of the aquifer is to be protected to the highest standard, then the concentration in the compliance borehole must not exceed the target value. This, however, is unrealistically stringent, and it will generally be more practical to protect the aquifer for its resource value. In this case a surrogate receptor (borehole) is set at 1 km down groundwater gradient from the site. The receptor should be allocated a realistic but surrogate discharge of 1000 $m^3\ d^{-1}$ (1 Ml d^{-1}). The calculation then is identical to that for an abstraction borehole:

$$c = L/365000 \text{ mg } l^{-1}$$

where L is the load in groundwater at the compliance borehole (g a^{-1}).

If a smaller source is required as the target receptor, then the equation can be modified accordingly.

4.3.5 Calculations to set clean-up standard

Case Study 1

The analytical solution appropriate to the situation must be selected from the four described. As the first example, the case history of the Helpston/ Etton incident will be used. In this case landfills at Helpston leaked mecoprop into the underlying aquifer and threatened the public supply borehole at Etton.

The data shown in Figure 4.9 are derived from the Dames and Moore Report (1993). The outline of the plume has been drawn to encompass all observation points with recorded mecoprop concentrations greater than 1 µg l^{-1} (ten times Drinking Water Standard). The uncertainty of the plume outline illustrates the difficulty of obtaining sufficient data on which to base standards. This is an intensively studied site yet the data on groundwater pollution are inadequate.

The Etton discharge is assumed to be 6000 m^3 d^{-1}. The aquifer thickness is 20 m and the aquifer permeability 200 m d^{-1}; thus the transmissivity is 4000 m^2 d^{-1} (T = K permeability or hydraulic conductivity x aquifer thickness).

A compliance borehole (Figure 4.9) is located downstream of the polluting landfills in the line of preferred flows identified by site investigations but some 250 m from the landfill boundaries and 1500 m from the Etton pumping station.

The maximum acceptable load at the compliance borehole, i.e. the load passing the section of the plume at the borehole that will result in the mecoprop concentration at the abstraction borehole exceeding 0.1 µg l^{-1} is:

$$L = c\,Q$$

where c = concentration of mecoprop in mg l^{-1}

Q = discharge rate in m^3 a^{-1}

L therefore equals 219 g a^{-1} (0.1 x 6000 x 365)

The concentration at a specific point in the plume, the compliance borehole, can be calculated using a simple relationship based on the aquifer flow. The flow is:

$$F = T.\ i.\ w$$

where T = aquifer transmissivity

i = groundwater gradient

w = width of the contaminant source at the compliance borehole(s) perpendicular to the direction of flow.

At Etton: T is accepted as $4000 \text{ m}^2 \text{ d}^{-1}$

$i = 0.0006$

$w = 1000 \text{ m}$

Therefore $F = 2400 \text{ m}^3 \text{ d}^{-1}$

This will be a good approximation close to the source but because of dispersion will be subject to errors which will increase with the distance from the source. The spread of the plume to the south east at Etton (shaded area of Figure 4.9) may be caused by such dispersion.

A reasonable relationship between concentration and load may therefore be inferred:

$$Cs = L / (365 \times T.\ i.\ w)$$

where Cs = the mean maximum permissible concentration in groundwater at the compliance borehole (mg l^{-1})

L = the maximum permissible load (g a^{-1})

At Etton $Cs = 219 / (365 \times 2400)$

$= 2.5 \times 10^{-4} \text{ mg l}^{-1}$

or $0.25\ \mu\text{g l}^{-1}$

This simple calculation would therefore set a clean-up standard at the compliance borehole of 2.5 times the Drinking Water Standard.

The relationship between load and concentration at the compliance borehole has two inherent assumptions (Figure 4.10).

1. The plume of pollution is uniformly mixed across its defined width.
2. The plume of pollution is uniformly mixed through the thickness of the aquifer.

These assumptions mean that the clean-up standard calculated is the mean concentration through the plume and will be lower than if it was calculated for the centre of the plume where the compliance borehole is located. The result is that the clean-up standard has a built in safety margin, but a possible disadvantage could be that the standard is unduly onerous.

A refinement of this analytical method could more closely define the pollution plume. The data from Etton are insufficient but, if the data were available so that across a section of the plume, as in Figure 4.10, the mean pollution concentration contour could be defined and the cross-sectional area encompassed by that contour calculated then a refined clean-up standard could be calculated. At Etton, if we assume that the median concentration is 10 $\mu g\ l^{-1}$ and the contour encompasses 14000 m^2 (from the cross-section on Figure 4.10) then the necessary calculation is:

$$L = 219\ g\ a^{-1}$$

$$F = KiA \quad (K = \text{Aquifer Permeability and } A = w \times b)$$

$$= 200 \times 0.0006 \times 14000$$

$$= 1680\ m^3\ d^{-1}$$

then Cs (Clean-up Standard) $= L / 365\ KiA\ mg\ l^{-1}$

$$= 0.00036$$

$$= 0.36\ \mu g\ l^{-1} \text{ mecoprop}$$

Case Study 2

The analytical methodology for setting standards has been tested by a further example from the Lincolnshire Limestone. The results obtained by using the equation $c = L / (365.T.i.w)$ have been compared with the predictions from the detailed numerical model used to generate the concentrations for Figures 4.8 and 4.11. The model simulates flow and contaminant transport from a source to an abstraction well in a 3-dimensional, dual porosity aquifer, such as the Lincolnshire limestone. The comparison is summarised in Table 4.2. In this example, a source 200 m wide, located 1200 m directly upgradient from an abstraction well pumping 5000 $m^3\ d^{-1}$. The background hydraulic gradient is 0.003, but under the action of pumping the gradient increases as the well is approached. The aquifer is assumed to be layered, with an overall transmissivity of 900 $m^2\ d^{-1}$. A comparison between the results from the model and the concentration calculated with the analytical model is shown in Table 4.2.

The simple analytical calculation of concentration from load gives an answer which is about one half of that predicted for a shallow monitoring well intercepting the plume, and about twice that predicted for a fully penetrating monitoring well. This is a good level of accuracy, and indicates that the analytical approach can be used to convert loads to concentrations. It is important to recognise that for other aquifer/source/receptor conditions the comparison shown in Table 4.2 will be different, but the sound physical basis of the analytical calculation suggests that a reasonable comparison will be obtained in general.

