# Water sources

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## Introduction

Source water is potential raw water, ie, it is natural fresh water that could be abstracted and processed for drinking purposes.

The chemical composition of natural fresh water is the end result of rainwater that has fallen on to the land and interacted with the soil, the material in or on the soil, and rocks as it moves down rivers, or into lakes, or percolates underground. Its overall quality is further modified by run-off from various land uses (non-point or diffuse sources) and by discharges (point source). The quality is modified further by biological activity, wind-blown material and evaporation.

The sections in this chapter are aimed at addressing what impacts on the quality of natural fresh waters, and what can be done to identify and limit these impacts, by taking into account recent research findings from New Zealand and abroad.

Half of the chapter discusses groundwater, including compliance issues related to demonstrating bore water security. Bore water security impacts on both bacterial and protozoal compliance. The concept of bore water security was originally developed for the DWSNZ as an alternative approach to monitoring *E. coli* at the rate required for surface water sources.

A summary of the legislation covering natural fresh water is included in this chapter.

Chapter 4 discusses the steps recommended in the selection of raw water sources and appropriate water treatment processes.

Monitoring surface source waters to determine the number of log credits required for protozoal compliance is covered in Chapter 8: Protozoa Compliance, section 8.2.

Chapter 17: Monitoring, section 17.2 discusses some aspects of water sampling and testing.

Rainwater is discussed in Chapter 19: Small and Individual Supplies.

General source water risk management issues are discussed in the MoH Public Health Risk Management Plan Guide PHRMP Ref. S1.1: Surface and Groundwater Sources; also see Chapter 2: Management of Community Supplies.

Source water quality management is discussed in Chapter 4 of AWWA (1990, new edition 2011).

WHO (2003a) is an excellent general text, some of which was used in compiling the *Guidelines for Drinking-water Quality* WHO (2004). The chapter titles are shown in Chapter 4: Selection of Water Source and Treatment, section 4.3.1.

A well-illustrated publication that describes groundwater quality protection very simply was published by the Vermont Department of Environmental Conservation in September 2005.

The AWWA’s fourth edition of their manual on groundwater appeared in 2014.

The USEPA (2008) published a guidance manual related to their groundwater rule, see References.

WHO in 2012 published an excellent book *Animal Waste, Water Quality and Human Health*.

GNS produced a guideline document (Moreau et al 2014) and a technical document (Moreau et al 2014a) to assist in the delineation of protection and capture zones for bores and springs.

The Commonwealth of Australia produced a report ‘Bore integrity, Background review’ in 2014.

## Groundwater

### Description of a groundwater system

Unlike surface water, many of the processes that affect the quality of groundwater occur underground, out of sight, so cannot be observed directly. Our understanding of how a groundwater system works is largely obtained by deduction from indirect observation. The following sections describe the general characteristics of a groundwater system and the processes that can affect bore water quality.

Groundwater comprises about 80–90 percent of the world’s freshwater resources. It is recharged from the surface, predominantly from rainfall, but can also receive leakage from rivers and lakes. Water seeps down through the soil and unsaturated formation until it reaches the water table. At this point it moves more horizontally through pores in sediments and fractures in rock. Aquifers are large areas of formation that act as reservoirs from which groundwater can be abstracted through a bore for supply.

In the DWSNZ, groundwater is considered to be the water contained in the aquifer; bore water is either in the bore or is the water that has left the bore. This distinction is necessary because previously there has been reference to secure groundwater, which led to people talking about secure aquifers. To be called secure, groundwater that has been abstracted from an aquifer through a bore to become drinking-water needs to comply with bore water security criteria 1 and 2 and 3.

Many New Zealand drinking waters originate from groundwater that enters the distribution system without any water treatment. For the water supply to remain safe to drink requires long-term protection of the aquifer, knowledge of the land-use, aquifer characteristics, and diligent day to day and long-term management.

Instead of requiring disinfection for all groundwater systems, the Ground Water Rule (USEPA 2006) establishes a risk-targeted approach to target groundwater systems that are susceptible to faecal contamination. The occurrence of faecal indicators in a drinking water supply is an indication of the potential presence of microbial pathogens that may pose a threat to public health. This rule requires groundwater systems that are at risk of faecal contamination to take corrective action to reduce cases of illnesses and deaths due to exposure to microbial pathogens. A central objective of the GWR is to identify the subset of groundwater sources that are at higher risk of faecal contamination. The primary monitoring tool is the total coliform test. The GWR does not apply to groundwater under direct influence of surface water (GWUDI) which is covered by the Surface Water Treatment Rule (SWTR).

The excellent publication by WHO (2006) reports that in the period between 1971 and 1982 untreated or inadequately treated groundwater in the US was responsible for 51 percent of all waterborne disease outbreaks and 40 percent of all waterborne illness. A recent analysis of public health data in the USA showed little change to the epidemiology of disease outbreaks.

#### Confined and unconfined aquifers

If a layer of relatively saturated impermeable material (an aquitard) overlies an aquifer, the system is known as a confined aquifer. The aquitard acts as a protective layer, often minimising or preventing further vertical movement of contaminants into the aquifer. Aquitards can also reduce the vertical interchange of water between aquifers at different depths. Where an aquitard is patchy or lacking (eg, tapers out) an aquifer may be more vulnerable to contamination from the ground surface ie, no longer confined, or springs can emerge at the ground surface. Springs can be contaminated directly from surface sources, and can act as conduits for contaminants to move down into the underlying groundwater particularly if they dry out during dry periods.

An unconfined aquifer is so called because of the absence of a confining aquitard layer (eg, clay). In contrast to a confined aquifer, it is relatively vulnerable to contamination from the land surface. For the purposes of the *Drinking-water Standards for New Zealand 2005* (revised 2008) (DWSNZ), when planning a drinking-water quality monitoring programme, unconfined groundwater systems less than 10 m deep should be regarded as being probably no safer than surface sources. Bores drawing from unconfined aquifers greater than 10 m deep may possibly be able to demonstrate security, but require more monitoring than if drawn from a confined aquifer.

When a bore is sunk, the drillers should collect substrate samples at different depths for inspection; this is called the bore log. When a bore is installed it is often pump tested for a prolonged period to establish the volume of water that it can supply. Bore logs and pumping test information from observation bores will often show whether an aquifer is confined, particularly adjacent to the bore. However, it doesn’t show how extensive the confining layer is, or whether it offers consistent protection of the aquifer over a wide area. The regional council may have additional data on file that may help to understand the whole aquifer. The confining layer only protects the water from what is happening above it; contamination from the surface nearer the recharge area can still occur.

Knowledge of the water levels in a bore can also indicate whether an aquifer is confined. Note that for DWSNZ purposes, depth is the length of casing to the top of the shallowest screen, not the total depth. Bores that are naturally free-flowing (artesian) are generally indicative of confined aquifer conditions, at least locally. This upward flow of groundwater that provides some natural aquifer protection can, however, be reversed during pumping, droughts, or intermittent use upslope.

USEPA (2008) describes 14 indicators of confinement and the characteristics used to identify the presence of a confining layer.

#### Groundwater flow

By measuring the depths of the water in a number of bores relative to a common datum, eg, seawater level, the depths in the various bores can be contoured to produce a map of the water table (unconfined aquifer) or piezometric surface (confined aquifer). Groundwater generally moves at a much slower rate than surface water. It seeps through the pores of sediments or fractures in rock, down-gradient from areas of high elevations to areas of low elevation. Eventually it discharges to rivers, lakes, the sea, or through springs.

Groundwater flows in the direction of greatest downhill slope or gradient (ie, perpendicular to the equal elevation contours on the water table or piezometric map).

The slope of the water table (i), the effective porosity of the aquifer (n), and the amount of water flowing through the pores (flow volume per unit time, Q, divided by the cross-sectional area through which it moves, A) can be used to determine the average linear velocity of the groundwater, v, using the D’Arcy equation:

v = Q/(nA) = K i/n

where k is the hydraulic conductivity of the aquifer.

The velocity, v, is known as the average linear velocity because it describes the gross flow rate through the aquifer material. Aquifers are not homogeneous but may, for example, consist of lenses of finer material (clay, silts or sands) alternating with coarser materials (gravels), such as in Heretaunga (Hawkes Bay), the Canterbury Plains and Waimea (Nelson). These have built up from braided rivers. Groundwater movement through these systems will be quicker through the coarser material than through the finer material. Consequently contaminants in the groundwater can be transported much faster through parts of the aquifer (up to 50 times) than is indicated by the average linear velocity. In addition, localised flow through the buried channels can deviate significantly from the presumed down-gradient flow direction. Consequently, care must be taken in assuming the rate and direction of the groundwater movement through non-homogeneous aquifers.

Tracer tests may be useful in determining the localised groundwater flow rate and direction. An easily detectable tracer can be introduced into the aquifer through an injection bore and its progress determined directly by measuring its concentration in samples of groundwater from down-gradient bores (or possibly indirectly, by geophysical techniques such as surface resistivity using a salt tracer). Flow direction, velocity, dispersion and attenuation characteristics can be estimated by measuring spatial and temporal variations of tracer concentrations. However, the cost of drilling bores is often high, the data interpretation complex and tracer selection critical. Tracer tests should only be carried out by an experienced hydrogeologist.

The temperature of water in very shallow aquifers (eg, less than about 10–15 m deep) may vary seasonally but deeper groundwater temperature remains relatively constant. This is why water from a bore may seem relatively warm in winter or cool in summer. Water temperature provides a useful means by which to infer river – groundwater interactions and can potentially be applied to reduce the uncertainty of aquifer recharge estimates (ESR 2016).

The effective insulation of deeper groundwater from temperature changes also occurs in respect of contaminants. Contaminants in an aquifer are not flushed from their source in the same manner or as quickly as surface water. Unless contaminants attenuate through die-off (microbial), decay or adsorption, they will be retained and move through the aquifer system, potentially affecting the use of the groundwater along its flowpath and probably for a considerable time.

Most groundwater supplies need to be pumped. Risk management issues related to pumping are discussed in the MoH Public Health Risk Management Plan Guide PHRMP Ref. P4.2: Treatment Processes – Pump Operation.

### The quality of groundwater

Groundwaters are generally of better microbiological quality than surface waters because of the range of mechanisms active under the ground that can attenuate the microbial contaminants initially present in the water. Moreover, changes in microbiological quality that occur are not as large or as rapid as those in surface waters. Although some aspects of the chemical quality of groundwaters may be a concern, these characteristics of the microbiological quality of groundwater often mean they are more preferable source waters than surface waters. However, once a groundwater becomes contaminated by chemicals, it takes a long time before the contamination is flushed out.

Table 3.1, copied from the WHO *Guidelines for Drinking-water Quality* (2004), provides a comparison of the levels of pathogens and indicator organisms found in surface and groundwaters.

The levels of microbial contamination in a particular water source will depend, amongst other things, on the nature of contamination sources in the catchment or recharge zone, and the barriers between these contamination sources and the water source. New Zealand source waters tend to exhibit much lower numbers per litre than appear in Table 3.1.

Tests undertaken over a period of time long enough to show seasonal variation are required to establish the microbial quality of a groundwater source. It is advantageous to consult someone familiar with the groundwater in the area for guidance about the most appropriate time to sample. A good reference on sampling groundwaters is *A Guide to Groundwater Sampling Techniques* by L Sinton, published by the National Water and Soil Conservation Authority as Water and Soil Miscellaneous Publication No. 99. Refer also to Sundaram et al (2009).

Table 3.1: Concentrations of enteric pathogens and indicators in different types of source water

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Pathogen or indicator group, per litre** | **Lakes and reservoirs** | **Impacted rivers and streams** | **Wilderness rivers and streams** | **Groundwater** |
| *Campylobacter* | 20–500 | 90–2500 | 0–1100 | 0–10a |
| *Salmonella* | – | 3–58,000 (3–1000)b | 1– 4 | – |
| *E. coli* (generic) | 10,000–1,000,000 | 30,000–1,000,000 | 6,000–30,000 | 0–1000 |
| Viruses | 1–10 | 30–60 | 0–3 | 0–2 |
| *Cryptosporidium* | 4–290 | 2–480 | 2–240 | 0–1 |
| *Giardia* | 2–30 | 1–470 | 1–2 | 0–1 |

a Should be zero if bore water secure.

b Lower range is a more recent measurement.

Groundwaters are not usually in direct contact with faecal material, as surface waters may be, but rainfall and irrigation provide means by which surface contamination can be carried into the groundwater. In some countries groundwaters have been contaminated by the very bad practice of pumping wastes down disused bores. The vulnerability of aquifers to microbial contamination is increased by (Sinton 2001):

* recharge water coming into contact with microbial contamination
* higher porosity aquifer media, which allow greater penetration and transport of microbes, see section 3.2.4.3
* shallow aquifer depth
* absence of a confining layer
* light overlying soils and porous subsoil strata, which reduce the efficacy of processes removing microbes in these layers.

Close et al (2008) tested multiple shallow bores (4.6 to 15 m deep) in South Canterbury to show the effects of intensive dairying and border-strip irrigation on the leaching of *Campylobacter* and *E. coli* to shallow groundwater. *E. coli* was detected in all bores, ranging from <1 to 2400 MPN/100 mL, the average for all bores was 40 MPN/100 mL, the median 2 MPN/100 mL. The mean level of *E. coli* in the individual bore samples ranged from 6 to 137 MPN/100 mL. Over the three years *Campylobacter* was detected in samples from each of the bores on at least two sampling occasions with levels ranging from 0.6 to .3.1 MPN/L. *Campylobacter* were detected in 16 out of a total of 126 samples, an overall detection rate of 12%. *Campylobacter jejuni* were isolated in 11 samples and the remaining 5 were thermophilic *Campylobacter sp*. other than *C. jejuni* or *C. coli*.

In many areas of the world, aquifers that supply drinking-water are being used faster than they recharge. Not only does this represent a water supply problem, it may also have serious health implications. In coastal areas, aquifers containing potable water can become contaminated with saline water if water is withdrawn faster than it can naturally be replaced. The increasing salinity makes the water unfit for drinking and often also renders it unfit for irrigation. To remedy these problems, some coastal authorities have chosen to recharge aquifers artificially with treated wastewater, using either infiltration or injection. Aquifers may also be recharged passively (intentionally or unintentionally) by septic tanks, wastewater applied to irrigation and other means. Aquifer recharge with treated wastewater is likely to increase in future because it can:

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

If aquifer recharge is haphazard or poorly planned, chemical or microbial contaminants in the water could harm the health of consumers, particularly when reclaimed water is being used. For a full discussion, see WHO (2003b).

The layer of unsaturated soil above the groundwater plays a major role in reducing the numbers of micro-organisms found in groundwaters. Factors affecting the survival of organisms (ie, how rapidly the organisms die off), and those influencing their transport (ie, how quickly they are carried through the unsaturated strata) both affect the levels of microbes reaching the groundwater.

Several studies demonstrate a considerable degree of variability between the inactivation or die-off rates of different groups of pathogens, and between inactivation rates of the same organism in different environments. However, as a general rule, enteric viruses persist longer in soils than bacteria. Among the enteric viruses, hepatitis A virus appears to be the most resistant to inactivation in soil and, in laboratory experiments, shows a lower capacity for adsorption to particle surfaces. The oocysts of *Cryptosporidium* are highly resistant to environmental stress and it has been estimated that they could be detected after 12 months in soil (WHO 2006).

Bacterial survival in soils is improved by (Sinton 2001):

* high soil moisture
* greater penetration into the soil profile
* low temperatures
* low pH values (in the range 3–5)
* high organic matter content
* low numbers of antagonistic soil microflora.

The most important attenuating processes for bacteria in soils are filtration and adsorption (Sinton 2001). The effectiveness of filtration is greatest in soils with low particle size, while some sedimentation can occur in zones where there is virtually no flow of water. Media providing large surface areas, such as clays, improve contaminant adsorption. The adsorption process is enhanced by conditions that minimise electrostatic repulsion between the micro-organism and surfaces to which they might adsorb. Increased levels of dissolved solids in the water assist in suppressing electrostatic repulsion, consequently rainwater, which contains little dissolved material, assists microbes in penetrating further into the ground.

Filtration and sedimentation, which are influenced by the size of the contaminant, are less important in the removal of viruses in soils, because viruses are very much smaller than bacteria. WHO (2006) reports results of a study that set out to determine the extent and penetration of microbial contaminants in the Triassic Sandstone aquifer underlying Birmingham and Nottingham in the United Kingdom. It found:

* sewer leakage-derived microbial contaminants are able to penetrate sandstone aquifers to significant depths (>90 m)
* human enteric viruses, including pathogenic species are widespread in the aquifer
* the species of sewage-derived human enteric viruses in groundwater are found to vary temporally, and in parallel with their predicted prevalence in the population. The dominant types found in March and June 2001 were noroviruses and Coxsackievirus B4 respectively
* particular horizons at depth within the sandstone aquifer were found to be rapidly susceptible to microbial contamination (ie, contaminant distribution is vertically and temporally heterogeneous).

Processes active during transport in groundwater further attenuate the levels of microbes that reach the water table. These mechanisms are similar to those active in removal in soils. However, the increased size of aquifer media, and the resulting larger pore sizes, and the higher water velocities in aquifers than through soils, result in filtration, sedimentation and adsorption being less effective. Transport distances are much greater in aquifers than soils.

The absence of sunlight (with its UV light) in the groundwater environment is an important factor leading to the differences in the rates of microbial inactivation in groundwaters and surface waters.

Organisms can be found at great depths. In karst regions, microbes and invertebrates can be found in caves and other openings 100 metres or more beneath the surface. Bacteria can exist in some groundwater thousands of feet below the land surface. However, invertebrates are typically found within 1 to 10 metres of the surface in consolidated materials, in what is called the hyporheic zone. Within this shallow groundwater zone, many macroscopic invertebrates have been identified. Furthermore, the species richness and community structure of these organisms has been shown to change with alterations in groundwater quality. Therefore, the relative presence or absence of different communities or populations of organisms may reflect the impact of changes in regional groundwater quality. As a result, the organisms living within the shallow groundwater zone can serve as indicators of the quality of the groundwater resource. Macroinvertebrates living in the hyporheic zone, such as oligochaetes, isopods, and ostracods, have evolved special adaptations to survive in a food-, oxygen‑, space-, and light-limited environment (USEPA 1998). Sinton (1984) described macro-invertebrates observed in a polluted Canterbury aquifer.

**Health-significant chemical determinands** may appear in waters from natural sources, as well as human activities. The naturally-occurring chemical determinand that appears most frequently at potentially health significant concentrations (greater than 50 percent MAV) in drinking-water sources in New Zealand is arsenic. Groundwaters in geothermal areas often contain arsenic and boron at concentrations above drinking water MAVs. Arsenic is also detected in groundwaters in other parts of the country, although generally at lower concentrations than in obviously geothermal areas (Nokes and Ritchie 2002). Higher arsenic concentrations are often associated with anaerobic (poorly oxygenated) groundwaters. Although arsenic may appear in association with iron and manganese, the presence of these metals in a groundwater does not imply the presence of arsenic. In some groundwaters, the presence of arsenic is thought to arise from the contaminant being leached from old marine sediments.

**Arsenic** has been observed to vary substantially with season, particularly in shallow bores (Frost et al 1993). Measurements should therefore be undertaken under a range of seasonal conditions. Further, the occurrence of arsenic in groundwaters is not always predictable, and tests for arsenic should be included in the investigation of any new groundwater source.

**Boron** is found in association with arsenic in geothermal areas. It can also appear at high concentrations in the absence of arsenic in some geothermally influenced (hydrothermal) springs, eg, near Auckland; few of these occurrences result in boron exceeding 50 percent of its MAV.

**High nitrate** concentrations occur in drinking-water sources in a number of areas in New Zealand. It has a number of possible sources, all related to human activities, such as: fertiliser application; disposal of wastewater from dairy factory operations; high grazing densities of dairy stock. It can also be found at high concentrations on a localised scale due to on-site waste disposal systems (eg, septic tanks). Ammonia is the commonest nitrogen compound in anaerobic water.

There is typically an increased leaching of nitrate from soils with increased rainfall or rising water table levels. In these cases, the highest nitrate concentrations will be found when the water table is highest, ie, usually in the winter and spring. Landcare Research has produced a National Map of Nitrate Leaching, see[: http://www.landcareresearch.co.nz/science/soils-and-landscapes/ecosystem-services/factsheets](file:///C:/Users/ROdean/AppData/Local/Microsoft/Windows/INetCache/Content.Word/:%20http:/www.landcareresearch.co.nz/science/soils-and-landscapes/ecosystem-services/factsheets).