Table 4.2 Comparison of analytical load/concentration relationship and model predictions

Distance from source (m)	Distance from well (m)	c = L/(365.T.i.w) (mg l^{-1})	Model prediction for shallow monitoring well intercepting plume (mg l^{-1})	Model prediction for fully penetrating monitoring well (mg l^{-1})
0	1200	7.8	14.4	4.0
200	1000	7.6	13.9	3.9
400	800	7.2	13.0	3.6
600	600	6.5	11.3	3.2
800	400	5.2	8.5	2.4
1000	200	3.3	4.8	1.4

4.4 Numerical models

The model applied first to a groundwater pollution situation or for setting clean-up standard will be an analytical solution. This first phase of modelling may be refined to take account of incomplete mixing across a pollution plume as described above. In complex situations, however, a numerical modelling approach may be necessary.

Detailed numerical modelling will be required whenever:

- the time scale of clean up effects is required;
- the input load varies significantly over time;
- hydrogeological conditions vary significantly over time, for example, changes in abstraction rates or locations of abstractions;
- clean up activities take place between the compliance borehole and the receptor, for example, pumping from a scavenging well;
- the contaminant degrades during transit through the aquifer;
- the contaminant is not fully soluble in water.

In many of these cases the analytical solution will give a worst case prediction, and can be used as a precursor to undertaking detailed modelling. As an example, the effect of a step change in input load is shown in Figure 4.11 for the analytical model and for a detailed numerical model. The numerical model is the 3-d, dual porosity model used before. It was assumed that the contaminant load halved after 5 years. The two analytical solutions shown are for the first five years (1), and for the second 5 years (2).

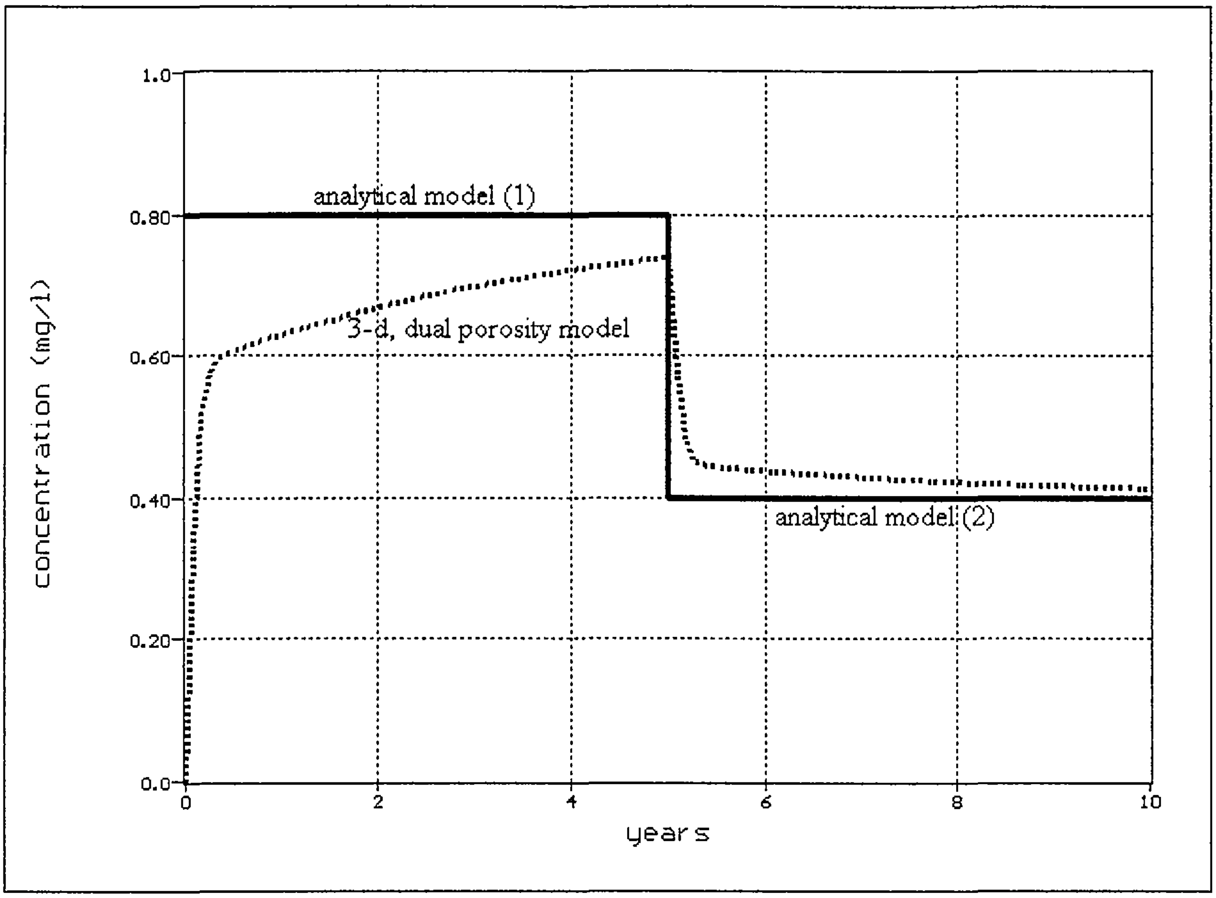

Figure 4.11 Comparison of analytical and detailed numerical model results for a step change in load

4.4.1 Features to be modelled

Both the selection of the model structure (i.e. the type of model) and the assembly of the descriptive data are important in obtaining the *correct* model for any situation. The selection of a suitable model is defined by a number of factors which include the local geology and hydrogeology, the extent of the aquifer and its boundary conditions, and the chemical and physical properties of the contaminant.

Figure 4.12 lists the factors which need to be considered in the selection process. The aims of the modelling are also important and must be considered prior to selecting the model to be used. Unrealistic aims will result in selection of an over-complex, data hungry model which may not provide useful results.

The first stage of model construction is to develop a conceptual model which will include those factors from the above list which are considered to be of significance to the problem in hand. It may not be immediately obvious which are the important factors, for example the representation of fissuring, and in many cases the conceptual model will be developed and refined as modelling proceeds.