**Fluoride** is often found overseas as a groundwater contaminant of health significance, but fluoride in excess of 50 percent of its MAV has been found in only three water supplies in New Zealand (Ritchie 2004). Slightly elevated levels of fluoride can found in geothermal areas, and in some geothermally influenced (hydrothermal) waters.

**Pesticides** have been found in a number of vulnerable New Zealand groundwaters (Close and Flintoft 2004; MAF 2006). Pesticides in excess of 50 percent of a MAV have been found in only two drinking-water supplies (Ritchie 2004). Dieldrin was the detected pesticide in both cases. Refer to individual pesticide datasheets for details.

A nationwide study of more than 1000 shallow bores and springs in the USA found that one or more pesticides were detected in more than half of the samples collected. 95 percent of the pesticide detections were at concentrations less than 1 μg/L. The most commonly detected pesticide was atrazine, reflecting both the widespread use of this chemical in agriculture, and its chemical stability and mobility in soil and groundwater. Investigations at pesticide mixing sites in Western Australia detected a range of pesticides in groundwater including atrazine, chlorpyrifos, diazinon, dimethoate, fenamiphos, maldison, aldrin, chlordane, DDT, dieldrin and heptachlor. The most commonly detected pesticide in groundwater was diazinon, which was detected in 63 of the 78 groundwater samples collected at concentrations ranging between 0.1 and 4 μg/L (from WHO 2006).

**Manganese** is a health-significant determinand, but it can also adversely affect the aesthetic properties of water. It is a naturally-occurring determinand, which dissolves into groundwater under oxygen-deficient, low pH conditions. Such conditions often arise in shallow groundwaters as the result of the respiration of microbes in the sub-surface media through which the water passes. During respiration organisms withdraw oxygen from the water and return carbon dioxide as a waste product. The carbon dioxide dissolves to form carbonic acid, thereby depressing the pH of the water and dissolving the manganese, particularly when the water is anaerobic. Some source of organic matter, such as peat, may be associated with the appearance of manganese, as the organic matter provides a source of carbon, which is required as a nutrient by the microbes.

**Iron**, which is of aesthetic but not health importance, is also mobilised from minerals under conditions similar to those that will mobilise manganese, and the two are often found together. The conditions that lead to the appearance of iron and manganese can be very localised. New groundwater sources should therefore be tested for both metals, rather than relying on results from nearby bores as an indicator of likely water quality. Iron and manganese concentrations may also vary significantly with time, particularly in shallow unconfined groundwater.

Complexation of iron or manganese with organic matter also present in the groundwater can inhibit the effectiveness of treatment processes designed to remove these metals from the water.

**Calcium and magnesium** are two major cations that can occur at high concentrations in groundwaters that have been in contact with calcareous rocks, such as limestone (chalk) and marble. These cause water hardness, which can lead to problems of scale formation on hot surfaces, and difficulty in getting soaps to lather.

Bores sited near the coast may undergo varying degrees of seawater intrusion, depending on the level of pumping, phase of the tide, distance to the sea, and the ease with which seawater can intrude into the aquifer. This phenomenon can lead to high concentrations of chloride and sodium, and elevated concentrations of calcium, magnesium, all of which may adversely affect the aesthetic properties of the water. It can be very difficult to reverse the process. High levels of sodium and potassium can also occur if the bore draws from old marine deposits.

### Factors affecting groundwater quality

#### The water source

Because contaminants that may have infectious or toxic properties can remain in the aquifer for a considerable time and affect its use, operators of groundwater-sourced drinking-water supplies should take every precaution to prevent contamination of the aquifer. Water managers should assess the potential for contamination arising from possible sources located in the immediate area of the bore (although some plumes, eg, nitrate plumes, can travel many kilometres). All possible precautions should be taken to protect the water supply from all potential impacts. Examples of sources include: stores of hazardous substances, underground storage tanks such as at petrol stations, effluent discharges, septic tanks, waste ponds, landfills, offal pits, application of pesticides or animal wastes to nearby land. As part of this assessment, advice should be sought to determine the protection and capture zones of supply bores (Moreau 2014, 2014a). This takes into account groundwater flows, drawdown effects and attenuation characteristics, see section 3.2.4. Note that contaminants that are not soluble in water will float on the top or sink to the bottom of the aquifer.

Under the Resource Management Act 1991 regional and unitary authorities have responsibilities for the management of water bodies in their region, including groundwater; water suppliers should be able to obtain information from them about their aquifers. The National Environmental Standard for Sources of Human Drinking Water (NES) is a regulation made under the Resource Management Act (1991) that sets requirements for protecting sources of human drinking water from becoming contaminated. It came into effect on 20 June 2008.

Some groundwater systems receive water from more than one source, and if these are not consistent, water quality may vary. For example, as observed in parts of Canterbury, an aquifer may be fed from a river system and from rainwater. The groundwater quality and composition will vary depending on the preceding and current river flow and rainfall conditions, and may even depend on the state of the river bed, and the land-uses in the recharge area.

In areas without reticulated sewerage, sewage is usually discharged into septic tanks. Micro-organisms in the liquid discharge from septic tanks or other systems are likely to enter shallow unconfined aquifers. Good design and maintenance of septic tanks can reduce the threat of microbiological contamination of aquifers from this source. Sewage discharges have been known to find their way directly into aquifers via abandoned bores or soil-absorption or soakage systems receiving the effluent from septic tanks. New bores should be drilled on higher ground than where the septic tank discharges, and should be cased to sufficient depth to prevent contaminated water mixing with the deeper groundwater. Some metal casings can have a surprisingly short life due to the corrosive effect of the high carbon dioxide content in the subsoil water. Most states in USA require a minimum of 50 feet separation between a bore head and a septic tank discharge, livestock yards, silos, and 250 feet from manure stacks, see section 3.2.4.3.

If the average groundwater travel time from the aquifer fed by a septic tank is (say) 10 years, then it is likely that all pathogens will be inactivated during their residence in the subsurface, and it is likely that the groundwater is not at risk. If the average groundwater travel time is two years, then some groundwater will take a fast path and arrive in one year or less, and other groundwater will take a slower path and arrive in three years or more. Because pathogens remain infectious in the subsurface for a maximum of about one year, the health risk depends on the proportion of groundwater that arrives most rapidly at the bore (USEPA 2008).

Other possible sources of microbiological contamination include seepage from sewers, landfills, and land application of domestic and animal effluent.

One potentially major pathway for contamination often overlooked is the conduit provided by a poorly sealed bore, particularly during a flood or after heavy rain. Contaminants can enter directly down the bore shaft or down the junction between the casing and the soil. To protect the groundwater against this source of contamination it is essential to design and construct the bore head to protect against such contamination from the surface, see Figure 3.2 and section 3.2.4.3). Bores should be secured in this manner regardless of the use made of the groundwater abstracted from them. Groundwater contamination can persist for a long time and affect a large area.

Contamination can also enter the aquifer when deep piles are drilled through the aquitard without due protection, or even a telegraph pole, if the protecting layer were very shallow, somewhat like a puncture. Regional and unitary authorities should be aware of old or disused bores. Old bores that have been shut down need to be located and inspected and the bore head repaired if necessary.

A nearby non-secure bore head is not necessarily part of the water supply itself, however, it is a potential threat to the supply that needs to be managed through the supply’s water safety plan, just as any other potential contamination source in the capture zone would be.

A distinction needs to be made between a threat to the “secure” status of a bore and a threat to the quality of water produced from the bore. Threats to the water quality from secure bores should be managed through the water safety plan, rather than modifying the “secure” status of the bore. Factors that determine the level of risk a non-secure bore presents to a groundwater source include the:

* + - 1. location of non-secure bores with respect to the capture zone. of the neighbouring bores(s) and the number of non-secure bores within the capture zone (Moreau et al 2014/2014a)
      2. amount of contaminant/surface water likely to enter the aquifer
      3. frequency at which this may occur.

Management of the risk associated with threats to bore water quality, which include non-secure bore heads, should be made through water safety plans. DWAs and water suppliers may find the advice below helpful in managing risks to bore water quality.

* + - * 1. Water suppliers need to do their best to establish what threats, including non-secure bore heads, to the quality of their bore water exist. Help from outside agencies, such as regional councils who hold catchment/capture zone information and also have responsibility for source water protection through the RMA and NES for sources of human drinking water, should be sought, including the location of disused bores.
        2. To determine the area in which a threat may affect the quality of the water from their bore, a water supplier needs to delineate the source bore(s) capture zone(s). The GNS report, *Capture Zone Guidelines for New Zealand* (Moreau et al 2014) defines a capture zone as: the total area that contributes water to the bore. It follows from this definition that, in principle, any contaminants from a threat within the capture zone could eventually reach the bore of interest. Because of mechanisms removing or inactivating pathogens, including die-off, during their subsurface transport, the concept of a protection zone may be helpful. The GNS report defines a protection zone as the portion of the zone that has a defined travel time for groundwater to arrive at the bore. This could take account of microbial die-off, although it may be more difficult to delineate if hydrological data are scarce. Where there is uncertainty about defining a protection zone, the water supplier should delineate the capture zone to ensure all possible threats that may affect their source have been identified. The health significance of these threats can then be estimated and the need for risk management strategies determined.
        3. Approaches to delineating capture zones can vary in their level of sophistication. The level of sophistication influences the time and cost of the approach. In selecting an approach to delineation, the water supplier needs to assess the severity of the consequences of the delineation being inaccurate. A simple approach to capture zone delineation (Moreau et al 2014a) should be used as a starting point, as it generally produces the largest capture zone. If there are no threats known within the most simply-defined capture zone, then this approach is probably adequate. However, if the delineated zone contains a threat, or a threat exists close to the margins of the delineated zone, then a more sophisticated approach to zone delineation should be used to obtain a more accurate assessment of the risk.
        4. If the risk that a threat presents, such as a non-secure bore head, cannot be readily assessed (ie, there is no information on the level of contamination that the threat may introduce into the aquifer or how frequently this may occur), then the supplier should carry out monitoring to help make this assessment. The most efficient and effective way of doing this is to focus sample collection on times when the risk is at its greatest, ie, when the pathways by which contaminants are transported into the aquifer are likely to be active, eg, during and following heavy rain. This approach offers the greatest likelihood of determining whether a threat can affect a bore’s water quality. A ‘passive’ approach to sampling, ie sampling regularly at set intervals, has a reduced chance of detecting contamination of the bore. The risks to water sources need to be actively managed.
        5. Bore-construction materials deteriorate with time. As a result, it should not be assumed that once a bore head is classified as secure, it will always be so. For this reason, the security of bore heads, both of secure bores and bores that are not classified as secure within the capture zone, should be checked at appropriate intervals.

Groundwater in coastal areas is prone to high salt levels. This is likely to be caused by seawater intrusion into the aquifer, salt drift, or possibly dissolution of salt deposits. The quality of groundwater supplies subject to seawater intrusion may change in association with pumping, other users, water level variation and tidal cycles. Pattle Delamore (2011) developed national guidelines for managing the risk of seawater intrusion; many examples of management in New Zealand are included.

Geological events such as earthquakes and volcanic eruptions have the potential to change the chemical and microbiological quality of a groundwater. These events can dislocate aquifers so that previously confined aquifers are no longer protected from the influence of surface events, or the source aquifer receives groundwater from other, previously separate aquifers. Fissures, resulting from volcanic eruptions, may also allow the introduction of contaminants into the source aquifer. Rutter (2018) discussed changes observed in groundwaters in Canterbury, Marlborough and Wellington following the Darfield and Kaikoura earthquakes. Groundwater levels at different depths converged after the Darfield earthquake, indicating closer hydraulic connection between aquifer layers.

Infrastructure damage can occur too. See ESR (2012) for a literature review of the impacts of earthquakes on groundwater quality. Note that following the earthquakes in Christchurch, 20 of the 174 deep bores needed to be redrilled and 82 needed repairs. See section 3.2.5. Aqualinc (2011) also issued a report on the impacts of the Canterbury earthquake on groundwater resources.

Daughney and Wall (2007) provided a summary of groundwater quality state and trends in New Zealand based on data collected from 973 sites over the period 1995 to 2008. The dataset includes sites in State of the Environment (SOE) monitoring programmes operated by regional councils and the National Groundwater Monitoring Programme operated by GNS Science. This study is intended to be ongoing.

As the above discussion explains, there are many reasons why the quality or composition of a groundwater source may change. It is recommended that sufficient determinands be tested at an appropriate frequency to confirm that the bore water quality remains as expected. Six monthly testing would normally be sufficient for observing long-term changes. The principle determinands in most bore waters are chloride, alkalinity, silica, sodium, calcium and magnesium. Online monitoring would provide even better information. Determinands that are relatively easy to measure online include conductivity, temperature and turbidity. Bore water from large deep aquifers usually have a very consistent composition. Results should be examined for long-term trends, changes in ratios of determinands.

Short-term variations can be detected by an increase in dissolved oxygen concentration which indicates ingress of surface water into the aquifer. Surface water in the summer will usually be warmer than the groundwater, but colder in the winter, so a change in the seasonal temperature pattern can also indicative of surface water ingress. The appearance of total coliforms must also indicate contamination from the surface.

#### Changes in the unsaturated zone

The composition of the water will probably change during its percolation through the unsaturated zone. Mineral or organic matter may dissolve from the substrates that the water passes through. Groundwater in low rainfall regions is generally harder and more mineralised than water in regions with high annual rainfall that will receive more dilution, although hardness tends to be associated more with limestone areas.

In the unsaturated zone, particulate matter and some micro-organisms may be filtered out, and dissolved constituents adsorbed or absorbed from the water. The effectiveness of the filtration process in making the water biologically safe depends on the characteristics of the substrates. Filtration can be very effective where the soil and rock consists of thick layers of fine particles. Where soils are thin, eg, overlying some gravel aquifers of the Canterbury Plains, or where the unsaturated zone is coarse, or fractured (some volcanic areas of Auckland), filtration can be almost non-existent. In areas where the groundwater table is shallow, the groundwater is unlikely to be microbiologically safe. USEPA (2008) defines karst, fractured bedrock and gravel aquifers as ‘sensitive’; the greater the sand content, the less ‘sensitive’ the aquifer.

#### Changes in the aquifer

The natural variation in groundwater quality must be established through baseline assessment in order to be able to attribute future contaminants to a new development or activity. The length of baseline monitoring and the frequency of sampling must be sufficient to establish the natural variability of groundwater quality in the area. A minimum of two years baseline data is recommended in ANZECC ARMCANZ (2000) but to establish natural variability a longer period may be required in some areas. This baseline groundwater quality data is needed to implement an effective monitoring programme. Trend analysis of individual determinands or suites of contaminants, and periodic mapping of groundwater quality over time, can be useful tools for understanding groundwater quality variations due to either natural processes or anthropogenic causes. NWQMS (2013). Chapter 14 of WHO (2006) is devoted to the assessment of pollution potential.

It is more difficult for contaminants to get into groundwater than into surface water. Once there, however, it is more difficult for the contaminant to be removed. The characteristics of the contaminants, the rate of groundwater flow through the aquifer, and the type of material the aquifer is composed of, will all influence whether the contaminants will attenuate through die-off, decay, adsorption or dispersion.

Once in the aquifer, the quality of the groundwater may change due to its interaction with the matrix or by mixing with other groundwaters. Sediments like sand may act as a filter and remove some types of contaminants from the groundwater as it flows through the aquifer. Fractured or karst (limestone exhibiting dissolution features) aquifers offer little filtration or adsorption of contaminants, but due to a localised higher pH, chemical reactions may occur. In the absence of recontamination, the bacterial quality of the water in an aquifer will usually improve during storage because of die-off due to unfavourable conditions. Passage through aquifer media will also reduce levels of viruses and protozoa, although the rate at which they are inactivated is much slower. WHO (2006) reports that straining has been identified as the principal mechanism for controlling the migration of *Giardia* and *Cryptosporidium* species (*Cryptosporidium* oocysts: 4–6 μm; *Giardia* cysts: 7–14 μm) through fine-grained soil types. Experience has shown that up to 99 percent of *Cryptosporidium* oocysts are retained in the upper layers of the soil. However, the isolation of *Cryptosporidium* and *Giardia* from a small but significant number of groundwater sources in the USA and the UK indicates that the protective effect of the soil layer is frequently evaded, probably by migration through preferential pathways or bypassing; for example, from sewers that are often located below the soil zone.

Simultaneously, the rocks may be releasing minerals into solution or exchanging ions with those in the water.

If iron and manganese are present, the water from the tap may be clear initially while these are in a reduced state but, once dissolved oxygen is present in the water, they can oxidise to coloured forms (generally rusty or black), which may be insoluble and settle out.

Deeper aquifers are more likely to contain higher concentrations of minerals in solution because the water has usually had more time to dissolve the minerals from the surrounding rock material.

Groundwater may contain significant concentrations of naturally occurring radiological determinands, notably radon. Section 9.4 of the DWSNZ requires an initial radiological test of new bores, thereafter testing 10-yearly. See section 11.3 of Chapter 11: Radiological Compliance for further discussion on monitoring requirements, and individual datasheets.

### Establishing the security of a bore water supply

To demonstrate bacterial compliance of water leaving each treatment plant (or entering the distribution system), a water supply serving a population of 10,000 or more needs to be monitored for *E. coli* daily. Section 4.3.2.1 of the DWSNZ allows compliance to be demonstrated without *E. coli* monitoring, subject to certain requirements being met, the main one being a continuous and adequate residual of chlorine. A partial concession has also been made for water supplies using groundwater sources, provided all three bore water compliance criteria are satisfied. An added advantage is that secure bore waters are also considered to satisfy the protozoal compliance criteria.

There is a range of approaches that could be used to assess groundwater vulnerability (ANZECC 1995, and see 2010 review). These include subjective rating systems, statistical and process-based methods. An example of a subjective rating system is DRASTIC (Aller et al 1987). This is a system developed by the US National Water Well Association and the USEPA which rates vulnerability subjectively, based on seven hydrogeologic setting factors. Such a system, while useful as a general indication of potential contamination from the ground surface, is not specific in respect to microbial risk.

An example of a statistical approach is that used for groundwater protection in the Netherlands (Schijven and Hassanizadeh 2002). It uses the Monte Carlo method in uncertainty analysis of factors influencing viral transport. This is used to determine the minimum size of bore head protection zones. Considering the large variation in New Zealand aquifer parameters such application is unlikely to be simple without being overly conservative. Information about viral transport in New Zealand is also limited. Groundwater protection zones are discussed extensively in Chapter 17 of WHO (2006).

Other process approaches such as deterministic models tend to have large data requirements and be situation specific. A useful summary of such methods and their hybrids is provided by Focazio et al (2002). While this is an area of continual development, a pragmatic approach is currently needed.

In the US, a tool, Hydrogeologic Sensitivity Assessment (HSA), is used to determine the pathogen sensitivity of a bore or bore field. The HSA uses all available data to assess the presence of hydrogeologic barriers at public water system bore(s) with detected *E. coli*. The definition of hydrogeologic barrier used is broad and includes any “physical, biologic, or chemical factors, singularly or in combination, that prevent the movement of viable pathogens from a contamination source to a water supply bore”. The HSA approach focuses on identifying recharge pathways and the hydrogeologic barriers that provide natural filtration at specific sites. Then, relative barrier values are selected to represent natural filtration effectiveness for eliminating pathogens for each hydrogeologic barrier. Finally, the barrier values are summed to generate a Barrier Index, which provides a relative measure of the risks of pathogen migration at a specific site. The continuum of pathogen sensitivity is divided into pathogen-sensitive, intermediate pathogen-sensitive and pathogen non-sensitive categories. This approach is described in Ohio EPA (2014).

The current DWSNZ approach using water dating or water quality variation criteria, along with consideration of bore head protection is based on a reasonable assumption that groundwater isolated from surface contamination is unlikely to contain pathogenic micro-organisms. Advantages of the current approach include that it is pragmatic, being easily applied with minimal information and is empirically derived from local information. It has also been supported through public submission.

While the term ‘secure’ bore water is used, it simply relates to a lower level risk of microbial contamination such that less frequent monitoring is justified. It does not indicate that it is ‘secure’ from other forms of contamination or ultimately from all risk of microbial contamination via preferential pathways. In this respect it is a misnomer for which in time an alternative label reflecting the different levels of risk may be introduced.