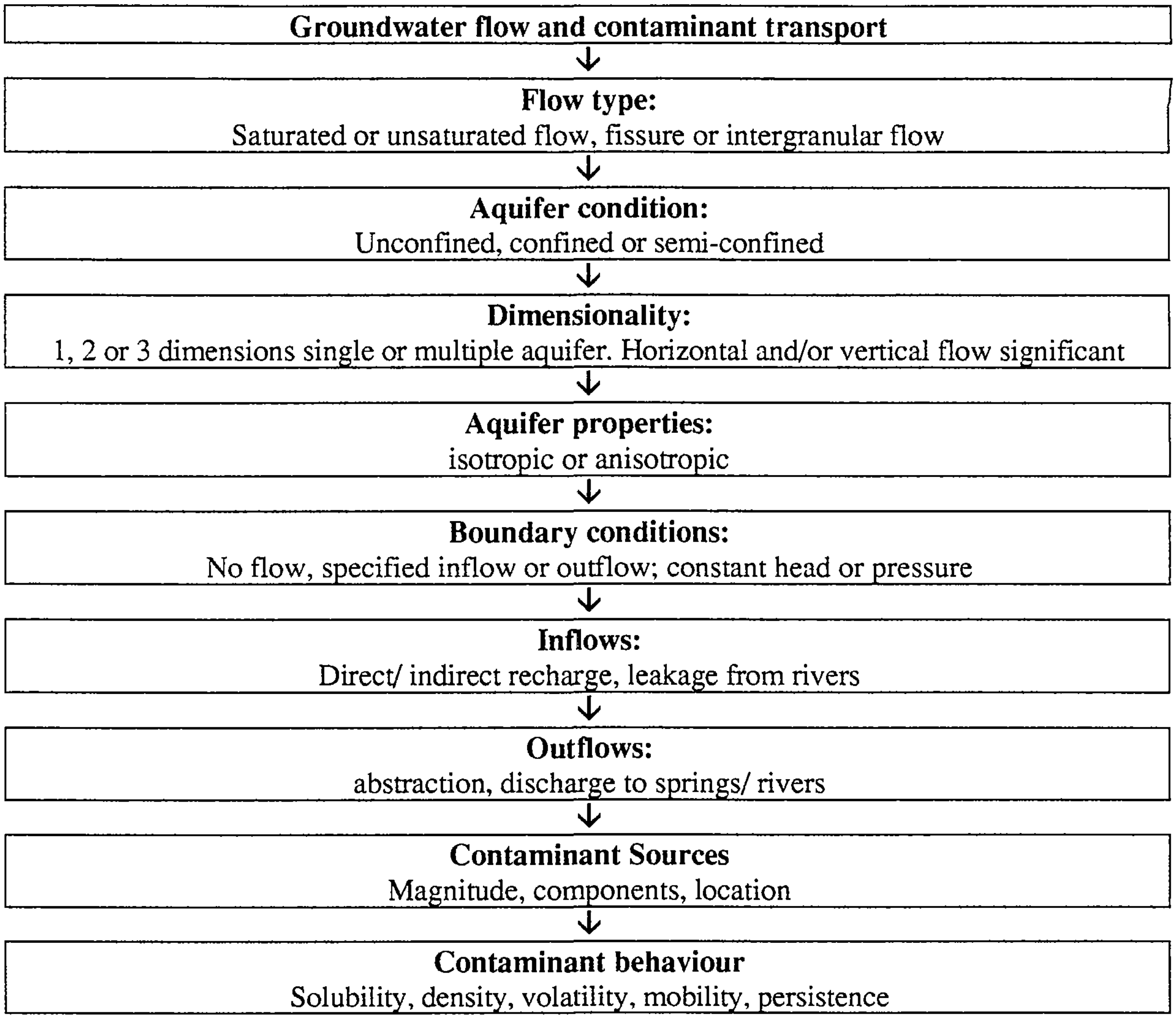

Figure 4.12 Factors for consideration in model selection

4.4.2 Key requirements of the conceptual model

Rates and directions of groundwater flows are of prime importance in determining the relationship between a pollutant source and impact at a receptor site. In outcrop aquifers, flows are largely determined by rainfall recharge and abstraction rates; porosity is important for calculating water flow velocities, but permeability is of lesser importance. Permeability does have a greater significance if the three-dimensional nature of aquifer flow is thought to be important. This may well often be the case for determining the source-receptor relationship.

Contamination from the source will reach the compliance borehole as a plume which will have a spatial distribution determined by the size of the source, soil and unsaturated zone characteristics, and the groundwater flow regime. Because groundwater is recharged at the water table, downward hydraulic gradients are developed in aquifers and pollution plumes will commonly penetrate to some depth below the water table. With increasing distance from the source the pollution plume will spread further both laterally and vertically. Concentrations will therefore vary with depth, and models to simulate the spread of pollutants need to take this feature into consideration.

Dispersion of pollutants in groundwater flow results from variations in flow paths arising from heterogeneities in the aquifer, and is manifested in spreading of the pollution plume and reductions in concentrations. Macro-scale and micro-scale fracturing can result in a high dispersion. Poorly sorted granular media can also give rise to significant dispersion. Dispersion coefficients increase with scale because at larger scales a greater degree of heterogeneity will be encountered. Dispersion coefficients have been measured in only a few media, and usually at a small scale.

In UK aquifers consideration must be given to both fracture and intergranular flows. In the principal aquifers fracture flow predominates and there is a high risk of rapid movement of contaminants from source to receptor with little opportunity for degradation or attenuation, except by dilution. Dilution may arise from diffusion of contaminants between the mobile water in the fractures and the essentially static water in the matrix. In many groundwater contamination studies this dual porosity behaviour has been identified as a key component of the flow regime and cannot be ignored. In some aquifers, for example alluvial sands and gravels, the movement of water and contaminants is entirely intergranular and a dual porosity model would be inappropriate, but all dual porosity models can be operated in single porosity mode by a suitable selection of parameters.