Sinton (in Rosen and White 2001) collated information from regional councils about microbial contamination of New Zealand aquifers. In general, contamination was reported from bores extracting groundwater from less than 30 m below the ground surface. Septic tank discharges and poor bore construction were implicated in much of the contamination.

Some bore water supplies can never be considered secure. The DWSNZ specifically state that a secure status will not be given to bores drawing from unconfined aquifers when the intake depth is:

* less than 10 m below the surface, or
* 10–30 m below ground surface, and there is less than five years’ monitoring data showing no *E. coli* contamination exists.

Secure bore waters are often those drawn from confined aquifers. However, when the water table in an unconfined area is at a great depth below the land surface, the water may be free from microbiological contamination even though it is unconfined, because of the time it takes for contaminants to move down to that depth through the aquifer materials. However, depth alone does not necessarily lead to freedom from microbiological contamination. Without effective filtration taking place in the soil above and through the aquifer, contaminants may still reach the bore; deep groundwaters in volcanic areas such as Auckland and karst limestone aquifers are examples.

WQRA (2011) gives an example:

A large norovirus outbreak occurred at a newly opened restaurant in Wisconsin, US. The premises had a private bore located in a fractured dolomite rock aquifer. The bore was 85.3 m deep and cased to 51.8 m. It was located 188 m from the septic leach field. Tracer tests using dyes injected into the septic system showed that effluent was travelling from both the septic tanks and infiltration field to the bore in 6 and 15 days, respectively. The private bore and septic system were newly constructed and had conformed to Wisconsin State Code.

Note that the DWSNZ do not define the procedure for demonstrating whether an aquifer is confined. Section 4.5.2.1 of the DWSNZ includes three techniques for demonstrating that the bore water is not directly affected by surface or climatic influences. In effect, this means much the same thing, but see next paragraph. The water supplier must provide sufficient information for an experienced groundwater engineer/hydrogeologist/scientist to be able to make that decision. The sort of data required will include any information already known about the aquifer, geological information gathered during the drilling (bore log), the depth to the screen, full details about the screen and casing, results of pump tests (piezometric survey), etc. Demonstration 3 offers a technique for situations when demonstration 1 and 2 are not feasible. In the absence of some of the above, existing bores may need plumbing (depth sounding), or a CCTV inspection. Ideally, water suppliers should also have information about the recharge area and relevant land uses.

Despite drawing from a confined aquifer, it may still be possible for the water to fail to satisfy the requirements of bore water security criterion 1. This may be because the:

* bore is too close to the recharge area
* confining layer is not extensive enough (wide or deep or intact) to be effective
* main aquifer can receive water from a subsidiary aquifer or other source that is not confined or contains ‘young’ water.

Also, the person stating that the bore is drawing from a confined aquifer may have been wrong, or that conditions have subsequently changed.

To be secure, bore water must meet all three criteria. These show that:

a) activities on the surface or climatic events have no direct influence on the quality of the groundwater (bore water security criterion 1)

b) the bore head provides satisfactory sanitary protection for the bore (bore water security criterion 2)

c) *E. coli* are absent from the water (bore water security criterion 3).

Once security has been demonstrated, the on-going *E. coli* monitoring requirements are reduced substantially, as shown by Table 4.5 in the DWSNZ. Bore water security criteria 1 and 2 need to be checked at least every five years.

The secure status of a bore water supply indicates that microbiological (both bacterial and protozoal) contamination of the water is unlikely. However, it provides no indication of the chemical quality of the water. Groundwaters isolated from surface events become better protected microbiologically as the time since the water entered the ground increases, because of processes such as dilution, filtration, adsorption and die-off. Some chemical determinands may be unaffected by processes that improve the microbiological quality of the water. Processes, such as the dissolution of minerals containing arsenic, will make the chemical quality worse with extended residence times. Consequently, the time the water is under the ground may bear no relationship to the chemical quality of the water.

#### Proving the microbiological quality of the water

*E. coli* is the micro-organism used to indicate the bacterial quality of bore water drawn from a groundwater source. Although the presence of *E. coli* in the water shows that faecal contamination of the water is very likely to have occurred recently, there is no reliable relationship between the concentrations of *E. coli* and protozoa in the water. Assurance about the protozoal quality of the water is obtained by demonstrating that the water quality is not directly influenced by events above ground, see section 3.2.4.2.

A bore water that has achieved secure status must continue to be monitored for *E. coli* at the point where it enters the distribution system, at the frequency stated in Table 4.5 and note 5 (DWSNZ). Samples must be taken prior to any treatment, and *E. coli* must be <1 per 100 mL in any sample (bore water security criterion 3). Note that once the bore water is in the distribution system, it (like drinking water from surface sources) must comply with section 4.4 of the DWSNZ.

The provision of an interim secure status allows a reduced *E. coli* monitoring frequency during the proving period and avoids situations in which a water supply, to comply with the DWSNZ, has to install treatment (primarily for protozoal compliance) during the year in which data have to be gathered to demonstrate security.

The Massey University Protozoa Research Unit has been monitoring 4 non-secure bores for 6.5 years (September 2009–July 2016) for the Ministry of Health. To date 104 samples have been collected; although 11% of the samples have contained *E. coli*, none has contained *Cryptosporidium* or *Giardia* and only one has contained *Campylobacter*.

However, some bore waters are less likely to be able to achieve secure status so their monitoring needs to be more extensive. The following situations apply:

a) **Unconfined aquifers <10 m deep:** If the depth from the surface to the end of the casing/beginning of the screen is <10 m, the bore water is considered to be equivalent to surface water (with respect to both bacterial and protozoal compliance).

b) **Unconfined aquifers >10 m deep:** It is possible for water in an unconfined aquifer >10 m deep to satisfy bore water security criterion 1, in which case, it can also be given ‘interim secure status’. But if bore water security criterion 1 is not, or cannot, be met – refer to (d ii).

c) **Water from confined aquifers:** There is no minimum depth requirement if drawing from a confined aquifer. But in reality, it is difficult to imagine an aquifer that lies <10 m below the surface being able to satisfy the requirements of being ‘confined’. And unless the confining layer is extensive, it is difficult to imagine it consistently satisfying “water younger than one year not being detectable ... etc”. But if the bore water is drawn from a confined aquifer, and satisfies bore water security criterion 1, then it can be given ‘interim secure status’ and be monitored for *E. coli* in order to prove bore water security criterion 3.

If the bore water is drawn from a confined aquifer, but does not or cannot satisfy bore water security criterion 1 – refer to (d ii).

To gain ‘secure status’ also requires bore water security criterion 2 to be satisfied, and a water supplier would be unwise to start *E. coli* monitoring before securing the bore head. The *E. coli* monitoring requirements are defined in Table 4.5 of the DWSNZ. Bores with ‘interim secure status’ serving >10,000 people (for example) require daily testing for three months, and if no *E. coli* is found, monthly testing for the next nine months. If no *E. coli* is found during that 12-month period, bore water security criterion 3 has been satisfied. If Bore water security criteria 1 and 2 are satisfied, the bore water can be called secure, and for the next 12 months, *E. coli* monitoring remains monthly. If still free from *E. coli* after 12 months, monitoring can be reduced to quarterly. Bore waters given ‘interim secure status’ and ‘secure status’ are assumed to satisfy protozoal compliance.

d) **Unconfined aquifers >10 m deep, not complying with (or not tested for) bore water security criterion 1:** There are two situations:

i) 10–30 m deep: If the depth from the surface to the end of the casing/beginning of the screen is between 10–30 m the bore water is to be monitored for (and be free from) *E. coli* for five years before its secure status can be considered. During this time it is considered equivalent to surface water for the purpose of both bacterial and protozoal compliance. Regarding *E. coli* monitoring, these bore waters will require weekly, twice weekly or daily testing (depending on the population served) for the first three months, and if no *E. coli* is found, monthly testing for the next four years nine months. If no *E. coli* is found during that five-year period, bore water security criterion 3 has been satisfied. If bore water security criterion 2 is still satisfied, the bore water can be called secure, and monthly *E. coli* testing continues for the next 12 months; if still free from *E. coli*, monitoring can be reduced to quarterly.

For at least five years, water from these bores will need to be disinfected. Section 5.2.1.1 and Table 5.1a of the DWSNZ explain that these bores will require three protozoal log credits.

As an example, applying UV disinfection at 40 mJ/cm2 with a validated unit should be able to achieve bacterial compliance (section 4.3.5) and protozoal compliance (section 5.16, up to 3 logs). If these bore waters are given secure status they will be assumed to satisfy protozoal compliance; that means the UV disinfection system can be turned off.

ii) >30 m deep: If the depth from the surface to the end of the casing/beginning of the screen is >30 m, and the water from the bore does not comply with bore water security criterion 1, it “must be drawn from a source for which hydrogeological evidence indicates that the bore water is likely to be secure” for bore water security criterion 1 to be satisfied, ie, to be granted interim secure status. Those words were used to cover the situation where the bore log information has been lost (for example), or the age results (demonstration 1) are marginal or confusing, or the chemical composition does not quite meet the requirements of demonstration 2. ‘Hydrogeological evidence’ is not to be confused with ‘hydrogeological model’ which is discussed in demonstration 3 of bore water security criterion 1.

Bores drawing >10 m deep from a confined aquifer, but which do not or cannot satisfy bore water security criterion 1 may be granted interim security by following the procedure in the preceding paragraph.

The hydrogeological evidence referred to above must be provided by suitably experienced, qualified and independent personnel, and must cover all pertinent aspects related to the aquifer and the bore (and any nearby bores drawing from the same aquifer), particularly seasonal effects, piezometric levels and pump tests.

What is hydrogeological evidence? Section 4.5.2.3 of the DWSNZ allows bores >30 m deep drawing from unconfined aquifers to produce evidence (hydrogeological) that the bore water is likely to be secure (ie, not directly affected by surface influences). Bores drawn from an unconfined aquifer could still undergo the age test or the chemical consistency test, ie, section 4.5.2.1. If the bore water satisfies those criteria, that will be good quality supporting evidence. If it ‘just misses’ to satisfy those criteria, that could be helpful supporting evidence. If it ‘misses by miles’ the information may suggest not to bother looking for any further hydrogeological evidence.

If the bore is in limestone (karst) or basalt country there is a risk that surface water will reach considerable depths in quite short time. Likewise for parts of the country like Canterbury where beneath the topsoil there are predominantly gravel, big river stones and boulders. Most old groundwater has lost its dissolved oxygen, so conversely, a bore water with quite a lot of dissolved oxygen (say >4 mg/L) is probably ‘young’, regardless of the depth or whether from a confined aquifer. Very old water should never have *E. coli*, and the total (heterotrophic) plate count is invariably <5 per mL. Bores with no *E. coli*, but with total coliforms, or with TPCs >100 per mL, probably don’t come from an aquifer where the bore is likely to be secure. Likewise, the temperature of a deep secure bore water will hardly change during the year when in use, maybe no more than 0.2°C, ie, often less than the ability of a technician to measure it. Bore water likely to be ‘directly affected by surface influences’ will probably show a distinct and reproducible seasonal pattern of temperature. Water suppliers might say the DWSNZ (or DWAs) don’t ask for these tests. But the DWSNZ are minimum requirements – water suppliers should want to know about the quality of their water. It is highly likely that a water supplier has tested the bore for *E. coli* for some time before approaching a DWA; that information will be an important part of their ‘hydrogeological evidence’ so they should provide it, all of it.

If the hydrogeological evidence does not exist, or does not indicate that the bore is likely to be secure, then the procedure in (i) may be followed, as above. That is, the water will be considered equivalent to surface water and some form of treatment will be required to gain protozoal compliance, for at least five years. Being deeper than 30 m means 2 log credits will be required (DWSNZ Table 5.1a).

It is unlikely that a bore will be granted interim secure status if it draws from an unconfined aquifer, especially if less than 30 m deep. A lot would need to be known about the recharge area and land-use practices will need to be watched diligently.

Section 3.2.4.6 discusses the procedures to be followed if *E. coli* is found.

The interim status lasts for 12 months only, subject to section 3.2.4.6. Sampling over a period of at least a year has the advantage of establishing whether there are any seasonal or climatic trends that would indicate surface influences. During this time, the water supplier must gather data to demonstrate security in full, ie, as well as the criteria met for interim status, the groundwater has to be shown not to be directly influenced by events above ground. If the full set of criteria for showing security cannot be met after the 12-month interim period, the source reverts to a non-secure status, and appropriate treatment processes must be in place and operating satisfactorily for the supply to be able to comply with the DWSNZ.

All *E. coli* samples must be taken upstream of any treatment that is likely to disinfect the supply. The purpose of these tests is to characterise the quality of the raw water, not determine the quality that can be produced after treatment. Conversely, if a secure bore water supply receives any form of treatment or onsite storage that is potentially able to result in microbiological contamination of the water before entering the distribution system, it will need to achieve bacterial compliance as for surface waters, eg, section 4.3 of the DWSNZ.

See Chapter 8: Protozoal Compliance, section 8.2.2 for a discussion on the procedure for determining the protozoal log credit requirement for bore waters.

#### Demonstrating groundwater residence time

Water significantly influenced by surface or climatic conditions is most likely to be water that has only been underground for a short time, or from an aquifer containing a fraction of ‘younger’ water. The opportunity for the levels of disease-causing micro-organisms to be reduced by die-off, dilution, or filtration as the water moves through the ground is much less in new than old groundwaters. Being able to show that there are no signs of above-ground influences therefore provides additional confidence that the acceptable microbiological (including protozoal and viral) quality of the water already shown by the *E. coli* testing will continue.

Bore water security criterion 1 provides two options by which the absence of above-ground influences can be demonstrated:

a) showing directly that the water is not new by isotopic (tritium) and chlorofluorocarbon (CFC) and sulphur hexafluoride (SF6) measurements

b) showing indirectly that the water is unlikely to be new by the low variability of its physicochemical characteristics.

A third option, based on hydrogeological modelling, is only available in the event that difficulties arise with the first two options. These difficulties are discussed below.

##### Determination of the residence time

The mean residence time of groundwater is the average time the water has been underground, from the time it leaves the surface to when it arrives at the point of abstraction. During this time the concentrations of microbiological contaminants will be minimised due to mechanisms including filtration, dispersion and die-off. The DWSNZ require the estimation of the residence time to be made by measurements of tritium and CFC (chlorofluorocarbon) and SF6 (sulphur hexafluoride) concentrations in the water.

Age dating of water yields an average age of the water. Although this is helpful, most groundwaters are mixtures of water with different ages because of the nature of flow in porous materials. What one really wants to know is: what is the fraction of the water with age less than one year? The DWSNZ specify that this fraction must be less than 0.005 percent of the water present in the aquifer. This young fraction can be determined from a series of samplings for tritium, CFCs and SF6, separated in time by several years. Single samplings of tritium, CFCs and SF6 can sometimes be used for less precise estimates of the young fraction, but must be confirmed by future sampling.

Note that a bore water that satisfies the age test may still contain contaminated water if the confining layer is ruptured, or when a nearby bore has a faulty bore head thereby offering a pathway to the aquifer, or from complications caused by earthquakes and flooding, or land-use changes, or increased use of the aquifer.

Tritium is a radioactive isotope of hydrogen, which decays with a half-life of 12.4 years (the time it takes for the number of tritium atoms present in a sample to decrease by 50 percent). It is an ideal tracer for groundwater because it is a component of the water molecule, and the information it provides is unaffected by chemical and microbial processes, or reactions between the groundwater and soil, sediment or aquifer material.

Cosmic rays passing through the atmosphere generate natural background levels of tritium, but during the 1950s and early 1960s large amounts were produced in the atmosphere by thermonuclear tests, see Figure 3.1. Tritium is distributed between the atmosphere and bodies of water such as lakes, rivers and most importantly the oceans, but its concentration changes with time and location. Concentrations of tritium in rainfall reflect the local tropospheric tritium concentrations, and allow the input of tritium into land-based hydrological systems to be assessed. Once rainwater infiltrates into the ground, it is separated from the atmospheric tritium cycle, and its tritium concentration, which is no longer affected by exchange with the atmosphere, decreases by radioactive decay. The tritium concentration in the groundwater therefore depends on the time it has been underground.

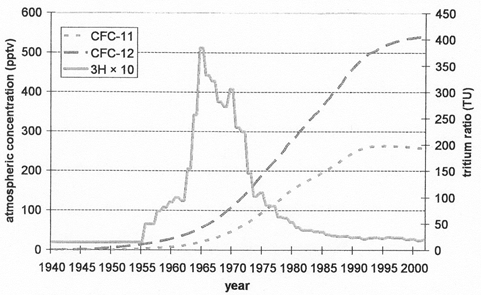
CFCs are a family of entirely man-made contaminants of the atmosphere and hydrological systems. They are used industrially for refrigeration, air conditioning and pressurising aerosol cans, and once released become widely distributed in the atmosphere because of their high chemical stability in this environment. Their use is being increasingly regulated.

Before 1940 CFCs were not present in the atmosphere, but since then their concentration has increased with their increased use, see Figure 3.1. They are slightly soluble in water and are therefore present in recharge waters at a concentration that depends on the temperature of the water and the atmospheric CFC concentration at the time of recharge. Measurement of the CFC concentration in a groundwater therefore allows the time at which the water entered the ground to be determined.

CFCs have given reliable age results for waters in the majority of groundwaters analysed in New Zealand. There are, however, potential sources of error with the measurements. CFC-11 (one member of the CFC family) is more susceptible to degradation than CFC-12 in underground environments that are oxygen-deficient; the water may therefore be calculated to be older than it really is. CFC-12 measurements, on the other hand, are more susceptible than CFC-11 measurements to interference from local sources of contamination. This may make the water appear younger than it is.

The properties of SF6 make it useful in a number of industries (eg, electrical and electronics industries, magnesium manufacturing, refrigeration, air-conditioning, foam production). Because of its chemical stability it has a lifetime in the atmosphere of the order of 3200 years. Atmospheric concentrations before 1970 were zero, but its concentration has increased at approximately 8 percent per annum since then. It is slightly soluble in water and provides another marker for establishing residence time. Its potency as a greenhouse gas is causing its use to decrease and its atmospheric concentration will eventually decrease. Natural occurrences of SF6 have been noted in some deeper aquifers in volcanic areas (Van der Raaij 2003).

Figure 3.1: Atmospheric CFC and tritium concentrations in rain water



##### Detection of new water

A sample of groundwater does not have a discrete age. Mixing processes underground ensure that any sample of water consists of a mix of waters of various ages. Different mathematical models have been developed to account for the distribution of water ages that may arise in a particular situation. Factors such as the nature of the aquifer (confined or unconfined) and the way in which the aquifer is recharged (rainfall or river) influence the form of the model.

The DWSNZ require that the new water fraction (water less than one year old) is less than 0.005 percent, based on a number of assumptions. This criterion has been set as a realistic lower limit for age determinations once accepting likely uncertainty in model fitting parameters and detection limits. Contact GNS to arrange for sampling, which has specific requirements, eg, Daughney et al (2006).

Sources with <0.005 percent of groundwater less than one year old are considered unlikely to be contaminated by disease-causing micro-organisms primarily because of die-off and filtration processes, as well as a high dilution rate. This is supported by monitoring data to date from groundwater supplies in New Zealand. Contamination from ‘young’ groundwater entering an aquifer via leakage through preferential pathways is not accounted for in dating analysis; this makes sanitary bore head protection a very important tool.

The mathematical models used to estimate residence time distribution require two parameters to describe realistic groundwater situations: the mean residence time and the dispersion parameter. The dispersion parameter reflects the degree of mixing and is a measure of the spread of the times different components of the water have been under the ground.

Adjustment of these two parameters to give the best match to the measurements allows the distribution of residence times to be determined. The presence of new water can be estimated from this. Where the analytical results are clear or the hydrogeological situation well-understood, a single dating exercise may be sufficient to allow the mean residence time and the fraction of new water present to be determined.

##### Age calculation

Measured CFC concentrations in groundwater are corrected for excess air and used to calculate corresponding atmospheric concentrations using Henry’s Law and an estimated recharge temperature. The excess air correction and recharge temperature are calculated from the ratio of dissolved nitrogen and argon concentrations, which are measured simultaneously with the CFC concentrations (Heaton and Vogel 1981). The calculated atmospheric concentrations are then used to calculate the CFC model age of the groundwater (Plummer and Busenberg 2000).

To calculate the mean recharge ages and young fraction, GNS uses an exponential piston flow model (Zuber 1986). A conservative estimate or the mixing fraction of 90 percent has been used. This approximates a mainly exponential mixing situation as may occur in an unconfined aquifer. For a confined aquifer, the mixing fraction is likely to be somewhat lower than this estimate. However, in this case, the use of the higher mixing fraction does not affect the calculated young fraction. The mixing fraction estimate may be further refined by additional measurements in two years’ time **OR** the ages have been calculated using a mixing fraction of 70 percent in the aquifer (E(70 percent)PM). The error shown represents the uncertainty in estimating the mixing fraction within the aquifer, and is estimated at ±10 percent. That is, the upper and lower limits of the error on the mean recharge age are given by E(80 percent)PM and E(60 percent)PM respectively.

##### Variability in physicochemical determinands

Water that has been in the ground only a short time is likely to be more variable in quality than older water. The processes of mixing and dispersion that occur as the water and dissolved constituents travel through the groundwater system mean that any variability in the quality of water that has recently entered the ground will tend to decrease with time and distance travelled. Deeper waters therefore tend to be older and less variable in quality than shallower waters, because of the time taken to travel to greater depths. Waters contained in confined aquifers (those overlain by confining layers (aquitards)) also tend to be older than shallow water in unconfined aquifers. In this case, the water has to travel along the aquifer from the recharge zone, which generally takes longer than permeating vertically into a shallow unconfined aquifer.

Variability in the composition of groundwaters may arise for any of six reasons (other than variations caused by a faulty bore or bore head):

1 **Seasonal recharge:** During winter and spring, high rainfall combined with lower evaporation rates and lower plant productivity causes greater leaching of chemicals, such as nitrate, stored in the soil. More new water enters the aquifer from the surface at this time and may therefore show higher concentrations of these chemicals. The dissolved oxygen content will increase too, and probably turbidity.

2 **River recharge:** High river levels during winter and spring may increase the amount of new water feeding into aquifers. Flood events may also increase the recharge with new water, but this may not be related to season. Long dry periods may allow fine silt to fill the pores between stones on the river bed thereby restricting recharge.

3 **Intermittent discharge events:** Activities that contribute to variability in water quality in this category include septic tanks leaking or overflowing during high rainfall, fertilising of pastures, land application of sewage, or chemical spills. Where bore heads are non-secure, floods may also cause spikes of contamination as the result of floodwaters running down the bore casing.

4 **Groundwater abstraction:** Changes in pumping regime in the supply bore or neighbouring bores or new bores can cause changes in groundwater flow directions or leakage rates. As a result, seawater (if near the coast), and water from nearby rivers, or overlying or underlying aquifers may be drawn into the vicinity of the supply bore.

5 **Changes in climate and land use:** Gradual, long-term changes in water quality may arise from changes in climate and land use. Although they may not affect the security of the bore water, because changes may be slow enough to allow removal of micro-organisms, they may have important implications for the future water quality and quantity from the aquifer.

6 **Earthquakes:** Earthquakes can disrupt the confining layers, rupture bores and alter flow paths, see section 3.2.3.1.

Table 3.2 summarises statistics for the conductivity, chloride and nitrate concentration data used to establish the variability criteria contained in Bore water security criterion 1, demonstration 2 (section 4.5.2.1 of the DWSNZ). The criteria are maximum allowable values, and are designed to be conservative. All three determinands must be used. Some supplies not meeting the criteria for secure status on the basis of chemical variation may still be found to be secure using the age determination techniques.

Table 3.2: Statistics showing the variability in conductivity, chloride and nitrate-N for groundwaters

|  |  |  |
| --- | --- | --- |
| **Determinand** | **Statistic** | **%** |
| Conductivity | Co-efficient of variation | 3.0 |
| Chloride | Co-efficient of variation | 4.0 |
| Nitrate-N, mg/L | Standardised variance | 2.5 |

The co-efficient of variation and standardised variance in Table 3.2 are defined as follows:

Co-efficient of variation (standard deviation/mean): 

Standardised variance (standard deviation2/mean): 

where x is a concentration of a determinand from the set of monitoring results.

Use of the coefficient of variation with data near the detection limit can create difficulties because the standard deviation is controlled more by uncertainty in the measurements than true variability in the determinand. This situation often arises with nitrate measurement. To minimise this effect of measurement uncertainty in nitrate measurements, the standardised variance is used as the statistic for nitrate measurements. The value of the standardised variance, however, is dependent on the units in which it is expressed, and nitrate concentrations must therefore be expressed as NO3-N in mg /L when making this calculation for assessing security.

##### Example

Table 3.3 lists two sets of results from measurement of nitrate in samples from two separate bores. The statistics calculated from these are tabulated at the bottom.

The first step in deriving the required statistics is to calculate the mean and the standard deviation for each data set. These functions are available on scientific calculators and in spreadsheets such as Excel®. Although the coefficient of variation is not the required statistic for nitrate, it is included here to show how it is obtained, and for comparison discussed below. The coefficient of variation is calculated by dividing the standard deviation by the mean. The DWSNZ require the statistics to be expressed as percentages, therefore it is multiplied by 100 to give values of 18 percent and 50 percent, for bores 1 and 2 respectively.

Table 3.3: Example of calculations of coefficient of variation and standardised variance

|  |  |  |
| --- | --- | --- |
| **Sample number** | **Nitrate, mg/L as NO3-N** | |
| **Bore 1** | **Bore 2** |
| 1 | 0.24 | 1.6 |
| 2 | 0.21 | 3.8 |
| 3 | 0.15 | 2.1 |
| 4 | 0.15 | 1.7 |
| 5 | 0.21 | 3.1 |
| 6 | 0.15 | 1.0 |
| 7 | 0.22 | 0.9 |
| 8 | 0.21 | 1.6 |
| 9 | 0.15 | 2.3 |
| 10 | 0.18 | 1.1 |
| 11 | 0.21 | 3.2 |
| 12 | 0.15 | 4.2 |
| Statistic |  |  |
| Mean | 0.186 | 2.217 |
| Standard deviation | 0.034 | 1.116 |
| Coefficient of variation | 0.184 | 0.503 |
| % Coefficient of variation | 18% | 50% |
| Standardised variance | 0.006 | 0.562 |
| % Standardised variance | 1% | 56% |

The statistic to use for checking on the acceptable level of variability in nitrate is the standardised variance. This is calculated by squaring the standard deviation, and dividing the result by the mean. Again, it is expressed as a percentage by multiplying by 100. This yields results of 1 percent and 56 percent for bores 1 and 2 respectively.

For the variation in the nitrate concentration to be considered acceptable, the standardised variance expressed as a percentage must be less than 2.5 percent. Bore 1 therefore meets this criterion, but bore 2 does not. Comparison of the coefficients of variation with the standardised variance shows a relatively small difference between these two statistics for bore 2 where the nitrate concentrations are well above the limit of detection (ca 0.05 mg/L as NO3-N). However, in bore 1, where the nitrate concentrations are less than 0.5 mg/L NO3-N and closer to the limit of detection of the method, there is a very large difference between the two statistics.

Some secure bore waters can show almost identical chemical composition year after year, so using chemical consistency can be an effective tool to demonstrate security. This applies in about one third of bores; residence time should otherwise be used. As an example of the complexity when using this approach, many of the groundwaters under Christchurch are considered to be secure, but their chemical composition can vary, depending on the relative proportions of water originating from rainfall or from the Waimakariri River. In situations where there may be difficulties in assessing variability in any of the three determinands at low concentrations, the water supplier should consult with the DWA, who may allow the result to be disregarded for the determinand of concern. Supporting data that show clearly the reason for high variability will have to be provided to the DWA for these data sets to be disregarded.

These samples must be collected with care. Ensure that the water in the sample bottle represents the aquifer, not stale water that has been sitting in the pipe for some time. Sample the bore head, before any treatment, and not from a tank or the distribution system. Refer also to Sundaram et al (2009).

A very competent laboratory is needed when using the chemical consistency technique. For example, as an indication of the likely variation that could be encountered (using somewhat generalised uncertainties!):

* a good laboratory technician measuring chloride in 12 replicate samples (ie, in one batch) should obtain a result in the order of (say) 15 ±1 mg/L
* the same person should achieve something in the order of 15 ±2 mg/L testing these samples in 12 separate batches (see Chapter 17, section 17.5.5)
* but if different technicians did the test each time, the results could be 15 ±3 mg/L
* if different laboratories were hired, the results may be something like 15 ±4 mg/L
* and if these laboratories used different methods of analysis, results could even be 15 ±5 mg/L.

If bore water security criterion 1 has been demonstrated on the basis of the variability of chemical determinands, these determinands must thereafter be measured annually to check that they continue to lie within the range found originally. In the event that they do not, the appropriate statistical parameter should be recalculated with the inclusion of the new data to determine whether the compliance criterion is still met. If the appropriate compliance criterion is not met, the DWA should be consulted, and the cause for the increase in the variability parameter considered. For example, the increase may result from a long-term trend of a change in the determinand concentration. This is not an indication that the groundwater quality is responding directly to events above ground. In this case, it may be appropriate to recalculate the variability parameter using the 12 data collected most recently. Where no significant trend is evident, the increase in the variability parameter may require a reassessment of the secure status of the supply.

##### Approach for demonstrating that a water is not directly affected by surface or climate influences

The following is suggested as an approach for establishing whether a groundwater is directly affected by surface or climate influences.

###### Approach 1: Chemical consistency

* Check whether enough chloride, conductivity or nitrate data are available to determine the variability of all of these determinands, carry out further sampling if necessary, and calculate the statistics.
* To calculate the statistical parameters for any one of these determinands, at least 12 data points are needed. They may be collected monthly over one year, bimonthly over two years or quarterly over three years.
* When collecting samples it is important to ensure that the bore is fully purged of stagnant water by pumping out at least three bore-volumes before the samples are taken.
* Once the data sets are obtained, calculate the appropriate statistical parameters (see Table 3.2 and the related *Example* above). The lack of surface or climate influences is demonstrated if the requirements of section 4.5.2.1 of the DWSNZ are met.
* All determinands must meet the criteria for chemical determinand variability if the source is to be regarded as secure. If this requirement cannot be met, the absence of the influence of surface effects must be shown another way (see Approach 2). As discussed in the section on chemical determinand variability, if a dataset for one determinand cannot meet the requirements of section 4.5.2.1 of the DWSNZ, the results for this determinand may only be disregarded following approval by the DWA.

###### Approach 2: Determine the residence time by tritium and CFC and SF6 analysis

* Samples for these measurements should ideally be taken between winter and spring when groundwater systems are most likely to be recharged with new water. Local personnel can collect samples for tritium measurements, but CFC or SF6 sample collection must be managed carefully due to potential air entrainment. Although some correction can be made for additional air, Institute of Geological and Nuclear Sciences (GNS) staff may be required to take the samples. GNS should be contacted for analytical charges and charges for sampling personnel. The USGS Reston Chlorofluorocarbon laboratory website (<http://water.usgs.gov/lab/>) has a lot of further information.
* Water suppliers should ensure that information provided to the MoH contains the following information:
* a full description of the procedure used to determine the residence time, which includes the mixing model assumptions, the justification for these assumptions and an interpretation of the data
* the percentage of water that has been under the ground for less than one year (rather than an average residence time).
* Further confirmational dating must be carried out if the analyst specifies it to be necessary.
* It is possible that a single measurement is insufficient to determine reliably the percentage of new water in a groundwater. For this reason more than one analysis is recommended. Subsequent residence time determination, eg, after two years, is likely to provide greater confidence, particularly for tritium analysis.

###### Approach 3: Hydrogeological modelling

* Should residence time determination not be considered feasible due to the presence of non-meteoric tracers, and Bore water security criterion 1 cannot be demonstrated by chemical variability criteria, then a hydrogeological model can be used to establish the security of the aquifer.
* For this approach, the water supplier is likely to require the services of a groundwater consultant who can contract to undertake modelling that meets the requirements stated in section 4.5.2.1 of the DWSNZ. The completed modelling must show that contamination by pathogens is very unlikely, to the satisfaction of a person (or persons) deemed (by the Ministry of Health) to be qualified for reviewing the modelling work.
* The results from hydrogeological models are calculated flowpaths and concentrations of determinands, microbiological or chemical, in the groundwater at a particular location and depth. The accuracies of the outputs from the model depend on:
* The appropriate choice of model: A range of models exists, not all of which may be suitable for a given situation. For example, distinctions need to be made between models designed to deal with point source contamination, and those intended for non-point source contamination arising over an area. The number of dimensions that the model takes into account also needs to be considered. Where preferential flow paths exist, these should be accounted for in the model. Knowledge of the geology of the area being modelled is needed to assess whether such flow paths are likely.
* The accuracy of the input parameters: Values for input parameters must be provided for the model; the number of input parameters will depend on the model’s complexity. Some parameters, such as reaction rates and microbial die-off rates may be universal, but others, such as adsorption coefficients and hydraulic conductivity, will depend on the soils, or aquifer media in the area being modelled. This latter type of parameter must be given either a value obtained from laboratory or field measurements, or a conservative value, ie, a value that is more likely to lead to the model producing a non-secure conclusion if the selected value is inaccurate.
* The modeller needs to be able to justify the selection of all input parameters, as the modelling will be have to be verified as being acceptable by a person deemed qualified by the Ministry of Health. It is suggested that proposed models be discussed at the conceptual stage with persons advised by the MoH to ensure applicability.
* Comparison of the predicted concentrations of microbial contaminants in the aquifer at the point of abstraction, with the levels of contaminants permitted for a secure bore water (less than 1 *E. coli* per 100 mL), will indicate whether the source is predicted to be considered secure.
* The model must be run under different scenarios designed to take account of all likely circumstances that would challenge compliance with Bore water security criterion 1. Models can be evaluated, see USGS (2004).
* Model auditing guidelines prepared by PDP (2002) for MfE will be used to provide guidance for model review. A conservative approach will be required given the uncertainties inherent in modelling.

USEPA (2008) discusses (in the appendices) several methods for assessing groundwater time of travel.

#### Establishing adequate bore head protection

See Chapter 18 of WHO (2006) for a full commentary on bore head protection (or sanitary completion as they call it).

A geological log and bore construction details should be available for all supplies being assessed for bore water security. Without this information the aquifer hydrogeologic setting and the suitability of bore design and construction cannot be assessed.

Proper bore construction including bore head protection is an essential requirement for establishing bore water security criterion 2 (DWSNZ section 4.5.2.2). It is also important for the protection of groundwater quality in the aquifer that the bore intercepts. As a minimum requirement, bore construction should comply with the Environmental Standard for drilling of soil and rock (NZS 4411, 2001), unless otherwise agreed by the MoH. Section 2.7 of NZS 4411 covers the decommissioning of bores. Another useful source of information is *Minimum Construction Requirements for Water Bores in Australia* (NUDLC 2012).

Good bore head protection is required for all bores used for drinking-water, not just those demonstrating security. Note that the Walkerton outbreak in Canada was caused by animal wastes entering the groundwater through the bore head. ANZECC (1995) discusses bore head protection plans. The AWWA (USA) has produced a thorough *Standard for Water Wells* (see ANSI/AWWA A100-06, 2006, see full list on <http://www.awwa.org/>).

##### Siting and construction

Assessment of bore design and condition, as well as hydrogeologic setting, should be made by a person deemed suitably qualified and experienced by the Ministry of Health. Sanitary bore head protection should include an effective casing grout seal to prevent contamination from the ground surface. Where there is doubt about bore integrity there are a number of techniques, such as casing pressure tests and down-hole photography, which could be used but are likely to be beyond normal requirement. In general, above ground visual inspection and bore construction data would provide sufficient information. WHO (2006) notes that poorly mixed concrete used for linings and aprons may result in seepage of contaminated water into groundwater sources.

Sanitary inspections should be undertaken frequently, at least as often as water samples are collected. Risks are not static, they change over time as new development occurs in the area and are sometimes due to poor maintenance practices. Certain risks may also be important only seasonally, for instance the collection of surface water uphill of a groundwater source may only occur during wet periods. Therefore inspections may be required in both wet and dry seasons; WHO (2006). All bores have a limited life and therefore thorough periodic reviews are required. A five-yearly assessment of bore head conditions, with any changes reported by the supply manager, should provide appropriate assurance. It is important to develop a protocol for assessment.

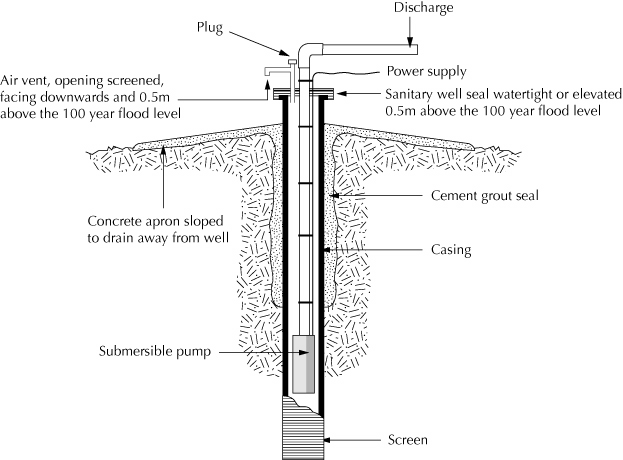
The threat of aquifer contamination may be reduced both by proper bore construction (see Figure 3.2), and by locating bores away from sources of contamination so that the normal movement of groundwater carries contaminants away from the capture zone of the pumped bore. Minimisation of the possibility of contamination from potential local sources such as septic tanks and other waste disposal systems, fertiliser and pesticide stores, underground petrochemical tanks etc, will assist in reducing the threat of contamination. When water is pumped from a bore, the water level drops, causing a vacuum. Without an air vent, suction can develop, which could cause contaminated water to be drawn in.

Septic tanks and similar potential sources of faecal contamination that discharge faecal matter at shallow depths need to be sufficiently distant from the bore head that their discharge will not be captured by the bore. For example (USEPA 2000), 41 states have setback distances (the minimum distance between a source of contamination and a bore) that are less than or equal to 100 feet for sources of microbial contaminants. Five states appear to require setback from all sewage sources of more than 200 feet. Some require 50 feet from a septic tank and 10 feet from a sewer line. Some of the differences relate to the nature of the soil, the depth to the groundwater, and pumping rates.

ECan (2007) presented results of a groundwater modelling study to investigate whether the separation distances for waste discharges in rules in their proposed Natural Resources Regional Plan were adequate to mitigate the risks of virus contamination of water in domestic bores, and to provide a foundation for possible alternative separation distances. Earlier work (ECan 1999) had found that a separation distance of 50 m would achieve a 3 log reduction of *E. coli*. However, viruses can be infectious at much lower concentrations and travel much further in groundwater than bacteria, and they are now considered to represent the highest health risk of contaminants in sewage discharges. A lot of ECan’s proposal was based on New Zealand studies by Pang (2009).

Note that the USEPA (2006) Ground Water Rule recommends a 4-log concentration reduction between drinking water supply bores and contaminant sources. The current study found that much greater distances are required to reduce the number of viruses, see Table 3.4.

Figure 3.2: Sanitary protection of a typical bore



Factors that will influence the required set back distance will be:

* the discharge source characteristics (eg, size of the disposal system)
* the volume and concentration of micro-organisms in the discharge
* the nature of the soil in the infiltration zone
* whether there is a confining layer above the aquifer
* the direction of the groundwater flow with respect to the septic tank and bore head
* the conductivity of the aquifer material, which influences the groundwater velocity
* the depth to the aquifer
* the groundwater pumping regime.

Table 3.4: Log reductions of viruses in Canterbury’s alluvial aquifers

|  |  |  |
| --- | --- | --- |
| **Log reduction in concentration** | **Average separation distance (metres)** | **Standard deviation of separation distance (metres)** |
| 2 | 25 | 27 |
| 3 | 140 | 67 |
| 4 | 389 | 125 |
| 5 | 764 | 227 |
| 6 | 1,186 | 337 |
| 7 | 1,594 | 427 |

ESR (2010) developed the ECan study to National Guidelines. Section 7.1.1 includes:

Where regional plan rules require a separation distance from a bore, these generally require that the discharge from the on-site wastewater system must be separated from a domestic bore by between 20–50 m, depending on the particular regional plan. These separation distances have been imposed generally without substantive scientific basis or specific consideration of the sensitivity of groundwater to contamination at the location of the discharge. Where separation distances have been based on some scientific evidence of contaminant transport, they relate to bacteria.

The development of the Guidelines has shown that, in many instances, bacteria-based separation distances will be insufficient to protect drinking water quality from viruses discharged in domestic wastewater, and that the potential for viruses to be present in groundwater should be recognised in many more situations than is currently the case.