Many contaminants are not fully miscible in water and consideration must therefore be given to liquids which float on, or sink to the bottom of, the water in an aquifer. Some miscible contaminants behave differently to water because of interactions with the aquifer materials and various degradation processes. Interaction with the aquifer material will rarely be irreversible, so that contaminants retained by sorption will generally later be released into the water phase. This will result in the contaminants moving more slowly than water molecules. This will be of significance to the mass and concentration of contaminants in water if the process of degradation is also operative. Organics commonly degrade in soils, but evidence for degradation in the sterile conditions in aquifers is scanty. Nevertheless, both retardation and degradation should be included as options in groundwater models.

4.4.3 Appropriate models for setting clean-up standards

Application of the detailed models to the clean up situation is different to that of the analytical models described in Section 4.3. The detailed models cannot be run backwards to provide directly the source input needed to meet a target concentration at the receptor. Fortunately, however, all of the major processes operating in aquifers during water and contaminant movement are linear (in mathematical terms). This means that the output varies pro rata with the input, and this fact may be used to determine what changes are needed in the source input to meet the requirements at the receptor. Thus, if a model run with ten units of concentration (or load) at the source produces a concentration at the receptor which is twice the target value, than it may be deduced that an input of five units at the source will result in the target concentration being met exactly at the receptor.

The models itemised in Figure 4.13 are considered appropriate for the conditions specified at the beginning of this section, when analytical models are too limited. The models have been selected from the reviews undertaken by HMIP (1991) or BGS (1995) or have been used at WRc. It has not been possible from the reviewed material to fully determine the degree of difficulty of use of all of the models.

Some of the codes are relatively simple but can be applied only to problems of miscible transport in single porosity media. MOC and SUTRA are 2-dimensional examples, whilst HST3D and the MODFLOW/MT3D combination can simulate flow and contaminant transport in three dimensions.

CRAFT, FRACFLOW, SWIFT and TRAFRAP have the additional feature of being able to simulate flow in fractures or fissures and the diffusion of contaminants between water in the fractures and in the adjacent matrix. CRAFT is the WRc code which was used to generate the numerical model results in Figures 4.8 and 4.11 and it forms the basis of the ANNA model used to simulate nitrate pollution of groundwater.

ARMOS and SWANFLOW are multi-phase models, able to simulate the movement of water and oil. BIOTRANS utilises results from ARMOS as source terms to predict the movement of miscible contaminants leached from oil bodies floating on the water table.

Groundwater models	**Transport Processes**								**Dimensions**
	a	b	c	d	e	f	g	h	
ARMOS	◆	◆						◆	2
BIOTRANS	◆	◆				◆			2
CRAFT	◆	◆	◆			◆	◆		3
FRACFLOW	◆	◆	◆				◆		3
HST3D	◆	◆		◆		◆			3
MOC	◆	◆		◆	◆	◆			2
MODFLOW/MT3D	◆	◆							3
SUTRA	◆	◆		◆		◆			2
SWANFLOW	◆	◆						◆	3
SWIFT I/II	◆	◆	◆	◆		◆	◆		3
TRAFRAP	◆	◆	◆	◆		◆	◆		2

Processes modelled	**a**	advection/ convection
	b	dispersion
	c	diffusion between mobile and static water
	d	adsorption
	e	ion-exchange
	f	decay
	g	fissure/fracture flow
	h	immiscible liquids

Figure 4.13 Groundwater models suitable for detailed studies of aquifer clean up

4.5 Modelling strategy for setting clean-up standards

This project is concerned only with contaminant transport from within the pollution plume in the aquifer saturated zone to the receptor. The strategy for modelling aquifer clean up for this part of the total pollution pathway must clearly link with the strategy for the complementary project which is concerned with contaminant transport from the source into groundwater.

The strategy for this project is to use relatively simple analytical models as the first stages of monitoring. In such models the Environment Agency should follow the programme described earlier in this section which can be summarised:

1. use loads rather than concentrations as the link between the two projects;
2. use the models described in this report to calculate the load at the compliance borehole which results in acceptable concentrations at the receptor; the load so calculated then becomes the target for the complementary project;
3. use the analytical relationship described in Section 4.3 to convert the load to a concentration;
4. whenever possible use the analytical solutions described in Section 4.3 for the compliance borehole to receptor calculations;
5. resort to detailed numerical modelling if:

   these calculations will result in an initial clean-up standard to be set for the compliance borehole. The analytical solution will be refined as more data are derived and the pollution plume and groundwater transport controls can be more closely defined. Detailed numerical modelling should be used as a third stage of model development if:

   - the time scale of clean up effects is required;
   - the input load varies significantly through time;
   - changes in abstraction rates and well locations are to be considered;
   - clean up activities between the compliance borehole and the receptor are to be tested;
   - the contaminant degrades during transit through the aquifer;
   - the contaminant is not fully soluble in water.

Select an appropriate model from Figure 4.11, having regard to the important properties of the aquifer and of the contaminant. Assemble the hydrogeological data, construct the model, calibrate and validate. Use the model to select appropriate compliance borehole locations and set their respective clean-up standards.

5. COMPLIANCE MONITORING

5.1 Introduction

Once clean-up standards for a given site have been defined, a programme of monitoring should be established to establish with adequate confidence whether or not compliance with the set conditions has been achieved.

There are a number of key objectives in any remedial or exceedance monitoring programme. The programme should be designed to show whether or not:

- the quality of the aquifer is deteriorating further (baseline water quality must be established);
- the agreed standard has been achieved with the time constraints (if imposed);
- groundwater contamination trends are decreasing;
- contaminants are not migrating across set boundaries; and
    - risk to receptors remains within boundaries set during initial evaluation.

The programme should also enable the quantity of contaminant removed to be measured (if active removal techniques are employed).

The clean-up operation may encompass: natural attenuation or intrinsic bioremediation; active pumping used to contain the source of pollution; or some form of *in situ* works installed to remove bulk quantities of the contaminant. In all cases a programme of sampling should be designed to allow progress of the remedial works to be accurately monitored. These data will be needed to demonstrate the success of any containment or active clean-up programme or that natural attenuation factors are effectively dealing with the source of pollution. The monitoring will aim to demonstrate the exceedance or not of the agreed standards at pre-agreed locations in the compliance borehole(s) (Figure 3.1).

The monitoring programme needs to be such that changes in the levels of contaminants in the aquifer and behaviour of any pollutant plume can be detected in such a way that enables further corrective remedial action to be initiated well before further environmental damage, breach of any regulatory standards or exceedance of agreed clean-up standards is inevitable. Monitoring at a 'Sentinel Well' (Figure 3.1) located down-groundwater gradient from the site and the pollutant plume in addition to the compliance borehole will allow any worsening in the situation to be identified before impact at the ultimate receptor causes general derogation of the aquifer.

In general, monitoring of groundwater in an aquifer system involves the collation and analysis of data in order to provide clear views of the overall size and quality of the groundwater resource, in such a way that changes and trends may be recognised. In contrast the potential impact of point sources of pollution will often take place over a restricted zone and more focused techniques can be used to design and operate the monitoring programme than is possible in the case of general aquifer surveillance. It is, however, important that there is comparability between monitoring programmes for point source pollutants and any aquifer monitoring programmes if set up in the future.

A number of monitoring boreholes will be required to provide adequate data for assessment of the contamination at most sites. A minimum requirement for monitoring a given site should be considered to be an upgradient borehole (evaluation monitoring) and three boreholes sited at the boundary or a short distance (offensive detection monitoring) from the polluted site. If the given pollutant is significantly denser than water the use of nested observation boreholes should be considered to ensure vertical transport gradients within the aquifer are identified. The downstream boreholes should be located in a line perpendicular to the direction of groundwater flow. Ideally a fifth borehole should be located further downgradient to act as a sentinel monitoring well.

The objectives of compliance monitoring might be two-fold: to monitor baseline water quality in the aquifer upgradient from the contaminated site; and to measure the effects of the pollution and of a pollution clean-up programme specific to the site within the surrounding aquifer. In management terms, the former could be thought of as obtaining a reliable indication of the mean quality in the area over the time period of interest, and to determine whether it is improving or deteriorating. The latter would be set as water quality standards at a given compliance borehole for one or more quality criteria that would be specific to each site. The statistical objectives of the monitoring programme could be described as follows:

- to determine the *mean* water quality in the borehole, or group of monitoring boreholes, over the period of interest, with adequate precision and confidence; and
- to detect important *changes in mean* quality, between one time (monitoring) period and the next, with adequate confidence.

Background monitoring shows the levels of contaminants already present in the aquifer at any given site. These levels are baseline and subsequent increases suitably correlated show the environmental impact of the site

5.2 Analytical suites

The suites of analyses to be undertaken will vary from site to site but, because these have severe budgetary implications, they should be carefully thought out at the start of the monitoring programme. The initial site investigation should have identified the range of contaminants present in the groundwater, the concentrations that were present and basic site hydrogeology. These factors will be specific to any given pollution incident.

Obviously it is important that the identified ‘pollutants’ should form a key part of the analytical programme. In addition to these site-specific pollutants a number of general groundwater parameters should be included as follows:

- On-site measurements of pH, conductivity, DO, alkalinity and Eh;
- Basic water quality determinations which provide indications of organic pollutant load, including BOD (Biochemical Oxygen Demand) and TOC (Total Organic Carbon;
- Periodic inclusion of a basic ion balance suite.

Once monitoring is established and relationships between individual determinations and indicator parameters proven, then these could be used in the long-term to monitor bulk changes in the pollutant load. For example, BOD can be used to provide a measure of bulk organic pollution. Total Petroleum Hydrocarbon (TPH) analysis can be used as an indicator for various petroleum components, but it would still be necessary to monitor for the most toxic component of any mixture on which the standards for clean-up would have been based, e.g. benzene in a petroleum spill or carbon tetrachloride in a mixed spill of chlorinated solvents.

5.2.1 Pollutant monitoring: Individual chemical parameters versus group indicators

Indicator groupings

There is a number of parameters in the current UK Water Supply (Drinking Water) Regulations and the WHO drinking water quality guidelines which specify group parameters for instance, total pesticides (0.5 $\mu g\ l^{-1}$), hydrocarbons (10 $\mu g\ l^{-1}$), trihalomethanes (100 $\mu g\ l^{-1}$), surfactants and total PAHs (0.2 $\mu g\ l^{-1}$). Similarly, the use of group parameters has been suggested as a convenient way to specify and monitor groundwater quality standards. Such an approach has been tried in the US with respect to clean-up levels for petroleum products in soils. Several states have attempted to use Total Petroleum Hydrocarbons (TPH) concentrations to set soil clean-up levels that are protective of human and ecological health and to initiate remedial actions. However, using TPH concentrations as a clean-up goal is problematic in several ways.

- TPH concentrations are difficult to define chemically as the composition of environmental petroleum hydrocarbon mixtures changes over time.
- Different analytical techniques result in different TPH concentration values as well as different compositions.

These factors make the use of TPH values in a risk assessment process extremely uncertain.

Another approach, being developed by CONCAWE to enable risk assessment on petroleum substances. Known as the Hydrocarbon Block Method (HBM), it recognises that there are difficulties in trying to apply risk assessment to individual petroleum substances e.g. petrol and diesel, because they are complex mixtures of hydrocarbons. On release to the environment various components of a petroleum substance selectively partition to air, water or soil according to their individual physico-chemical properties (e.g. vapour pressure and solubility) and concentrations are determined according to fate (e.g. biodegradation and photo-oxidation) properties.

In theory, it is possible to determine a PEC and a PNEC for each constituent of a petroleum substance. However, the vast number of hydrocarbons makes this approach impractical and, in any case, the degree of analytical resolution that is achievable for most petroleum substances precludes this. However, since hydrocarbons of similar generic type and molecular weight behave very similarly in terms of environmental distribution and fate, it is possible to group or block such hydrocarbons together. Thus, petroleum substances can be grouped together with those constituent hydrocarbons having similar structure, size, physico-chemical properties, environmental fates and effects, so as to give a manageable series of blocks. For example, petrol, which is made up of more than 150 hydrocarbons, was assessed using the HBM based on 31 blocks and by further simplification, this number could be reduced still further. PNEC values could be assigned for each block and each environmental compartment based on either measured or calculated data. Thus, the PEC/PNEC ratio can be found for each block and these can be aggregated proportionally to derive an overall PEC/PNEC ratio for the whole petroleum substance.

The HBM approach allied to the use of a suitable computer model potentially offers a flexible procedure for the environmental risk assessment of petroleum substances. However, considerable development of the HBM is still necessary before the procedure could be used routinely.