Their Guidelines develop separation distances for different hydrogeological settings and geology; these are summarised in their Table 8.2. Worksheets are included.

##### Backflow prevention

The design and construction of a bore water supply should effectively prevent the ingress of contaminants from the ground surface by using a grout seal. Mixing of hydraulic or aquifer units of different water quality should also be prevented. The bore head should thereby minimise the possibility of contamination of the aquifer from the surface due to backsiphoning, by contaminants passing down the outside of the bore casing due to a poor seal between the casing and the ground, or through cracks in the bore head or casing.

The possibility of backflow of contaminated water from the treatment plant or distribution system down into the bore should also be guarded against by the use of a backflow prevention device. Backflow is defined as a flow that is contrary to the normal intended direction of flow. Increasingly, sludge and/or fertilisers are being applied to pasture by pumping them into the flow of water (which is often groundwater) being used for irrigation; this is a classic situation where contamination by backflow can occur.

One of the requirements in providing a satisfactory bore head is that an effective backflow prevention device is in place; see section 4.5.2.2 of the DWSNZ. The exact nature of the mechanism is not specified in the DWSNZ, as it was appreciated that different circumstances or local requirements may require different approaches to providing this protection. The NZWWA (2006) Code of Practice *Backflow Prevention for Drinking Water Suppliers* describes backflow prevention devices as including reduced pressure backflow devices, double check valves – testable and non-testable, dual check valves, vacuum breakers and air gap separation. The NZWWA Code of Practice has other requirements too, such as: *The water supplier shall ensure that those involved in the specifying, installation and monitoring of backflow devices are appropriately trained to carry out their work*. The NZWWA Code of Practice covers backflow technician qualifications and backflow device certifications. The MoH supports the CoP. The MoH also prepared (in 2001) a PHRMP Guide, D2.4: Backflow Prevention. NZWWA (2006) was revised in 2013. This includes:

The water supplier should ensure that all groundwater takes from an aquifer have adequate backflow protection. The backflow protection programme should require that bores are drilled, constructed and maintained in a manner that avoids any contamination of, or cross-connection with, groundwater aquifers. This should include ensuring that bore head construction on all bores incorporates a boundary device and, where required, a flow measuring device. Groundwater takes for irrigation or stock water with direct injection of chemicals should require, as a minimum, a double check backflow device to protect the aquifer.

Regional councils (see section 2.5.5.8 in NZS 4411, 2001): Environmental standard for drilling of soil and rock) require bores to include some form of backflow protection to prevent contamination of the groundwater system, so the regional council should be contacted first. Once they have explained their requirements, contact an experienced backflow prevention company because backflow prevention can be expensive, and it has to be installed so that the device can be tested, and without damage to the pump, etc.

A side-effect when using a backflow prevention device is the pressure loss it can create. This loss increases depending on whether a single-check valve, dual-check valve, double-check valve, or reduced-pressure backflow preventer is in use. Single-check valves and dual-check valves are not approved backflow prevention devices because they cannot be tested. Double-check valves are available as ‘testable’ and ‘non-testable’. Some regional councils may permit the use of the non-testable double-check valves, particularly if it can be shown the bore will be used in a non-hazardous situation.

The installation of devices that offer greater levels of public health protection, but greater pressure losses, such as reduced-pressure backflow preventers, require regular testing and maintenance.

Single-check valves provide a lower degree of protection. Some regional councils may allow the risk associated with a possible loss of pressure to be reduced by installation of two single-check valves: one at the pump and the second a little downstream of the bore head.

##### Animal control

Control measures are important to protect abstraction facilities against the potential for inundation by contaminated surface water or damage by animals or overland flows caused by heavy rainfall by diverting surface water away from the bore head. Diversion ditches should circle the bore head and drain the water away from the source. Diversion ditches should be located some way from the bore head, but not so far that significant overland flow will be generated within the area between the ditch and the bore head. A general rule of thumb is a minimum of 6 m and preferably 10 m for boreholes and dug wells and up to 20 m for protected springs (WHO 2006).

Restricting access by both humans and animals to the bore head is also important to reduce risks of contamination and thus, where possible, water sources should be enclosed by a fence. The bore heads serving a piped distribution system should be located within a locked building which only the operation staff of the water supplier should have access to (WHO 2006).

The MoH PHRMP guide for abstraction from bores (PHRMP P1.3 – Groundwater Abstraction – bores and bores) recommends that the fence keep stock a minimum of 10 m from the bore head, and certainly no closer than the 5 m stipulated in section 4.5.2.2 of DWSNZ.

#### Multiple bores serving a drinking-water supply

Where a drinking-water supply is sourced from a number of bores, separate monitoring of each bore can lead to a large number of *E. coli* samples having to be taken. The DWSNZ (section 4.5.3) allow water suppliers to reduce the number of monitoring samples taken, if they can show that all bores are receiving water of the same quality, and that contamination of this water is very unlikely. If this is done, the bore considered to be the most vulnerable must be used to represent the bore field.

The water supplier must be able to show that:

* the bores (ie, screens) draw from the same aquifer under similar conditions
* any aquitard protecting the source is continuous across the bore field
* each bore head satisfies Bore water security criterion 2
* the chemical character of the water from each bore is similar
* monthly samples from the individual bores must contain <1 *E. coli* per 100 mL for three consecutive months.

These requirements are designed to give confidence that water quality information gathered from one bore in the field accurately reflects the quality of the water from other bores in the field. Information from several sources will probably be needed to provide this confidence. A number of these are discussed below. A preliminary step is to check that the depth the bore draws from is truly as reported.

##### Stratigraphic data

Stratigraphic information can be obtained from bore logs. These reveal the nature of the various geological strata the bore passes through at that location and the thicknesses of these strata. The nature of the aquifer media in each stratum will allow the permeability of the layer to be evaluated and potential aquitards identified.

For a small bore field, identification of the same aquitard at each bore location will strongly support the assumption that it is continuous throughout the field. However, where the bore field is extensive, and there are substantial distances between locations for which stratigraphic data are available, the presence of the same aquitard in neighbouring bores does not guarantee that there are no breaks in the aquitard in the intervening area.

An indication that there are breaks in the aquitard may be gained from pump tests.

##### Pump tests

Values for the transmissivity (the rate at which water is transmitted through a unit width of an aquifer under a unit hydraulic gradient) and the storage coefficient (the volume of water an aquifer releases from or takes into storage per unit surface areas of the aquifer per unit change in head) define the hydraulic characteristics of a water-bearing formation. Their evaluation at locations across a bore field will help in determining whether the aquifer drawn from at one bore is the same aquifer intercepted by other bores in the area.

Storage coefficient values will help to determine whether the pumped aquifer is confined or unconfined, as the coefficient values in confined aquifers are orders of magnitude smaller than those of unconfined systems, cf 10-5–10-3 (confined) and  
0.01–0.3 (unconfined); the units are dimensionless.

These values can be evaluated by a number of means, but the most reliable is to undertake a pump test at the bore field. Constant-rate and step-drawdown tests are the two most useful forms of pump test. Both require observation bores to be sunk at distances out from the production bore. The reader is referred to texts (eg, Driscoll 1986; Dominco and Schwartz 1998) for more detailed information on the subject of pump tests.

Interpretation of time-drawdown plots, obtained by measuring the levels of drawdown in production and observation bores with time after a pump tests starts, can provide information about additional aquifer characteristics such as potential recharge from nearby rivers or springs and leakage though confining layer/s. Depending on the location of abstraction bores, pump tests may also indicate intersecting hydraulic effects of drawing water from multiple bores.

##### Water chemistry measurements

Microbiological water quality data are of little value in helping to assess whether a number of bores are all drawing from the same aquifer, especially if the water is of generally good microbiological quality and the levels of indicator organisms are below the limit of detection. However, most key chemical constituents of a bore water will be present at detectable concentrations. Comparison of the chemical characteristics and determinand ratios would indicate similarity.

Several different ways of displaying water quality data can be used, eg, tabulation of the data; bar graph; pie chart; Piper diagram; Stiff diagram. The most useful diagrams are probably those developed by Piper (1944) and by Stiff (1951), both of which require the concentrations of the constituents, which are usually expressed in mass per volume, to be expressed in equivalents per litre. Piper diagrams are trilinear and more complicated to interpret, but Stiff diagrams produce a simple pictorial representation of the water quality. Comparison of the shapes of the polygons that result from this form of analysis allow waters from similar hydrogeologic environments to be traced over large areas. The use of cluster analysis and/or principle component analysis may also be useful in illustrating similarities or differences in groundwater chemistry (Wilkinson et al 1992).

Waters that are chemically the same may be from the same aquifer, but marked differences in water chemistry show that either the aquifer is different or the aquifer is being influenced by input from another water-bearing formation.

##### Hydrogeological modelling

Modelling of the aquifer system by groundwater consultants may be undertaken to support rationalisation of monitoring. Modelling is most applicable where the bore field is extensive. See section 3.2.4.2, Approach 3, for some discussion on hydrogeological modelling.

#### Changes in security

The classification of a bore water as secure is not necessarily a permanent status. This is reflected in the on-going checks required within each of the three bore water security criteria specified in the DWSNZ. Signs that a supply may lose its secure status include:

* extreme events, such as floods or droughts, which may affect groundwater quality
* the aquifer structure being altered by a geological event, such as an earthquake
* a breach in the aquitard from developments in the confined area of an aquifer
* a major change in land use
* a large new bore affecting flow patterns or water levels
* corrosion of the bore casing, damage or deterioration of the bore head leading to surface water, or water from a poor-quality aquifer, entering the bore.

Some groundwater supplies may be a mixture of waters from different depths. Droughts, floods, or periods of excessive drawoff may affect the relative contributions from these sources.

When the water supplier is aware of an event that may affect the secure status of the bore water, action should be taken, where possible, to minimise the impact of that event on the water quality.

*E. coli* should not be detected in a supply classed as secure. However, if it is, the actions that need to be taken will depend on the number of samples in which the indicator has been found. The consequences of *E. coli* detection in the water are given in section 4.5.5 of the DWSNZ and are discussed in the next section.

#### Response to *E. coli* detection

Section 4.3.9 and Figure 4.1 of the DWSNZ cover the responses that must be followed when finding *E. coli* in any sample of drinking-water entering the distribution system. For bore water supplies, there are additional requirements.

Section 4.5.5 of the DWSNZ describes the response should *E. coli* be found in bore water. This involves:

* additional *E. coli* testing (confirming bore water security criterion 3)
* checking the chemical consistency (confirming bore water security criterion 1)
* a sanitary inspection of the bore head (confirming bore water security criterion 2).

A secure bore water that only has one sample containing *E. coli* is reclassified as provisionally secure for the following 12 months. That means a faulty result will not require expensive treatment to be installed. If *E. coli* is obtained in another sample during this 12-month provisional period, the water must be reclassified immediately as non-secure. If a secure bore water is classified as provisional more than twice in five years, retention of its secure status will be at the discretion of the DWA.

If bore water that has been given interim secure status (section 4.5.2.3 of DWSNZ) contains *E. coli* in any sample, the 12-month interim sampling regime must recommence. If *E. coli* is found in a second sample during the 12-month interim period, the water must be reclassified immediately as non-secure. Because the bore water no longer complies with the bacterial compliance criteria, it will need to be disinfected. Non-secure bore waters also require treatment to satisfy the protozoa compliance criteria in the DWSNZ.

Section 4.5.5 of the DWSNZ also specifies the actions to be followed if *E. coli* is found in the bore representing multiple bores drawing from the same field.

#### Grandfathering

This section applies to bores that have been in use for some time and have a monitoring history, but whose secure status has not previously been determined, or where the secure status may have been given in error.

Water suppliers may present to the DWA for consideration the full history of a bore’s *E. coli* monitoring and test results, along with all other relevant information. If these results indicate that Bore water security criterion 3 is highly likely to have been satisfied, *E. coli* samples may be collected quarterly for compliance monitoring, as per note 5 to Table 4.5 in the DWSNZ.

To be granted full secure status, bore water security criteria 1 and 2 must have been satisfied in the previous five years.

Some guidance is offered regarding the number and frequency of samples required.

In the normal process, once a bore supply satisfies the requirements for being given **interim secure status**, water from the bore must be monitored at least as follows:

a) weekly/twice weekly/daily depending on population (= 13, 26 or 90 samples in three months)

b) if no *E. coli* found for three months monitoring as in a) – monitoring can be reduced to monthly, regardless of population (= another nine samples in nine months)

c) if no *E. coli* found for nine months monitoring as in b) – bore water criterion 3 is satisfied. If criteria 1 and 2 are satisfied, the bore supply can be called **secure**.

That means an interim secure bore needs 22, 35 or 99 *E. coli*-free samples in 12 months, depending on population, before it can be called secure.

Thereafter, the secure bore water must be monitored for *E. coli* for evermore. For the first 12 months after having been given secure status, the monthly monitoring continues. If no *E. coli* is found during that 12-month period, monitoring can become quarterly.

That means a bore that is given interim security and then becomes secure will need at least 34, 47 or 111 *E. coli*-free samples before being allowed to reduce to quarterly sampling.

For these bore supplies (as per paragraph 1 of this subsection), a DWA could consider accepting three years of monthly, nine years of quarterly, or 18 years of six-monthly *E. coli*-free samples. These three examples require 36 consecutive *E. coli*-free samples.

It is more serious for those bores that would normally need a five-year proving period. Ideally these bore supplies shouldn’t be grandfathered. However, some may have been given secure status incorrectly, so to avoid having to disinfect for five years, an alternative is suggested. Normally, to satisfy Bore water security criterion 3, these bores would need at least 70 *E. coli*-free samples over a five-year period. Therefore based on the previous paragraph, it is suggested that six years of monthly, or 18 years of quarterly, or 36 years of six-monthly *E. coli*-free results may be an acceptable alternative.

### Operations and maintenance

#### The groundwater source

A secure bore water supply has a lower risk of *E. coli* contamination compared with non-secure bores, but it does not mean there is no risk. That is why Bore water security criterion 1 needs to be confirmed five-yearly (annually if using the physicochemical option), bore water security criterion 2 needs to be confirmed annually, and bore water security criterion 3 needs to be met quarterly. That will satisfy the DWSNZ. But there is a difference between meeting the DWSNZ, and managing a water supply.

Despite drawing from a confined aquifer, it may still be possible for bore water to fail to satisfy the requirements of Bore water security criterion 1. This may be because the:

* bore is too close to the recharge area
* confining layer is not extensive enough (wide or deep or intact) to be effective
* main aquifer is receiving water from a subsidiary aquifer that is not confined or contains ‘young’ water
* aquifer may draw in other water due to over-pumping by the bore or nearby bores
* statement that the bore is drawing from a confined aquifer may have been wrong, or that conditions have subsequently changed.

Callander et al (2014) describe how bore water that had been considered to have a very low risk of contamination (equivalent to a secure supply) drawing from a confined aquifer appears to have become contaminated by water from an unconfined section of the aquifer after some excavation de-watering activities and possible sewer leakage in the capture zone. They gave a second example of microbiological contamination of secure bore water following construction of soakage pits 300–400 m upgradient of the affected bores.

These cases illustrate the need for water suppliers to know as much as possible about the aquifers they are using and the recharge (capture) areas. They (and the regional council) also need to assess the likely outcomes of activities in recharge areas and land use changes, just as they would for a river water source.

Because of the possible adverse effects of earthquakes, volcanic eruptions and floods, the microbiological and chemical quality of groundwater after such an event should be checked immediately, and for up to a year after a major event because the effects may take some time before reaching the abstraction point (several months or years). Changes in water level or pressure can also indicate a change in an aquifer.

The regular testing for *E. coli* required by the DWSNZ for secure supplies may identify a change in water quality, but these tests are infrequent and episodes of faecal contamination may be missed. Frequent, regular (or online) monitoring of the temperature and conductivity of the bore water will provide an additional check for changes in water quality. Turbidity monitoring has also proved to be a useful tool for detecting changes. These tests are rapid and inexpensive, and by charting the results changes will become apparent. These should be used as a signal that closer investigation of the security of the bore is needed, and that an increase in testing of the water’s microbiological quality should be undertaken. Regular monitoring of bore water conductivity is a valuable check on the groundwater quality, and significant changes in any bore, secure or not, shows that something has changed and may require attention.

#### Meeting bore water security criterion 1

Secure bore water should not be affected by surface or climatic influences. There are three methods for demonstrating this:

1 residence time (age test) – preferred

2 physicochemical consistency – where the age test produces unclear results

3 verified model – where the above tests are unsuitable.

A sample of groundwater does not have a discrete age. Mixing processes underground ensure that any sample of water consists of a mix of waters of various ages. The age test is pragmatic rather than absolute. Some results may be unclear; see section 3.2.4. In time, the models used for this approach may be refined. Bore waters with unclear age test results may still be granted secure status.

The constant chemical composition test may also produce variable results. This may be because the aquifer **is** affected by surface or climatic influences, or the groundwater composition may vary depending on the relative contributions from river and rainfall; see section 3.2.3.1.

There seems to be two types of secure bores:

a) those that clearly produce old water, are deep and occur beneath an extensive low permeability confining stratum with constant above ground artesian pressure with an upward hydraulic gradient, have extremely consistent chemical composition, unvarying temperature, never contain *E. coli*, have total plate counts <5 per mL, and even more so if the dissolved oxygen content is zero and the nitrate-N content is less than ammonia-N

b) those where the age test does not give clear cut information. These bores often have variable chemical composition, contain some dissolved oxygen, the temperature shows seasonality, nitrate-N is usually higher than ammonia-N, and they are not always particularly deep. Callander et al (2014) stated: “Variable results in groundwater age determinations indicate bores that may have a higher risk of contamination, as do the detections of total coliforms”. The age tests on both bores discussed above by Callander et al were described as producing variable information.

Bores that belong in the second group need additional surveillance. Online monitoring of the temperature and/or conductivity of the bore water will provide an additional check for changes in water quality. Monitoring for total coliforms as well as *E. coli* will give an indication of ‘surface influences’.

#### Commissioning

Generally, new bores undergo a lot of pump testing and maybe screen development. By the time that has finished, the bore should be well and truly flushed out, so the water should be able to be used following a satisfactory *E. coli* result.

Following repairs or maintenance, it is recommended that samples be tested for total coliforms because this test is more sensitive than *E. coli* for indicating whether the bore water is directly affected by surface influences. The amount of flushing required will depend on the reason for the repairs. This should be covered in the Water Safety Plan.

#### Changes due to the bore

Changes in water quality and quantity from a bore can result from changes, notably including chemical incrustation, biofouling and corrosion. Section 3.2.4.3 includes discussion about backflow protection.

a) **Chemical incrustation:** Disturbance of chemical equilibria that control the solubility of compounds in the groundwater can result from drawing water from the aquifer. Substances that have been dissolved but are just on the edge of remaining dissolved can be precipitated.

Precipitation may occur at screen slots where water velocities are high, and disturbance of the equilibria is greatest. This reduces bore capacity, but increases the water velocity through the slot because of its smaller size. Fine particles entrained in the higher velocity water then act to erode the screen. Precipitation may also occur in the aquifer material around the screen, which cements sand grains together and reduces the flow of water into the bore.

The substances most commonly associated with chemical incrustation are calcium carbonate and the insoluble hydroxides of naturally-occurring iron and manganese. Warm groundwaters can deposit silica.

Actions that will help reduce incrustation problems are:

* use of the maximum possible slot area in the screen. This reduces flow velocity through the slots, and therefore the degree to which chemical equilibria are disturbed
* thorough development of the bore
* use of minimum pumping rate (because of the influence of flow velocity on chemical equilibria)
* use of a number of small bores rather than one, or a few, large bores
* frequent maintenance and cleaning of the bore; preventive maintenance is preferable to drastic corrective actions.

Mechanical methods such as wire brushing or scraping, or controlled blasting, can be used to remove incrustation that does develop, but the most effective method is the use of strong acid, such as hydrochloric acid (requiring careful handling and flushing). Ultrasonic cleaners may have a role to play too.

b) **Biofouling and regrowth:** Biofouling of screens most often occurs as the result of infection of the bore by iron bacteria. The organisms mainly responsible for biofouling catalyse the oxidation of soluble iron and manganese in the water, and as a by-product produce slimes containing large amounts of ferric hydroxide. As well as affecting bore yield, this phenomenon degrades water quality with respect to taste, odour and organic matter content (which increases the disinfectant demand of the water).