Individual chemical parameters

There are several important reasons for analysing for individual contaminants despite the potential penalties of time and cost. Decisions on whether to remediate groundwaters and to what level (or to leave them alone) are to be based upon an assessment of the potential adverse effects of the specific contaminants on given receptors. A definitive assessment of the potential impact can only be given if the exact contaminant or combination of contaminants is known. Knowing only the concentration of a group parameter could lead to serious under-estimations of potential impacts. For instance if aromatic hydrocarbons were considered as a group parameter, a level of 9 $\mu g\ l^{-1}$ would not necessarily be of concern since it is below the UK drinking water standard of 10 $\mu g\ l^{-1}$ for total hydrocarbons. However, if the total hydrocarbon was comprised entirely of benzene this would approach the WHO health-based drinking water guideline value of 10 $\mu g\ l^{-1}$ (benzene is not specifically regulated in the UK/EC drinking water regulations at present).

Similarly, analysis of PAHs as a group might indicate no apparent concern at a level of 0.1 $\mu g\ l^{-1}$ (which would be below the drinking water regulation total PAH value of 0.2 $\mu g\ l^{-1}$. However, if this concentration were comprised entirely of benzo(a)pyrene, this would exceed the UK drinking water regulation for benzo(a)pyrene which is 0.01 $\mu g\ l^{-1}$.

A similar problem can be envisaged for discharge of groundwater to surface waters by only considering pesticides as a group parameter (i.e. 0.1 $\mu g\ l^{-1}$). The toxicity of pesticides varies greatly and hence a toxicity-based assessment would need to consider individual pesticides. For instance, the EQS for permethrin and dieldrin derived for the protection of freshwater aquatic life are 0.01 $\mu g\ l^{-1}$. Therefore, these compounds would be of concern at concentrations ten times lower than would raise concerns for pesticides in drinking water (0.1 $\mu g\ l^{-1}$).

Summary

The primary benefit of considering group parameters is that the efficiencies are gained in monitoring such parameters. In the process of risk-assessment the emphasis is on reducing the uncertainties in the calculations. This enables any decision, such as deciding upon potentially expensive remediation measures, to be made with the maximum confidence that the decision is based on the best information. Monitoring for individual determinands reduces uncertainty whereas monitoring for group determinands increases uncertainty, since the exact composition, and hence the risk associated with that composition is also not known. Although progress is being made to enable assessment of environmental risk hydrocarbon groups, this approach would be unacceptable for the assessment of drinking water risk. It is therefore considered that the benefits of assessing the risks of exposure to individual parameters out-weigh the limited benefits of considering group parameters.

5.3 Minimum number of samples - a statistical evaluation

Compliance with the standards agreed for clean-up will usually need to be tested statistically and should always be considered in the context of background variations.

Where the objective is related to *mean* concentrations, we may reliably use statistical theory based on the Normal distribution. This is because, thanks to the Central Limit Theorem, the uncertainty in sample means tends rapidly towards Normality as the number of samples increases, however non-Normal the underlying distribution of the individual data values may be. A full statistical treatment is complicated, nevertheless, and the reader should refer to standard texts on sample programme design. Usually, however, the approximate calculations given below will be quite adequate for planning purposes.

Estimating mean quality

Single borehole

The number of random samples needed to determine mean quality at a single borehole depends on three quantities:

- the desired **precision,** D - that is, the tolerable maximum difference between the estimated mean and the true underlying mean;
- the desired **confidence level** (90% in the examples given here); and
- the **variability** of the system or process being sampled - as characterised by an estimate of standard deviation, S (assumed to be reliably known from past data).

Normal theory gives us the following approximate formula for the number of random samples, N, necessary to obtain a future estimate of the mean with precision D at 90% confidence:

$$N = (1.645 \times S/D)^2 \qquad (1)$$

Suppose, for example, that we wish to estimate mean TCE (tetrachloroethylene) - a determinand usually measured in the hundreds of $\mu g\ l^{-1}$. A typical standard deviation might be 8 $\mu g\ l^{-1}$ (i.e. S = 8), and an acceptable target precision for TCE might be 5 $\mu g\ l^{-1}$, (i.e. D = 5). The required number of samples calculated from equation (1) would then be:

$$\begin{aligned} N &= (1.645 \times 8/5)^2 \\ &= 6.9, \text{ which rounds up to 7 samples.} \end{aligned}$$

Group of boreholes

The mean quality for a group of boreholes should be obtained by averaging the individual means. (This avoids possible bias in cases where not all boreholes have been sampled the same number of times.) The task of selecting a sample frequency is similar to the case for a single borehole, except that we have to work with a pooled estimate of standard deviation. Specifically, suppose we are concerned with a group of M boreholes, and that for each individual borehole we have N_i past data values whose standard deviation is S_i. The pooled standard deviation, S, is then given by:

$$S = \sqrt{(\Sigma(N_i - 1)S_i^2 / \Sigma(N_i - 1))} \qquad (2)$$

the summations running from 1 to M.

Having obtained the pooled standard deviation estimate, and selected the desired precision and confidence, we may then use equation (1) as before to calculate the total number of samples required.

For illustration, suppose we are dealing with benzene for a group of three boreholes (i.e. M = 3) for which the standard deviations are 50.0, 60.0 and 30.0 $\mu g\ l^{-1}$, based on 12, 6 and 12 samples respectively. From equation (2), the pooled standard deviation is:

$$\begin{aligned} S &= \sqrt{((11\times2500 + 5\times3600 + 11\times900) / 27)} \\ &= \sqrt{2052} = 45.3. \end{aligned}$$

Suppose now that we wish to achieve a target precision of 20 $\mu g\ l^{-1}$ with 90% confidence. From equation (1),

$$\begin{aligned} N &= (1.645 \times 45.3/20)^2 \\ &= 13.9. \end{aligned}$$

It is sensible for the same number of samples to be taken at each borehole (unless the variability differs markedly between boreholes). Here, therefore, the value of N could be rounded up to 15, giving a requirement of five samples per borehole.

Detecting a change in mean quality

Single borehole

Because of random sampling variability, no two mean values will ever agree exactly (except by a lucky chance) even when both sets of data have been drawn from the same underlying distribution. Thus, the two *observed* means must differ by at least some amount D before we are able to declare with 90% confidence that the *true* underlying means are different.