For biofouling to develop, most micro-organisms need a bore that is open to the atmosphere (for oxygen), sufficient concentrations of iron and/or manganese in the water (a level of iron of less than 0.1 mg/L is usually too low for iron bacteria to survive) as well as dissolved organic matter, and bicarbonate ions or carbon dioxide. Some micro-organisms can extract their oxygen requirements from chemicals such as nitrate and sulphate. The problem seems to be worse with intermittently used bores, or bores that are used seasonally. The end-products of this reduction process include ammonia (which will increase the chlorine demand) and hydrogen sulphide (which makes the water smelly).

Steps to prevent problems with iron or manganese bacteria include:

* disinfection of drill rods, bits, and tools to avoid cross-contamination from previous drilling activities (50–200 mg/L FAC)
* preparation of drilling fluid with chlorinated water (initially 50 mg/L FAC, but a minimum of 10 mg/L FAC must be maintained)
* the bore, once completed, must be sealed to prevent entry of airborne bacteria.

One proprietary system prevents growth of iron bacteria by starving them of the dissolved iron they need. The system works by injecting aerated water, degassed of carbon dioxide, into the aquifer through a field of aeration bores surrounding the production bore. The increased oxygen level in this water assists naturally-occurring bacteria that oxidise iron and manganese to remove these metals from the water by precipitating their oxidised form. Plugging of the surrounding aquifer material does occur, but at a much lower rate than would result from biofouling.

As well as iron and manganese bacteria, bacteria that are released into the water can give rise to positive total coliforms results. Regrowth can result from bacteria that live in the aquifer attaching to the screen, the surface of the casing or fittings, or even the sample tap. Regrowth can also result from surface contamination or subsoil water at the bore head.

Treatment of the water with a chemical disinfectant, such as chlorine, is the best approach to reducing biofouling and regrowth once it has occurred. Details of the procedure can be found in Driscoll (1986), AWWA (2013) and the Centers of Disease Control and Prevention (see <http://www.cdc.gov/healthywater/emergency/safe_water/wells/> – accessed in 2010). AWWA (2004) discusses the biology, ecology, identification and control strategies.

c) **Corrosion:** Corrosion of bore materials is a consequence of the chemistry of the water (either side of the bore) and can result in pinhole corrosion or can affect broad areas of the material. This may allow ingress of lower quality water. The rate at which it occurs generally increases with increasing concentrations of constituents such as carbon dioxide, oxygen, hydrogen sulphide, chloride, sulphate and the total dissolved solids content of the water. Acidic pH levels (ie, lower levels) are more corrosive to most materials than an alkaline pH. Corrosion of steel pipes has been known to occur during storage, even before becoming part of the bore.

The main types of casing used in bores are:

* steel
* uPVC
* ABS
* FRP and FRE
* stainless steel.

Each of these has different properties in relation to column, collapse and tensile strengths, resistance to corrosion, reaction to ground and water chemistry, and temperature. For corrosive water, PVC-U, ABS, FRP, or stainless steel casing provides the longest life possible. Because of the many and sometimes conflicting factors involved in selecting the most suitable casing material, the driller should consult with the manufacturer/supplier and bore owner before selecting the casing (NUDLC 2012).

Electrochemical corrosion can also affect bores (Driscoll 1986). This type of corrosion results from differences in electrical potential on metal surfaces. The potential difference may arise between two different metals, or on two different areas on the same metal surface. To minimise the likelihood of failure of the bore from electrochemical corrosion, care is needed in selection of the materials used, contacts between different materials, and factors that may lead to potential differences on the surface of the same metal, such as heat-affected areas near welded joints, heated areas near torch-cut slots, work-hardened areas around machine cut slots, exposed threads, and breaks in surface coatings (eg, paint).

Changes in the bore that can result from corrosion are:

* the development of sand pumping as the result of the enlargement of screen slots or the development of holes in the casing
* structural failure of the bore screen or casing because of their reduced strength
* reduced yield because of the blocking screen slots by corrosion products
* ingress of poor quality water through corrosion of the casing
* high iron levels in the water.

The ingress of poor quality water is the result of greatest concern with regard to the safety of the water. Perforation of the casing has the potential to allow water from shallow unconfined aquifers, carrying water of unsatisfactory microbiological quality, into the bore water. A potentially secure groundwater source may, through this mechanism, become unsafe without the water supplier being immediately aware of the degradation in water quality.

Corrosion and its control are complex subjects, and expert advice is valuable in ensuring the bore is constructed in a way that will minimise corrosion, and in assessing the steps necessary to minimise corrosion after bore construction. Some discussion on corrosion processes appears in Chapter 10: Chemical Compliance, section 10.3.4.

d) **Permeation:** Some plastic materials used for casings allow low molecular weight compounds such as solvents and petroleum products to permeate into the water; metallic casing may be needed if a bore casing is to pass through such a contaminated area (AWWA 1997).

#### Maintenance

Once a secure bore head has been commissioned, its security needs to be maintained. The status of the bore head should be reported to the DWA annually. The information required includes the adequacy of the surface seals and/or caps, integrity of materials, observations about any contaminant sources located in the immediate area of the bore (eg, storage of hazardous compounds, stormwater or effluent discharges, the application of pesticides, or animal wastes near the bore), and anything that may be important like a change in pressure or flow.

In addition, whenever a bore water sample is taken for testing, the integrity of the bore head should be checked and recorded. Any problems (eg, vandalism, earthquake or other physical damage or change) should be reported immediately, followed by additional sampling to see if the water quality has been affected. Where possible, steps should be taken to protect the bore head from accidental damage. For example, protective barriers (for physical protection rather protection against faecal contamination) should be placed around bore heads located where they may suffer damage by vehicles. This step is designed primarily to ensure security of supply.

There are several reasons for a bore to be closed down for maintenance, such as camera inspections, repairing or replacing the pump, screen or casing, extending the depth, or repairs following an earthquake or flood. As discussed in Chapter 16 for mains repairs, procedures for these various activities should be covered in the Water Safety Plan. It is recommended that the Water Safety Plan should also include samples to be tested for total coliforms as well as *E. coli,* to aid in determining any changes in the risk to the water supply.

Where there is doubt about bore integrity there are a number of techniques, such as casing pressure tests and down-hole photography, which could be used but are likely to be beyond normal requirement. In general, above ground visual inspection and bore construction data would provide sufficient information.

Secure bore water simply means lower risk bore water (but not “no risk”). Bore water may go very long periods without *E. coli* being detected in samples. However, for various reasons, *E. coli* may be found in consecutive samples. It may be wise to disinfect the bore. One technique is to insert a narrow diameter pipe down the borehole and then to pump in a strong chlorine solution in an attempt to kill off any contamination. Alternatively, the strong chlorine solution can be poured down the bore. If the contamination persists after this, a permanent disinfection system will be needed. See WHO (2011a) and the Centers of Disease Control and Prevention (US Government) web page for a helpful procedure. (<http://www.cdc.gov/healthywater/emergency/safe_water/wells/> – accessed in 2010).

Checks on water quality should also be made after other events causing physical damage to bore heads, such as damage by a vehicle or vandalism. WHO (2011a) includes a Technical Note covering cleaning and disinfecting bores.

## Surface water

Surface freshwaters (rivers, streams, lakes and impoundments) comprise those natural waters that are open to the atmosphere and contain only relatively small quantities of dissolved materials; generally (in New Zealand) much less than 1000 mg/L (Harding et al 2004).

Section 3.2 discusses groundwater. The DWSNZ treat springs as surface water, so they are discussed in this section. Rainwater (usually roof water) is discussed in Chapter 19: Small and Individual Supplies.

The convenience of having readily available and accessible sources of water rapidly renewed by rainfall is offset somewhat by the susceptibility of surface waters to pollution from a variety of diffuse and point sources. Point sources are clearly identifiable, have specific locations, and are typically pipes and drains discharging wastes (Davies-Colley and Wilcock 2004).

In most catchments used for water supply, pollution will be from diffuse sources, arising from land-use activities (urban and rural) that are dispersed across a catchment (Novotny 2003). Diffuse sources include surface runoff, as well as subsurface drainage, resulting from activities on land. The main categories of diffuse pollutants are sediment, nutrients and pathogenic micro-organisms. Other categories of diffuse pollutants are heavy metals (principally from urban land) and pesticides (mainly from agriculture and horticulture). Water UK (2012) summarises helpful ideas for catchment protection.

The Parliamentary Commissioner for the Environment (PCE 2012) published an excellent report that describes in a simple, readable manner, the relationship between land use and water quality in New Zealand. Landcare Research has developed the New Zealand Forest and Agriculture Regional Model (NZ-FARM), see: <http://www.landcareresearch.co.nz>

A summary of human activities that impinge on the suitability of freshwaters for potable water is given in Table 3.5. Note that birds may be a significant source of faecal pollution in surface waters as indicated by standard faecal indicators (eg, *E. coli*), and shed pathogens (eg, *Giardia* cysts, *Salmonellae* and *Campylobacter*) (McBride et al 2002).

Table 3.5: Human activities and associated inputs into freshwater ecosystems with human health risks

|  |  |  |
| --- | --- | --- |
| **Activity** | **Contaminants** | **Health risks** |
| Agriculture and horticulture | Sediments  Nutrients  Pesticides and other toxic chemicals and metals  Faecal microbial contaminants | Immune and endocrine disruption  Retarded physical and cognitive development, blue baby syndrome  Foetal malformation and death  Nervous system and reproductive dysfunction  Behavioural changes  Cancers  Waterborne disease |
| Industry | Nutrients  Toxic chemicals and metals  Oils |
| Mining | Sediments  Toxic chemicals and metals |
| Urbanisation, infrastructure and development | Sediment  Pesticides and other toxic chemicals and metals  Oils  Faecal microbial contaminants |
| Recreation | Oils and fuel  Toxic chemicals |

Modified after Slaney and Weinstein 2004.

### Rivers and streams

About half of New Zealand’s drinking-waters are pumped from the ground, with the remainder coming from surface sources (MoH 2003). Flowing waters (rivers and streams) are thus an important source of supply in New Zealand and there is a need to ensure that adequate quantity and quality is maintained in order to provide a reliable and safe source.

Water quantity

It is advantageous to have a good understanding of the flow (or hydrologic) regime of a river selected for water supply. The regime defines the character of a river, how liable it is to floods or to have long periods of low flow, and whether it is useful for the purpose of water supply, and whether impoundment is necessary to provide the volumes required (Duncan and Woods 2004). The continuous time-series record of river flow (hydrograph) can be analysed to estimate the incidence of extreme flows as well as the response of flows to rainfall events.

During low flows there may simply not be enough water for supply, and intakes may even be above low water levels.

During flood flows, water quality is often poor due to sediment discharged in high concentrations, along with other contaminants, notably faecal micro-organisms, and coarse floating material (logs, etc) that are potentially damaging to intakes are often present.

Routine monitoring of New Zealand river flows began between 1900 and 1930 for hydroelectric power generation, and from 1930 for designing flood protection works (Pearson and Henderson 2004). The National Hydrometric Network (Pearson 1998) is a key source of New Zealand stream and river flow data, and is complemented by monitoring networks operated by regional and district councils (Pearson and Henderson 2004).

Flow extremes, such as the frequency of floods of a given return period, or the mean annual seven-day low flow, are useful in this respect and have been summarised for many New Zealand rivers (eg, Hutchinson 1990).

Flow regimes are perhaps most simply linked to the flow-duration curve, a cumulative frequency plot of flows that shows the proportion of time during which flow is equal to or greater than given magnitudes, regardless of chronological order. The overall slope of flow-duration curves indicates the flow variability of rivers. Clearly rivers that have fairly steady flow (eg, owing to spring or lake sources) are preferable for supply to highly flow-variable (flashy) rivers. Flow-duration curves at the extremes are often fitted to analytical distribution functions for the purpose of analysing risk of floods and low flows. For example, annual seven-day low flows are often well-fitted by a log-normal distribution (Pearson and Henderson 2004). Flow regimes are affected by climatic cycles, notably the El Niño-Southern Oscillation (ENSO), with stronger westerly winds and more rainfall in the south and west during El Niño periods and less rainfall in the south and west and more in the northeast during La Niña periods (Scarsbrook et al 2003).

Water quality

Water that has not yet been treated and is to be used for domestic supply is referred to as raw water, in contrast with treated water that has passed through some form of treatment (eg, filtration, disinfection). Drinking-water standards and guidelines mostly apply to treated waters.

There are few chemical constituents of water that can cause health problems from a single exposure, except through massive accidental contamination of a drinking-water supply (WHO 2004). Where short-term exposure to a contaminant does not lead to health impairment, it is often most effective to focus remedial action on finding and eliminating the source of contamination, rather than on treating the water for removal of the particular chemical constituent (WHO 2004).

Most chemicals posing a health risk are of concern only when long-term exposure occurs at concentrations above the MAV, and where treatment to remove the chemical is not employed (eg, Table 3.6). At times when flows and velocities are low, dissolved oxygen in small streams may be very low because of sediment oxygen demand and insufficient reaeration (Wilcock and Croker 2004). At such times appreciable concentrations of soluble, reduced forms of iron and manganese may be released from anoxic sediments or from groundwater inflows which, on contact with air, readily convert to insoluble oxide precipitates that have to be removed during water treatment because they impart unpleasant metallic flavour to water and deposit reddish-brown (iron) or black (manganese) stains.

Table 3.6: Some chemical constituents in untreated surface water used for drinking-water supply that present a potential problem

|  |  |
| --- | --- |
| **Constituent of concern** | **Associated problem** |
| Arsenic (As) | Cancer, skin lesions |
| Fluoride (F–) | Mottling of bones and teeth, fluorosis |
| Nitrate and nitrite (NO3– + NO2–) | Methaemoglobinaemia for bottle-fed infants. Note that this has been disputed recently, see Lundberg et al (2004) and Addiscott et al (2004) |
| Dissolved organic carbon | Trihalomethanes produced by chlorination may be toxic, carcinogenic |
| Iron (Fe), manganese (Mn) | Unpleasant taste, discoloration caused by oxide precipitates |

Other water quality chemical variables are important with regard to operational requirements when water is treated for supply. These include: pH (a measure of the aggressiveness of water with respect to corrosion), alkalinity (capacity to buffer natural waters against pH change), total hardness (mainly divalent ions like Ca2+ and Mg2+ that may be prone to forming precipitates), humic substances that impart undesirable colour to water and can influence corrosion of copper, and total dissolved solids (affects palatability when greater than about 1000 mg/L).

Typical New Zealand rivers and stream waters can be described as being dilute (low total dissolved solids), soft (having low concentrations of Ca2+ and Mg2+), with slightly acidic to slightly-alkaline pH, and weak buffering (low-moderate alkalinity). They may be broadly described as calcium-sodium-chloride-bicarbonate waters (Close and Davies-Colley 1990). There are some notable exceptions to this where, for example, pH is low and bicarbonate alkalinity is high, or total hardness exceeds 100 mg/L as CaCO3. These are generally well-documented through the regular surveillance programme of surface waters used for drinking-water supplies, operated by the Ministry of Health (MoH).

Microbial pollutants are generally of greater relevance to New Zealand surface waters than chemical pollutants. Most microbiological agents of disease (pathogens) are derived from the faeces of warm-blooded animals including humans. The presence of pathogens in waters is sporadic, only occurring when waters are polluted by faecal matter from sick individuals or carriers.

The Freshwater Microbiology Research Programme (McBride et al 2002) involved the monitoring of 22 river and three lake sites for a suite of pathogen and indicator organisms, fortnightly for 15 months (1998–2000). Of the 25 sites, five were source waters for treatment as community drinking-water supplies, of which three were also recreational sites. Pathogenic viruses and *Campylobacter* were detected at least once at all sites and there was little difference between the drinking-water supply sites and the other sites with respect to the occurrence of pathogens and concentrations of faecal indicator organisms.

The main issue for source waters was the high proportion of samples that contained *Campylobacter* (60 percent) and viruses (54 percent) and the ability of drinking-water treatment to kill (or inactivate) or remove them (McBride et al 2002).

Routine testing for pathogens is seldom conducted because a wide range of pathogens might conceivably be present, but the tests are expensive at the required detection levels so testing for several pathogens in each sample quickly becomes prohibitive. Instead, microbiological indicators of faecal pollution, such as the bacterium *Escherichia coli* (*E. coli*) that is ubiquitous in faecal matter, are used also as indicators of disease risk. See Chapter 5.

Lowland streams in New Zealand continue to receive discharges from community sewage schemes, farm oxidation ponds and other point sources (NZWWA 1998; Wilcock et al 1999), but diffuse sources of faecal pollution are now generally dominant.

Faecal contamination of streams can be very high during floods owing to mobilisation of contaminated sediments and wash-in from contributing land areas of catchments. For example, *E. coli* concentrations of 41,000 MPN/100 mL were measured in a flood event in an agricultural stream, compared with a pre-flood level of about 100 MPN/100 mL (Nagels et al 2002).

Diffuse faecal microbial pollution from pastoral agriculture may come from runoff (eg, from farm raceways), livestock accessing unfenced streams, and cattle crossings of streams (Davies-Colley et al 2004). Thus, it is important that key land uses within the catchment of rivers being used for water supply are known so that implications for water treatment are understood. Some median concentrations reported for New Zealand streams and rivers are shown in Table 3.7. Sediments are the main reservoir of faecal contamination with concentrations of *E. coli* approximately 1000 times baseflow levels (Muirhead et al 2004). These sediment stores are mobilised by storm-flows that may have much higher (100 times) *E. coli* concentrations than base flows (Nagels et al 2002). See also section 8.2.1 of Chapter 8 for further monitoring results.

Table 3.7: Faecal contamination in a range of New Zealand streams and rivers

|  |  |  |  |
| --- | --- | --- | --- |
| **Region** | **Land use** | **Median *E. coli* (number per 100 mL)1** | **Reference** |
| Whanganui catchment tributaries (steep hill country) | 100% pasture | 8302 | Davies-Colley & Stroud (1995) |
| Waikato hill-country | Pasture  Native forest  Pine forest | 400  100  83 | Donnison et al (2004) |
| Waikato lowland | Dairy | 280–440 | Davies-Colley and Nagels (2002) |
| Westland lowland | Dairy  Native forest | 60–1000  4 | Ibid |
| Low-elevation rivers throughout New Zealand | Pasture | 700 | Larned et al (2004) |

1 Note that these data are from fixed-interval sampling and are generally taken at low-flows. Flood-flow concentrations may be 100-fold higher.

2 Based on a median faecal coliform concentration of 920 cfu/100 mL, assuming about 90 percent *E. coli.*

### Lakes and reservoirs

Lakes and reservoirs are used to store water during runoff periods for use during other times of the year. The water in the reservoir is used to supply the needs of municipalities, industrial users and the farming community, and can also be used to protect aquatic life by maintaining a continual flow in the stream downstream of the reservoir.

Issues related to land-use and lake management were covered in the *Lake Managers’ Handbook* which was published by the Ministry of Works in 1987 and is still widely used and highly regarded by lake managers and others involved in water management. Aspects of this publication were updated by MfE in 2002. NIWA produced *Guidelines for Artificial Lakes* in 2012. The Guidelines provide a range of management strategies required to produce the highest possible lake water quality in a new lake and how to improve water quality in existing degraded artificial lakes.

An impoundment can range in size and impact from:

* a weir where the water supplier takes ‘run of flow’; this is usually when the required water volume is small compared with the flow in the river,

to:

* a multi-day retention impoundment; this is usually a stream, where winter flows are stored to meet the summer water demand.

A weir does not change the water quality very much. A multi-day retention impoundment can modify the water quality in the impoundment and downstream, and the quality of the water in the impoundment can vary with depth, see Chapter 4: Source and Treatment Selection, section 4.4.1. Off-river storage is discussed in Chapter 12: Pretreatment Processes.

Excessive productivity of phytoplankton (eutrophication) is probably the main water quality problem in New Zealand lakes and is manifested by algal scums, turbid waters, deoxygenation of bottom waters, unpleasant tastes and odours, and excessive macrophyte (aquatic weed) growth (Vant 1987).

Blooms of blue-green algae (cyanobacteria) may release toxins at a level that is harmful to human health when critical concentrations (about 15,000 cells/mL for contact recreation) are exceeded, see Chapter 9.

Phytoplankton blooms can also impart unpleasant taste and odours to water that may require costly forms of treatment. Faecal contaminants are less problematic in standing waters than in rivers and streams, because of inactivation by lengthy exposure to sunlight and other inactivation processes, predation and sedimentation (Auer and Niehaus 1993).