The amount of sampling needed to detect a change in mean quality at a borehole between one time period and another therefore depends on the same three quantities as before, namely:

- the precision D (now interpreted as being the 'minimum significant difference');
- the confidence level (90%); and
- the variability S.

Specifically, the number of samples, N, needed in each of the two periods is given by:

$$N = 2(1.645 \times S/D)^2 \quad (3)$$

Note how the total number of samples required over the two periods is now four times that given by Equation (1). This reflects the additional complexity introduced when two means, each subject to variability, are involved rather than just one.

Suppose we are looking at TCE, and have determined from historical data that the standard deviation is 50 $\mu g\ l^{-1}$. Our requirement might be that we wish to be 90% confident that if two observed means differ by 20% or more, then a real shift in means has occurred. If the mean concentration has been in the region of 300 $\mu g\ l^{-1}$, then this indicates that we want to declare as statistically significant a difference of 60 $\mu g\ l^{-1}$ in the observed results from each time period, (i.e. D = 60). We would therefore need the following numbers of samples:

$$\begin{aligned} N &= 2(1.645 \times 50/60)^2 \\ &= 3.8, \text{ which rounds up to 4.} \end{aligned}$$

So if we take four random samples in each time period, and the observed means differ by 60 $\mu g\ l^{-1}$ or more, we can be 90% confident that the true mean TCE values for the periods are genuinely different.

Group of boreholes

As with the earlier task of mean estimation, the result for a single borehole readily generalises to the case of a group of boreholes. Again, suppose we are concerned with a group of M boreholes and that for each individual borehole we have N_i historical data values whose standard deviation is S_i. The pooled standard deviation, S, is obtained from equation (2). Then if a difference D in the *observed* group means for two time periods is to signify a genuine difference in the *true* group means, the total number of samples, N, required in each time period is given by Equation (3).

To continue with the earlier example, suppose we are dealing with benzene for a group of three boreholes (i.e. M = 3) for which the standard deviations are 50.0, 60.0 and 30.0 $\mu g\ l^{-1}$, based on 12, 6 and 12 samples respectively. From Equation (2), the pooled standard deviation was found to be 45.3 $\mu g\ l^{-1}$.

Suppose now that we wish an *observed* mean difference of 20 $\mu g\ l^{-1}$ to indicate with 90% confidence that a *real* change has occurred. From equation (3), we have

$$\begin{aligned} N &= 2(1.645 \times 45.3/20)^2 \\ &= 27.8. \end{aligned}$$

Thus, 30 samples would need to be taken in each of the two periods (i.e. ten samples per borehole) to ensure that an observed mean difference of 20 $\mu g\ l^{-1}$ would be statistically significant at the 90% level of confidence.

5.4 Timing and location of samples

The initial, and in some cases subsequent, detailed site evaluation steps of the site investigation programme should have served to establish baseline water quality in the aquifer and characterise the contaminant plume. When standards for clean-up have been agreed, based on either natural attenuation, containment pumping or physical removal of contaminants, a programme of sampling should be established to identify any changes in the baseline water quality upgradient from the site, in the concentrations in the pollutant plume or in the water quality in the compliance borehole. The number of samples required to estimate mean quality to 90% confidence limits and to detect a change in that mean quality have been discussed in the previous section, the frequency or time period over which those samples should be taken must then be fixed.

The time interval between sampling events may be decided rationally if an assessment can be made of:

(a) the probable rate of change in concentration that could occur (R per year); and

(b) the maximum change in concentration (C) that would be acceptable between consecutive sampling surveys.

An appropriate time interval would then be C/R years.

It is essential that natural variability in the aquifer has been assessed in the earlier stages of the investigation (Appendix B) to adequately characterise the hydrogeological situation since this will also influence the required frequency of sampling. The site evaluation should have served to identify a minimum period that remedial or monitoring works will be initially undertaken, this should be taken in association with natural variations and the identification of sample numbers to determine a sensible sampling frequency. For example if statistical analysis of a given case study identifies that only two samples are required, to evaluate a given situation, then a weekly sampling period would obviously be inappropriate to identify any long term changes in quality levels. However if thirty samples are required then a weekly or two weekly frequency could be appropriate, since this will provide data over a period of at least a year. The monitoring frequency should be such that the individual company and Environment Agency are in a position to implement immediate action if a situation arises where there is a deterioration in quality at the monitoring and compliance boreholes and there is a real threat to the resource in general or a specific receptor.

Rational methods for optimising the positions of sampling points for groundwater and pollutant plumes from point sources of contamination are specific to each site and should be discussed and agreed in detail with the regulatory authority. All data should be regularly compiled into a site manual. Periodic review of the data may allow, in consultation with the regulator, a reduction in the frequency and extent of the monitoring programme.

5.5 Data evaluation

Long-term trends in the data are often most easily interpreted and presented in a graphical manner. There is a number of ways that graphical plots can be used to present data. Many of the ideas summarised in this section have been dealt with in the recent Draft Waste Management Paper 26D which deals with landfill (i.e. point source pollution) monitoring. All monitoring data should be carefully collated and stored in a site manual for easy access and interpretation.

Temporal distribution: A plot of determined concentration with time can be used to identify where the pollution plume is becoming stronger or weaker. Given sufficient data it should be possible to predict if the pollution will reach an unacceptable level with respect to the specified clean-up targets. In some instances the conclusion will be evident merely by inspection, but in less clear-cut cases the use of specialist statistical software is advisable.

Triangular plot: The triangular plot shows the relationship between the samples in water analysis. To interpret the data the water quality of the pollutant plume must be clearly established together with 'background' or uncontaminated groundwater quality. If groundwater is sampled upgradient of the polluted site and the pollutant plume is sampled at the site boundary, if a sample of groundwater taken at a monitoring well away from the site is a mixture of the two components, the plots should lie on the line or curves joining the end points.

Lines of equal contamination: Each pollutant determinand is plotted on a map at a position from which the sample is taken. With sufficient monitoring points the groundwater map can be contoured to show lines of equal contamination, the shape of the contours will indicate the extent and direction of the pollution plume, and possibly its rate of movement.