Algal blooms occur in lakes with high nutrient concentrations, such as those in pasture catchments, during periods of calm, fine weather when high sunlight and stratification permit algal cells to occur. Elevated concentrations of nutrient elements (N and P) are associated with intensification of agricultural land. Furthermore, diffuse runoff from farms contributes inputs of faecal matter and potentially, pathogens. Thus it is important that impounded waters being used for drinking-water supply have in place ways of intercepting runoff, such as riparian buffer zones that trap N, P and faecal microbes; protected wetlands that enhance N removal by denitrification; and adequate fencing to keep stock away from waters and hence, minimise inputs of faecal matter (Williamson and Hoare 1987). These issues are explored more fully in section 3.5.1.

Prevention by riparian strips and control of land use etc is more effective than using algicides such as copper sulphate. Algicides have difficulty in removing an algal bloom; they are more effective at preventing a bloom if dosed early enough. Risk management issues relating to algicides are discussed in the MoH Public Health Risk Management Plan Guide PHRMP Ref. P4.1: Pretreatment Processes – Algicide Application.

A commercial product, PhoslockTM, has been developed in Australia to remove phosphorus from water. PhoslockTM is a reaction product of bentonite clay and lanthanum chloride in which the proportion of exchangeable cations (mainly sodium) is replaced by lanthanum cations through electrostatic binding. PhoslockTM is designed to adsorb oxyanions, predominantly phosphate, from a variety of natural aquatic environments notably in order to reduce the incidence of algal blooms. The recommended dosage is 100:1 PhoslockTM to filterable reactive phosphorus (FRP). Bentonite has been used in the past as a coagulant aid. Lanthanum is not normally found in significant concentrations in natural waters, and is not normally added. NICNAS (2014) has assessed the use of PhoslockTM. The Guidelines include a datasheet for lanthanum.

A report of three applications of PhoslockTM in appeared in McIntosh (2007). The plan involved lake treatment to remove phosphorus by 100 kg/year for three years. The author stated that the phosphorus load in the sediment has been reduced by nearly 100 kg/year with a strengthened phosphorus absorbing capacity of Lake Okareka expected for 3‑4 years. After the Phoslock application the date when the deoxygenation of the bottom waters occurred was delayed by over a month. NIWA (2008) compared four P-inactivation agents.

Reservoir catchments that are predominantly native or plantation forest are likely to have lower specific yields (kg/ha/y) of pollutants such as sediment and nutrients (Table 3.8).

Table 3.8: Specific yields (kg/ha/y) for different land uses in New Zealand

|  |  |  |  |
| --- | --- | --- | --- |
| **Land use** | **SS** | **TN** | **TP** |
| Intensive dairy | 142 | 35 | 1.16 |
| Average grazed pasture | 600–2000 | 4–14 | 0.3–1.7 |
| Urban development | 200–2000 | 2.5–11 | 0.4–1.6 |
| Plantation forest – disturbed | 300–2000 | 0.06–0.8 | 0.4–8 |
| Plantation forest – undisturbed | 500 | 0.07–0.2 | 0.15 |
| Native forest | 27–300 | 2–7 | 0.04–0.68 |

Source: Davies-Colley and Wilcock (2004). SS = suspended sediment; TN = total nitrogen; TP = total phosphorus

Rates of water movement in lakes (and reservoirs) are very slow by comparison with rivers and this permits water composition to change substantially between inflows and outflows as well as allowing large variations within different parts of a lake (Hamilton et al 2004). Density stratification related to gradients in water temperature within deeper lakes results in contrasting water chemistry in the upper (epilimnion) versus lower (hypolimnion) water layers.

The mean water residence time (in days) of a lake is given by:

τ = *V/Q*

where Q is the outflow (m3/day) and V is the lake volume (m3). τ varies from several hours to a few days for reservoirs behind dams on rivers, and to several years for lakes where the lake catchment is small relative to the lake volume (Hamilton et al 2004).

Phytoplankton are less of a problem in lakes and reservoirs with short residence times (weeks – months) because cells tend to be washed out faster than they can multiply (Howard-Williams 1987). For flushing to be effective as a means of controlling algal biomass, the lake inflow must be large enough, and there must be control facilities that allow the inflow to be regulated. Rapid flushing of lakes may prevent buoyant scums formed by blue-green algae, by creating instability in the water column (and reducing average light exposure and bicarbonate availability) through increased circulation. Other methods for reducing nutrient concentrations and thus lowering algal biomass in lakes include diversion of waters with high nutrient loads, and flocculation to strip P from the water column by converting it into a solid form that settles (eg, alum was used to strip P from Lake Okaro, in a trial in 2004).

At times when lakes are thermally stratified, hypolimnion waters may become deficient in dissolved oxygen (anoxic or anaerobic) causing many constituents to occur in a reduced state (eg, inorganic N will be predominantly NH4+).

By comparison, the epilimnion is well-oxygenated through exchange with the atmosphere, and constituents are nearly always in an oxidised form (eg, NO3- is the dominant form of inorganic N). Thus, waters drawn from deeper waters may undergo changes associated with oxidation, when passing through a water treatment plant.

The Hayes Creek reservoir (supplying Papakura District Council) has anoxic bottom waters containing reduced Mn2+ that is readily oxidised to form black precipitates of MnO2 on exposure to air. To prevent this, an oxygen curtain is deployed upstream of the water intake to oxidise the Mn before it gets to the treatment plant.

H2S from geothermal sources, or produced by anaerobic metabolism of SO42- in sediments, may produce the rotten egg smell associated with many highly reducing environments (Hamilton et al 2004). A recent example of this was H2S produced during decomposition of drowned vegetation and soils in the lake formed by the Opuha dam in South Canterbury (Hamilton et al 2004).

Lakes and reservoirs can be aerated purposefully in order to reduce stratification. A common result of destratification is an improvement in water supply quality (the first artificial circulation system was used in 1919 in a small water supply reservoir). By introducing oxygen to the (previously anoxic) hypolimnion, problems caused by reduced iron and manganese, and gases like H2S, are greatly reduced as well.

The purpose of aeration in lake management is to increase the dissolved oxygen content of the water. Various systems are available to help do this, either by injecting air, mechanically mixing or agitating the water, or even injecting pure oxygen. Aeration can increase fish and other aquatic animal habitats, prevent fish kills, improve the quality of domestic and industrial water supplies, and decrease treatment costs. In some cases, nuisance algal blooms can be reduced, or a shift to less objectionable algae species can occur. However, starting aeration after the nutrient levels have built up in the hypolimnion is likely to raise their concentration near the water surface, causing biological growth. Risk management issues relating to destratification are discussed in the MoH Public Health Risk Management Plan Guide PHRMP Ref. P4.2: Pretreatment Processes – Destratification.

See Chapter 12: Pretreatment Processes, section 12.3.2: Off-river Storage for further information, including some information re reduction times for selected micro-organisms.

### Springs

Springs are sources of emergent groundwater and may have very long or very short path lengths from their source surface waters. When used for drinking-water supply, springs should provide a reliable (continuous) supply of water and be of suitable quality. Non-piped water supplies, such as water collected from bores or springs, may often be contaminated with pathogens and require special treatment to achieve safe supply (WHO 2004). Even if spring water has reached the surface from a great depth, it is likely to contain sub-soil water too. Springs can be contaminated at the point at which water issues from the ground, eg, if animals are permitted to graze nearby. Waterfowl may also contribute to high levels of *E. coli* around springs. Springs can also be contaminated by runoff from the catchment that they drain, or from the soils that surface water passes through before re-emerging in spring water. For example, springs draining the Bombay Hills market garden areas have high nitrate concentrations that sometimes exceed the drinking-water Maximum Acceptable Value (MAV) of 50 mg/L (as NO3) (Wilcock and Nagels 2001). Hickey Spring in Pukekohe was monitored from 1959 to 2015; the nitrate concentration steadily increased from 6.5 to 20 mg/L as N.

Geothermal springs may contain health-significant concentrations of toxic chemicals, ie, arsenic, mercury and sometimes, elevated levels of fluoride. Geothermal springs are not used for drinking-water supply in New Zealand but they do influence the chemical composition of Waikato River, which is a major water supply resource, by contributing significant levels of arsenic and boron. Median concentrations of As and B are 0.028 mg/L and 0.26 mg/L respectively, at the nearest upstream site to the Hamilton water treatment intake (Smith 2003). The drinking-water MAVs for these elements are 0.01 and 1.4 mg/L, respectively (MoH 2005). Some water supplies may be drawn from hydrothermal springs, where the water temperature is higher than expected. These should be tested for the chemicals mentioned above.

Risk management issues relating to springs are discussed in the MoH Public Health Risk Management Plan Guide PHRMP Ref. P1.4: Groundwater Abstraction – Springs.

## Legislation

### General

Catchment protection involves firstly defining the boundaries of the catchment and determining who is responsible for the catchment and the protection of the water quality. A catchment is the drainage area upstream of the raw water abstraction point, or the aquifer and recharge zone of a groundwater system. The *Resource Management Act 1991* allows (Schedule 2 Part1 clause 1), but does not require, provision to be made in a District Plan for ensuring an adequate supply of water with regard to the subdivision of land. Steps taken to achieve this could include:

* leak reduction plans to be implemented before increased abstraction is allowed
* assessment of environmental effects of capital expenditure projects, including assessment of the costs they impose on present and future drinking-water supplies
* domestic water saving programmes
* industrial water use audits and water-use efficiency programmes
* development of alternative supplies
* restrictions on new subdivisions where the regional and district plans do not provide for an adequate drinking-water supply.

Under the *Local Government Act 2002* (Part 7, Subpart 1, sections 124–126)a territorial authority is obliged to assess the provision of water services and other sanitary services within its district, and describe the means by which drinking-water is obtained by residents and communities in the district, including the extent to which water supply is provided within the district by the territorial authority or other persons. It must also describe whether the water is potable (section 126(1)(i)(B)), and make an assessment of:

* any risks to the community relating to the absence in any area of either a water supply or a reticulated wastewater service or both (section 126(1)(b))
* the quality and adequacy of supply of drinking-water available within the district for each community (section 126(1)(c)
* a statement of current and estimated future demands for water services within its district (section 126(1)(d))
* any issues relating to the quality and adequacy of supply of drinking-water for each community (section 126(1)(d)(i))
* a statement of the options available to meet the current and future demands for drinking-water (section 126(1)(e))
* the suitability of each option for the district and for each community within it.

The territorial authority is also to provide:

* a statement of the territorial authority’s intended role in meeting the current and future demands (section 126(1)(f)) identified in section 126(1)(d) and proposals for meeting the current and future demands identified in section 126(1)(d), including proposals for any new or replacement infrastructure.

A local government organisation that is defined under the Local Government Act 2002 to mean “a local authority, council-controlled organisation, or a subsidiary of a council-controlled organisation, that provides water services”, that provides water services to communities within its district or region must continue to provide water services and maintain its capacity to meet its obligations. It must not lose control of, sell, or otherwise dispose of, the significant infrastructure necessary for providing water services in its region or district, unless, in doing so, it retains its capacity to meet its obligations.

Local government organisations must not close down or transfer a water service unless there are 200 or fewer persons to whom the water service is delivered who are ordinarily resident in the district, region, or other subdivision; the opinion of the MoH has been made public; and 75 percent or more of the public have agreed.

A local government organisation may only close down a water service under section [131(1)(a)](http://www.legislation.govt.nz/libraries/contents/om_isapi.dll?clientID=150237756&hitsperheading=on&infobase=pal_statutes.nfo&jump=a2002-084%2fs.131-ss.1&softpage=DOC#JUMPDEST_a2002-084/s.131-ss.1) if it has first reviewed the likely effect of the closure on the public health of the community that would be affected by the closure (section 134(a)(i)); on the environment in the district of that community (section 134(a)(ii)); and assessed, in relation to each property that receives the water service, the likely capital cost and annual operating costs of providing an appropriate alternative service if the water service is closed down (section 134(b)); compared the quality and adequacy of the existing water service with the likely quality and adequacy of the alternative service referred to (in section 134(b)) identified above (section 134(c)).

A local government organisation may enter into contracts for any aspect of the operation of all or part of a water service for a term not longer than 15 years (section 136(1)).

A local government organisation may only transfer a water service under section 135 if it has first:

* developed a draft management plan under which the entity representative of the community would maintain and operate the water service
* assessed the likely future capital and operating costs of the entity representative of the community to maintain and operate the water service
* assessed the ability of the entity representative of the community to maintain and operate the water service satisfactorily.

Knowledge of localised hydrological conditions that contribute towards water quantity and quality are essential for the design and implementation of a catchment protection scheme. These conditions include natural inputs such as seasonal rainfall variations or regional geology, and man-made inputs such as agricultural chemicals, industrial and domestic wastes, erosion and animal activity. Once all factors contributing to water quality have been identified, the planning and design of a catchment protection strategy can commence. Landcare Research has produced a National Map of Soil Erosion, see[: http://www.landcareresearch.co.nz/science/soils-and-landscapes/ecosystem-services/factsheets](file:///C:/Users/ROdean/AppData/Local/Microsoft/Windows/INetCache/Content.Word/:%20http:/www.landcareresearch.co.nz/science/soils-and-landscapes/ecosystem-services/factsheets)

The prime objective of a catchment management strategy, or planning for a drinking-water supply, should be to protect and, if necessary (and achievable), to enhance the quality of source waters. The rules in the plans define the activities that can take place in the catchment.

Current legislation allows for the protection of the quality and other aspects of the source waters. The predominant legislation under which this can be achieved is the Resource Management Act 1991 (RMA), with its key purpose as the sustainable management of natural and physical resources, and the Health Act 1956 (HA).

Regional councils have responsibility, under the RMA (s30(1)(c)), to control land use in order to protect the water quality within their respective catchments. Responsibilities include controls over the use and diversion of source water (RMA s30(1)(e)), discharge of contaminants into the water (RMA s30(1)(f)), and in relation to any bed of a water the planting of vegetation on land for the purpose of maintaining and enhancing of water quality of that water body (RMA s30(1)(g)). Regional plans, district plans and resource consents under the RMA are the main tools for managing the water quality of source waters.

However, although these tools are available under the RMA, they are frequently not used effectively. The provisions for drinking-water values in regional plans are an example of this. Thus, only six of sixteen regional councils or unitary authorities have a comprehensive approach to the management of drinking-water catchments. The other councils have either not addressed the issue, or have done so in a very general way (Ministry for the Environment 2004).

There is currently (2005) **no** **specific requirement** in the Resource Management Act for consent authorities to consider the impact proposed activities may have on source water in a drinking-water supply catchment. Consequently there is potential for land use activities/ discharges to be consented that reduce water quality at the point of abstraction to below that which the plant is designed to treat. This presents potential health risks to the community and may result in significant costs to the supplier in upgrading treatment facilities.

Part 7 (section 126) of the Local Government Act (2002) requires local authorities to undertake a specific assessment of the quality and adequacy of drinking-water supplies. However there is no requirement to manage source water quality, which is the aim of the National Environmental Standard (NES), see next section.

While section 5 of the Resource Management Act refers to social, economic and cultural well-being for people and communities, there is no **specific** requirement for consent applicants to consider the impact of their proposed activity on community drinking-water supplies. Whilst it can be argued that the definition of environment in the Resource Management Act includes public health, there is no specific reference to community drinking-water supplies in the Act.

The Ministry for the Environment has produced a National Environmental Standard under the Resource Management Act to improve how drinking-water is managed at source. This standard is intended to complement Ministry of Health legislation and standards for improving drinking-water supply and delivery; see section 3.4.2.

The Health Act 1956 allows the Governor General to declare, by Order in Council, any water supply source, whether publicly or privately owned and operated, to be under the control of a territorial authority if this is necessary in the interests of public health (HA section 61(2)). The Health Act also makes it an offence to create a nuisance or to allow a nuisance to continue (HA section 30) including allowing a water source to be offensive, liable to contamination, or hazardous to health (HA section 29(p)).

Catchments dedicated for water supply purposes and under the control (by ownership and/or declaration) of a territorial authority or regional council, may be controlled simply by the use of bylaws. The Model General Bylaws for Water Supply define appropriate management controls for the protection of water quality in such catchments (NZS9201: Chapter 7: 1994). There are circumstances where specific legislation has been developed that relates to water supply, for example the Wellington Regional Water Board Act 1972. This Act is an important statute for the regional council under which it holds large areas of land in the Wellington metropolitan area.

In other situations abstractions for water supply are often only one of many demands on the water resource. In this case the catchment management strategy or plan will need to be incorporated within the overall regional (or district) plan process.

The reliability of production of a continuous, adequate, supply of safe water from a large river or active catchment will be enhanced by use of off-river storage. This offers the ability to choose when raw water should be abstracted, thus avoiding periods when water treatment may be difficult, or when the river may be contaminated.

### National Environmental Standards (NES)

For the multi-barrier principle to be implemented properly in the management of drinking-water supplies, the water supplier needs to be able to put barriers to contamination in place from the water source through to the consumer’s property. In the past, this has been difficult for many water suppliers in New Zealand by legislation which separated responsibilities for catchment management from those of treatment and reticulation of water. The Resource Management Act (1991) makes regional councils responsible for the management of source catchments, while health legislation makes water suppliers responsible for the water supply from the point of abstraction to the consumer.

To ensure that the supply of water for drinking-water production is taken into consideration when decisions are made regarding activities in catchments, the Ministry for the Environment developed a national environmental standard for raw public drinking-water, ie, source water. The original proposal for this standard was that it be a grading standard. This approach did not require a minimum water quality to be achieved, but it proposed the generation of a grade for the raw water to assist communities in making decisions about the management of their water resources.

Following public consultation, the form of the NES was revised in early 2005. It is now a narrative standard. The National Environmental Standard for Sources of Human Drinking Water came into effect on 20 June 2008. The standard is intended to reduce the risk of contaminating drinking water sources. See MfE (2009): *Draft Users’ Guide.*

The standard requires regional councils to ensure that effects on drinking-water sources are considered when making decisions on resource consents and regional plans. Specifically, the standard requires that regional councils:

* decline discharge or water permits that are likely to result in community drinking-water becoming unsafe for consumption following existing treatment
* be satisfied that permitted activities in regional plans will not result in community drinking-water supplies being unsafe for consumption following existing treatment
* place conditions on relevant resource consents requiring notification of water suppliers and consent authorities if significant unintended events occur that may adversely affect sources of human drinking-water.

The draft standard was refined following public consultation in 2005 and several key changes were made based on submissions received. These included:

* applying the consent component of the NES to water and discharge permits only
* assigning regional councils (not territorial authorities) the primary responsibility for implementing the majority of the standard (reflecting existing responsibilities and expertise in water quality)
* increasing the community water supply population threshold for application of the standard from 25 to 500 people, to reduce implementation costs.

The Ministry for the Environment will produce guidance material to assist regional councils and consent applicants apply the new standard.

The NES is a regulation, so it is binding and prevails over rules and resource consents. More details of the standard are available at the following link: <http://www.mfe.govt.nz/laws/standards/drinking-water-source-standard.html>

The NES covers emergency notification provisions. Under the NES, emergency notification refers to the notification (preferably by phone) of authorities when an unintended activity occurs (this differs from the notification of a consent application under the RMA). One key difference between emergency notification provisions and previous parts of the regulation is that they now apply to a smaller population threshold: activities with the potential to affect registered drinking water supplies that provide 25 or more people with drinking water for 60 or more days of a calendar year must be notified.