5.6 Completion of compliance monitoring

The progress of a successful remediation programme will be seen as a gradual decrease in the concentration of pollution in groundwater sampled at the compliance borehole. Eventually the clean-up target concentration will be reached and the clean-up or remediation may be considered complete and the active remediation works can be switched off. This may not be the case: it is common to observe a degradation in groundwater quality immediately following the switch-off of active works. Usually this is caused by continued desorption of pollutants from the aquifer matrix into the groundwater.

It is recommended, therefore, that monitoring of the groundwater quality in the compliance borehole should be continued for an extended period (several months) following switch-off of remedial works. If the groundwater quality is seen to degrade to above the clean-up target concentration then the works should be reactivated and continued until the clean-up target is once more achieved. The reactivation of remediation may need to be repeated until the quality of the groundwater in the compliance borehole meets the clean-up target for a prolonged period of monitoring.

Quarterly monitoring should continue until agreement is reached with the Regulator that the site poses no further risk to the resource.

GLOSSARY OF TERMS

Analytical model	Analytical solutions for flow or transport problems
Aquifer	A permeable geological stratum or formation that is capable of both storing and transmitting water in significant amounts.
Attenuation	The loss of properties by a substance as it passes through a medium
Base Flow	The contribution of groundwater to surface water
Borehole	Drilled hole made in soil/rock
Chain of Custody Document	Records movement of a sample from the time of collection to the time it is disposed of.
Clean-up target	The goal of remedial activity set at the compliance borehole in the form of a load or concentration in the groundwater
Compliance borehole	Location where clean-up target must be achieved
Concentration	Amount of chemical per amount of medium in which it exists
c	Clean-up standard at receptor or surrogate borehole
Cs	Clean-up standard concentration (maximum permissible concentration in the groundwater at the compliance borehole
Confined	Where permeable strata are covered by a substantial depth of impermeable strata such that the cover prevents infiltration
Conservative Pollutants	Pollutants which can move through the aquifer with little reaction with the rock matrix and which are unaffected by biodegradation
Controlled Waters	Defined by Water Resources Act 1991 Part III Section 104. They include all groundwaters and inland waters and estuaries
Cyprinid Fish	Coarse fish like roach, dace and bream
Degradable Pollutants	Pollutants which readily break-down
Derogation	Term used for loss of water resources or deterioration in water quality
Diffuse Source	Pollution from widespread activities with no one discrete source
Dispersion	Irregular spreading of solutes due to aquifer heterogeneities at pore-grain scale (mechanical dispersion) or at field scale (macroscopic dispersion)
Drift Deposits	Term used to include all unconsolidated superficial deposits overlying solid rocks.
Fracture/ Fissures	Natural cracks in rocks that enhance rapid water movement

Groundwater	The mass of water in the ground below the vadose zone occupying the total pore space in the rock and moving slowly down the hydraulic gradient where permeability allows
Groundwater Source Protection Zone	The area of land needed to sustain an identified groundwater abstraction by natural recharge
Hydraulic conductivity (permeability)	A coefficient of proportionality describing the rate at which water can move through a permeable medium. The density and kinematic viscosity of the water must be considered when determining hydraulic conductivity.
Hydraulic Gradient	The rate of change of groundwater potential per unit distance along the flow path
Intrusive investigation	Procedures incorporating techniques such as boring and trenching to facilitate the investigation of subsurface conditions.
Leachate	The liquid that has percolated through solid waste and dissolved soluble components
Monitoring	The gathering of data over a period of time
Numerical model	Computer programme based on the numerical approximation of the flow and/or transport equation
Pathway	The route taken by a substance
95-Percentile	A level of water quality, usually a concentration, which is exceeded for 5 percent of the samples.
Perched Water Table	Water level supported by low permeability layer above main water table
Permeability	Measure of the ability to transmit water. Defined as the volume of water passing through 1 m^2 of aquifer under unit hydraulic gradient; m^3/m^2 d or m d^{-1}
Plume (contaminant)	The resultant region of groundwater contamination arising from a source of contamination having been influenced by groundwater flow
Point Source Pollution	Pollution from a discrete source e.g. petrol station
Porosity	Ratio of volume of void space to the total rock volume
Receptor or target	A particular entity on the land, or on other land, which is being or could be harmed by a potential pollutant, or controlled waters which are being or are likely to be polluted.
Source	Point of abstraction of water e.g. well, borehole or spring
Spring	Natural emergence of groundwater at surface
Recharge	Water which percolates downward from the surface into groundwater
Potable Water	Water of suitable quality for drinking water

Porous	Having microscopic pores between the rock grains (not necessarily interconnected)

ACRONYMS AND ABBREVIATIONS

BGS	British Geological Survey
BOD	Biochemical Oxygen Demand
BRGM	Bureau de Recherches Geologiques et Minieres
BSI	British Standards Institution
CBA	Cost Benefit Analysis
CIRIA	Construction Industry Research and Information Association
CLLB	Contaminated Land and Liabilities Branch (DoE)
DoE	Department of the Environment
DO	Dissolved Oxygen
EC	European Community
EPA	Environmental Protection Act
EU	European Union
HMIP	Her Majesty's Inspectorate of Pollution
HMSO	Her Majesty's Stationery Office
GPZ	Groundwater Protection Zones
GQA	General Quality Assessment
HBM	Hydrocarbon Block Method
ID	Internal Diameter
ICRCL	Interdepartmental Committee on the Redevelopment of Contaminated Land
MAC	Maximum Admissible Concentration
MAFF	Ministry of Agriculture Fisheries and Food
MRC	Minimum Required Concentration
NAPLs	Non Aqueous Phase Liquids
NRA	National Rivers Authority

PEC	Predicted Environmental Concentration
PNEC	Predicted No Effect Concentration
SVOC	Semi Volatile Organic Compound
SWQO	Surface Water Quality Objectives
TDI	Tolerable Daily Intake
TPH	Total Petroleum Hydrocarbons
TOC	Total Organic Carbon
QA	Quality Assurance
QC	Quality Control
UK	United Kingdom
uPVC	Ultra Poly Vinyl Chloride
USEPA	United States Environmental Protection Agency
VOC	Volatile Organic Compound
WAMS	Water Archive Management System
WHO	World Health Organisation
WQO	Water Quality Objectives

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