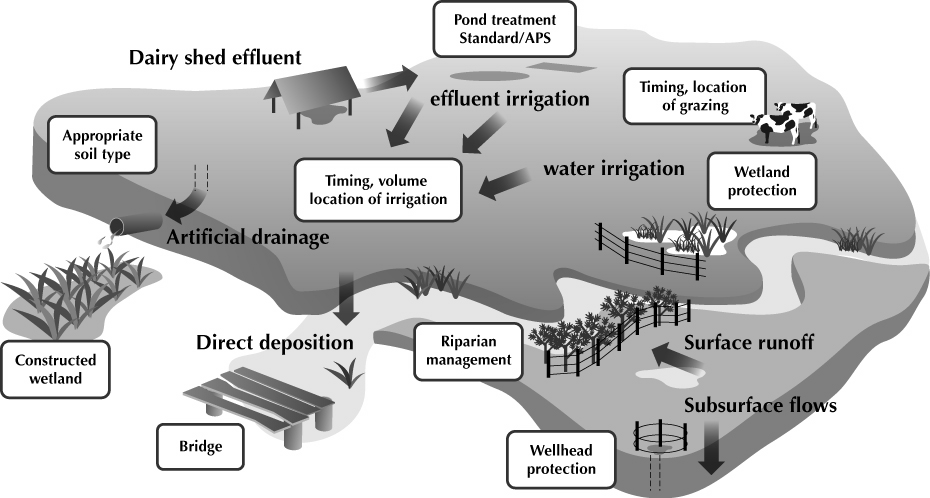
## Mitigation of pollutants and catchment protection

Water contamination may arise from a variety of sources, including seepage from pipelines, human and animal effluent, landfill leachate, industrial effluent disposal, use of pesticides and fertilisers, mining, leakage from underground tanks, transportation accidents, salt water intrusion, and poorly constructed bores or bore head protection, see Figure 3.3. Groundwater contamination usually occurs in a far less conspicuous manner than surface waters, and is discussed in section 3.2. This section discusses catchment protection and the mitigation of surface water contamination. Some nutrient and sediment control practices are discussed in MfE (2002). Refer also to Appendix 4 of MfE (2009): Activities and Contaminants that may Contribute to Source Waters.

If a river has the potential to receive contamination that the treatment plant is not designed to remove, consideration should be given to the use of off-river storage. Off-river storage is discussed in more detail in Chapter 12: Pre-treatment Processes. Chapter 4: Source and Treatment Selection also includes some discussion on catchment protection, mainly related to micro-organisms.

WHO (2016) covers catchment risk assessment, the various hazards, and their control. The link includes excellent editable checklists and tables.

Figure 3.3: Catchment protection



### Rural activities

Whilst drinking-water catchments ideally should be devoid of inputs of human and animal waste, in reality total absence is rare. Typically, therefore, the water treatment process can benefit from attempts to mitigate such pollution at or near to its source. Wastes from animals are known to contain nutrients, pathogens, heavy metals and endocrine-disrupting chemicals, all of which can be transferred to water bodies by the deposition of urine and faecal material directly to a stream or lake, and via surface and subsurface flow pathways. MPI (2011) assessed the ammonia production from animal excreta.

A number of mitigation options exist to reduce this transfer, although the research to-date typically has excluded heavy metals and endocrine-disrupting chemicals, focusing upon sediment, nutrients and faecal microbes. However, treatment systems that effectively remove sediment might be expected to also remove metals (eg, cadmium from phosphatic fertilisers; zinc used for facial eczema treatment). If the water supplier owned the catchment, they would be able to control most land uses and activities. Some have done this, then converting from pastoral farming to forestry.

Nutrient levels, particularly nitrate have been increasing gradually in many waters for many years. Sewage can contain about 25 mg/L of ammonia, a lot of which ultimately is oxidised to nitrate. Agriculture activities also contribute nitrate (Landcare 2016):

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **kgN/ha/yr** | **Dairy** | **Apples** | **Grapes** | **Outdoor vegetables** | **Other pasture** | **Forest and scrub** |
| N-NO3 loss | 24–69 | 3–18 | 4–18 | 16–51 | 10.7 | 2.5 |

Farming (general)

The effect of agriculture on water quality is dependent on the size of the catchment vs the flow in the river (or volume of the lake), the type, intensity and management of farming, and climatic effects.

Problems commonly arise from animal wastes, especially from cowsheds, holding pens, holding paddocks and yards, and whether the animals have direct access to water. Problems can also arise from septic tanks wastes, and the transport, storage and use of pesticides and fertilisers.

Approaches that can be considered for mitigating these effects include:

* allowing only approved animals
* specifying stocking rates and grass/fodder length
* standards for fencing
* installing riparian strips – specifying size, planting
* adopting approved fertiliser application rates
* using approved fertiliser applicators
* using approved pesticides and application rates
* using approved pesticides applicators
* requiring bunded chemical storage areas
* instituting waste controls and treatment, including dairy shed, offal pits, sheep dips, etc
* introducing holding paddock/yard/pen waste controls (pens include buildings for pigs, chickens, saleyards, etc).

Environment Waikato (2010) reviewed information about the origins of faecal contamination in Waikato waterways and the mitigation options available to address this issue, focussing on rural farmland. Monaghan (2012) looked at the impacts of animal wintering which is increasingly recognised as a critical phase of pastoral farming that has an important influence on animal performance and on contaminant losses from farms to water.

A study of the public health issues associated with stock accessing waterways upstream of drinking water off takes in Australia was reported by the Victorian Department of Health (2011); risks to public health were estimated to be 5 log above tolerable levels. This report includes the statement that the costs of outbreaks overwhelmingly exceed the costs of their prevention. A key finding was that the major source of risk posed by *Cryptosporidium parvum* in typical grazing water supply catchments arises from pre-weaned calves and lambs. Removing calves and lambs from the catchment or housing them in hydrologically isolated areas can reduce the risk by approximately 3 log.

WHO (2012) stated that although there are a large number of zoonotic pathogens that affect humans, five are known to cause illness around the world with high-frequency: *Cryptosporidium*, *Giardia*, *Campylobacter*, *Salmonella* and *E. coli* O157. Efforts to control these pathogens are likely to be effective in controlling other related zoonotic pathogens whether known, as-yet-unrecognised or emergent. Domestic animals such as, poultry, cattle, sheep and pigs generate 85 percent of the world’s animal faecal waste, proportionally a far greater amount than the contribution by the human population. The faecal production rate and contribution to the environment of these animals can be as high as 2.62 × 1013 kg/year. Limiting zoonotic pathogen-shedding in farm or production facilities for domestic animals should be accomplished by preventing illness in livestock, through minimising exposure to pathogens, by increasing immunity, by manipulation of the animal gastrointestinal tract microbial ecology and by managing (including treating) animal waste to reduce the release of zoonotic pathogens into the environment.

See DWI (2012) for a discussion on the effect of veterinary medicines on water, where the usage, treatment regimes, metabolism, environmental fate and toxicity of around 450 active ingredients in use in the UK were assessed. Twenty-six substances were identified of potential concern and these were then evaluated using more complex modelling approaches for estimating exposure levels in raw waters and for estimating removal in different drinking water treatment processes. For 14 of the 26 selected priority veterinary medicines, the estimated intakes from conventional or advanced treated water were less than 10 percent of the Acceptable Daily Intake (ADI) for all sections of the population evaluated. It is concluded, therefore, that these 14 veterinary medicines – albendazole, amoxicillin,\* chlortetracycline, chlorsulon,\* cypermethrin, cyromazine, diazinon, enrofloxacin, eprinomectin, lasalocid, salinomycin, tiamulin, trimethoprim and tylosin – are not a potential risk to consumer health. Very minor exceedances of the guide value (equivalent to 10 percent of the ADI) in all populations assessed were found for a further two compounds: halofuginone and tilmicosin. However, these were not considered to be a potential risk to consumer health. For the remaining 10 compounds (acetylsalicylic acid,\* altrenogest, apramycin, cefapirin, dicyclanil, florfenicol, lincomycin, luprostiol,\* monensin, sulfadiazine), the worst case predicted exposure levels, based on consumption of either raw (environmental) water or conventionally treated water were close to or exceeded ADI values. In some cases the predicted levels of exposure significantly exceeded ADI values. The highest exceedances of ADI values arose from exposure to water sourced from groundwater. There is some evidence that the groundwater model that was used in the study significantly over estimates actual concentrations in the real environment. In the advanced water treatment scenario, worst case predicted exposure estimates only exceeded the ADI value for four compounds (acetylsalicylic acid, florfenicol, lincomycin and luprostiol). All of these ADI exceedances were related to the groundwater scenario. Those marked \* are not on the ACVM Register as at 2012.

MPI (2015) assessed effects of a proposed dam on the Waimea Plains, and MPI (2015a) began to look at ways to protect and restore the Waikato and Waipa Rivers.

Managing pesticides – an interesting trial

At the request of the UK Government, the Crop Protection Association (CPA) was asked to develop its thoughts on a focused approach towards minimising the environmental impacts of pesticides as an alternative to a proposed pesticide tax. In collaboration with other farming and crop protection organisations, the CPA prepared a five-year programme of voluntary measures. In April 2001, after public consultation, the Government accepted this approach as an alternative way forward, now known as The Voluntary Initiative. Early results in some catchments show up to 60 percent reductions are possible (The Voluntary Initiative 2005). Key measures that have been identified as needing a high level of farmer uptake include:

* Crop Protection Management Plans: a self-assessment which helps farmers review the potential environmental risks associated with crop protection on their farm
* The National Sprayer Testing Scheme: ensures that the spray equipment is correctly maintained and capable of applying the product accurately with no leaking joints or drips
* The National Register of Sprayer Operators: recruited over 20,000 active professional spray operators who are being encouraged through continuous professional development to obtain extra training and information.

Treatment of dairy farm effluent

Historically, the most common form of treatment for dairy farm effluent has been by a two-pond system combining both an anaerobic and facultative pond (Sukias et al 2001). This method is efficient at removing sediment and biochemical oxygen demand (BOD), but high concentrations of nutrients and pathogens can remain (Hickey et al 1989), often discharging directly to a waterway.

Following the introduction of the Resource Management Act in 1991, land treatment of dairy effluent is now favoured by most regional councils. This approach, relative to the two-pond system, generally results in a marked reduction in the loss of nutrients and pathogens to waterways. However, excessive levels of these pollutants can still occur. For example, Houlbrooke et al (2004a) reported that 2–20 percent of nitrogen and phosphorus applied to land with dairy effluent is leached directly through the soil profile to enter a water body. Whilst reducing the propensity for surface runoff, artificial subsurface drains are known to transfer both nutrients and pathogens to water bodies (Monaghan and Smith 2004; Ross and Donnison 2003). Pollutant transfer via drainage can occur under both grazed and irrigated systems. Dodd and McDowell (2014) describe land uses that can contribute phosphorus to groundwater, and later cause biological growths when discharging to surface water. Landcare Research has produced a National Map of Phosphorus Leaching, see[: http://www.landcareresearch.co.nz/science/soils-and-landscapes/ecosystem-services/factsheets](file:///C:/Users/ROdean/AppData/Local/Microsoft/Windows/INetCache/Content.Word/:%20http:/www.landcareresearch.co.nz/science/soils-and-landscapes/ecosystem-services/factsheets)

The success of land treatment of wastes depends strongly upon soil type. For example, Aislabie et al (2001) showed poorly drained gley soils to be much less efficient than allophanic and pumice soils in attenuating bacterial indicators applied in effluent. Generally, soils with a fine structure and absence of macropores are more appropriate for receiving and treating effluent and, faecal material deposited by grazing animals.

Improved timing of effluent application to land, ie, through avoiding irrigation of effluent during wet weather, has been shown to reduce pollutant transfer to waterways (Monaghan and Smith 2004). Deferred irrigation, which involves effluent storage until a suitable soil water deficit arises, has resulted in only 1 percent of applied nutrients reaching subsurface drains (Houlbrooke et al 2004b).

Recent studies using constructed wetlands have shown potential in the treatment of drain flows under grazed dairy pasture, particularly with respect to nutrients (Tanner et al 2005). This approach is also applicable to drainage flows generated by the application of effluent to land.

Advanced pond systems are an alternative to the land application of effluent. These consist of four types of ponds arranged in series (an advanced facultative pond, a high rate pond, algal settling ponds, and a maturation pond) that result in effluent of a considerably higher quality than the traditional two-stage oxidation ponds (Craggs et al 2004).

Several regional councils have been addressing dairy farm effluent issues more proactively in recent years, producing guidelines and best practices documents, eg, Horizons (2008).

Riparian buffer strips

In addition to subsurface processes, agricultural pollutants can be transferred to waterways by surface runoff generated under rainfall (Houlbrooke et al 2004b; Collins et al 2005). Riparian buffer strips are a potential means of attenuating pollutants carried within surface runoff, with the dense vegetation of the buffer encouraging infiltration of the runoff and deposition of particulates.

The efficiency of riparian buffers varies with topography, soil type and the magnitude of a rain event (Parkyn 2004; Collins et al 2004). In addition, soluble nutrients, clay-sized particles, and free-floating (ie, unattached to soil or faecal material) faecal microbes are less susceptible to deposition and, therefore, are less readily attenuated than particulates.

Riparian management guidelines are available with respect to control of nutrients and sediment (Collier et al 1995) and faecal microbes (Collins et al 2005). Some regional councils are also developing guidelines for their parts of the countryside, eg, Auckland Regional Council (2001), and Environment Canterbury (based on ECan 2003). Also, government departments have issued guidelines for managing waterways on farms (MfE/MAF 2001). A summary of recent research in this area is now available (MAF 2004, MAF 2006a).

The Waterways Centre for Freshwater Management was established in 2009 as a joint partnership between Canterbury and Lincoln Universities. Their website gives links to and produces publications on enhancing water quality in agricultural areas; see <http://waterways.ac.nz>. For example, WET (2014) includes a practical section on riparian management.

Vegetated buffer strips were tested to see if they were effective at removing *Cryptosporidium* during rainfall rates of 15 or 40 mm/h for four hours. Buffers were set on a slope of 5 to 20 percent and soil textures consisted of silty clay, loam, or sandy loam. It was found that vegetated buffer strips consisting of sandy loam or higher soil bulk densities had a 1 to 2 log reduction/m. Buffers consisting of silty clay, loam, or lower bulk densities had a 2 to 3 log reduction/m. Also, it was found that vegetated buffer strip made of similar soils removed at least 99.9 percent of *Cryptosporidium* oocysts from agricultural runoff when slopes were less than or equal to 20 percent and had a length of at least three metres (Atwill et al 2002 – reported in Appendix E of USEPA 2009).

Natural wetlands

Near-channel saturated areas or wetlands are found extensively in pastoral landscapes in New Zealand. These typically develop where steep hill slopes cause the convergence of surface and subsurface flows, or where an impervious layer exists within the soil profile. Such wetlands have been shown to attenuate nitrate through the process of denitrification, provided that water moves through a wetland slowly enough (Burns and Nguyen 2002; Rutherford and Nguyen 2004).

Modification of wetland drainage through cattle trampling, installation of subsurface drains or artificial channels is, therefore, likely to diminish their pollutant attenuating properties. Cattle are attracted to the smaller, shallower areas of the wetlands for grazing, and excluding stock from them is likely to yield improvements in wetland bacterial water quality (Collins 2004). For guidelines for constructed wetlands treatment systems for dairy farms, see Tanner and Kloosterman (1997).

Preventing direct deposition to waterways

Faecal contamination of freshwaters arises, where animals have access, through the deposition of faeces directly into waterways. Direct deposition can occur when cattle cross a stream on the way to or from the milking shed (Davies-Colley et al 2004) and through sporadic incursions into the water at access points along the stream bank (Bagshaw 2002). Bridges and the fencing of stream banks are the key mitigation measures for each of these processes, although providing alternative water sources (drinking troughs) can also reduce sporadic incursions, reducing faecal contamination of waterways (Sheffield et al 1997).

Human activity may need to be curtailed too, particularly at impoundments. Water suppliers will need to decide whether to allow swimming, boating and fishing in the impoundment, and how close houses, public toilets and car parks should be to the water.

Forestry

Land clearing, planting and felling can cause large increases in silt run-off. These activities can be controlled by adopting guidelines such as developed by Environment BOP (2000). BOP Regional Council updated this (BOPRC 2013). ARC (2007) published *Forestry Operations in the Auckland Region: A guideline for erosion and sediment control*. Methods for mitigating fine sediment in coastal areas were addressed in MDC (2015); this report includes references to many other recent New Zealand studies, and a review by Landcare Research staff.

If the headwaters of a catchment used for water supply is in native bush, the water supplier should do everything in its power to ensure that the area remains forested, eg, implementing pest control programmes.

The National Environmental Standards for Plantation Forestry (NES-PF) permit core forestry activities provided there are no significant adverse environmental effects. A new nationally consistent set of regulations will create more certainty (MPI 2017). Rules within regional plans may be more stringent.

Viser (2018) reported on the relationship between harvesting, harvest residues, and the best practices that help mitigate debris flow events and or the delivery of harvest residue in New Zealand.

### Urban and transportation pollutants

Urban pollution of waterways is primarily caused by contaminants being washed off streets and roofs and flushed through the stormwater drainage system to the receiving water. Contaminants of concern include nutrients, sediment, heavy metals, hydrocarbons, toxic organics and pathogens.

Urban pollutants are associated primarily with particulate material and this offers the potential for contaminant entrapment within, for example, stormwater retention ponds, wetlands, vegetated filter strips and swales, and the addition of filters or screens. Measures such as minimising imperviousness and retaining natural drainage channels will reduce both the source and transport of pollutants.

Sewage can be an intermittent pollutant via leaks or when pumping stations break down and discharge raw sewage to drains (Williamson 1991). Smaller settlements that still use septic tanks should adopt guidelines (and inspections) related to their design, construction, operation, maintenance and cleaning.

Water suppliers with urban communities upstream of their water supply intake should ensure that the appropriate authorities police trade waste bylaws, and consent conditions relating to activities such as landfills and storage of substances likely to have a deleterious effect on the water supply. For example, Sapere (2015) estimated the quantifiable impacts from the 2013 diesel spill into the Raetihi water supply at $2 million. Trade waste bylaws should require bunding of stored chemicals. The Hazardous Substances and New Organisms Act provides guidelines on storing hazardous substances. The Act has regulations and codes of practice to determine how substances should be transported, stored and used. The storage of hazardous substances must also comply with the New Zealand Building Act and the Resource Management Act.

Spills of hazardous substances during transport have the potential to cause serious problems for water suppliers. Measures that water suppliers may consider include requiring:

* trucking companies/drivers to use approved roads, and follow appropriate standards, for example: NZS 5433:1999 Transport of Dangerous Goods on Land, and The Liquid & Hazardous Waste Code of Practice (NZWWA)
* regional councils to co-ordinate with Fire Service and hazardous substances technical liaison committees re spills, etc
* also see: Stock Effluent from Trucks: Resource Management Guidelines for Local Authorities, Prepared by the Planning Subgroup for The National Stock Effluent Working Group; and the three companion documents:
* Volume I: Industry Code of Practice for the Minimisation of Stock Effluent Spillage from Trucks on Roads. National Stock Effluent Working Group. April 1999
* Volume II: A Practical Guide to Providing Facilities for Stock Effluent Disposal from Trucks. National Stock Effluent Working Group. Second edition, March 2003
* Volume III: Resource Management Guidelines for Local Authorities. March 2003.

### Petroleum development activities

In 2014 the Ministry for the Environment produced *Managing Environmental Effects of Onshore Petroleum Development Activities (Including Hydraulic Fracturing): Guidelines for Local Government*. This document is a non-statutory guide which provides clarity on the roles of central and local government in managing onshore petroleum development activities (including hydraulic fracturing). These guidelines cover the *whole* life cycle of onshore oil and gas well development, not just hydraulic fracturing.

The potential effects on drinking water supplies need to be managed in accordance with the National Environmental Standard for Sources of Human Drinking Water (discussed in section 5.6.2). Potential risks to drinking water sources are discussed further in section 3.8.4. Drilling muds are covered in section 4.2.2.

A variety of chemicals are used in fluids required for drilling and hydraulic fracturing operations. The environmental effects of these chemicals are primarily related to the transportation, storage and make-up of the hydraulic fracturing fluids before injection, along with managing the flow-back fluids after stimulation of the well. The management of these activities is discussed in sections 5.5 and 5.6. The *Guidelines* include New Zealand case studies. Appendix H lists the chemicals used most often in hydraulic fracturing operations.

The Parliamentary Commissioner for the Environment (PCE 2014) reported on the environmental impacts associated with drilling for oil and gas and analysed the complex system of laws, agencies, and processes that oversee and regulate the industry in New Zealand.

The Australian Department of the Environment (2014) produced a report providing an overview of Australian and international experiences of coal seam gas and coal mining co-produced water and risks to aquatic ecosystems. Table 3 shows the range of concentrations for many determinands found in coal seam gas co-produced water. Table 4 lists the chemicals which may be used in the hydraulic fracturing process. Also in 2014 they produced a report on hydraulic fracturing, and a report on modelling impacts of coal seam gas extraction on groundwater.

Ferrer and Thurman (2015) present a table showing the chemicals found in fracturing fluids and their specific purposes. They also discuss the chemicals that occur naturally in shale waters. This paper includes several informative references.

See also USEPA web site which links many reports: <http://www2.epa.gov/hydraulicfracturing>. One of these is their report discussing impacts on drinking water resources – see USEPA (2016).

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