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Faecal Contamination of Rural Waikato Waterways

Sources, Survival, Transport and Mitigation Opportunities

A review for Environment Waikato

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Executive summary

This report reviews information about the origins of faecal contamination in Waikato waterways and the mitigation options available to address this issue. The focus is on rural farmland. Human sewage is not included in this review, although it affects water quality in some localities.

Water quality monitoring shows that indicator species for faecal contamination are commonly found in Waikato waterways, often breaching standards for contact recreation. Indicator bacteria (such as the widely-used species *Escherichia coli*) show that faecal contamination has occurred, but do not identify the sources (e.g. wild animals vs livestock). Faecal pathogens sourced from different species have variable association with the incidence of human disease. A small proportion of *E. coli* strains can cause human disease, but other pathogens (such as *Campylobacter*) are more common causes of waterborne gastric infection. Indicator bacteria are hardy in the environment, whereas *Campylobacter* die off more rapidly. Therefore, while the presence of indicator bacteria demonstrates faecal contamination, it does not show when this occurred or the source, both of which factors influence the degree of health risk. However, broadly speaking, when *E. coli* are found in sufficient numbers, there is a correlation between *E. coli* and the pathogen *Campylobacter*. This association is reflected in the indicator published on the Environment Waikato (EW) website for unsatisfactory water quality.

A Waikato review (Collins 2002a) shows that catchments with lighter stocking density generally have lower levels of faecal indicator bacteria in waterways, but that only about a third of the variance in waterway faecal contamination can be explained by stock density differences. Dry-stock and dairy catchments can have similar *E. coli* loads (Wilcock 2006). Sheep manure is a concentrated source of faecal microbes, and microbe loading per hectare under intensive sheep grazing can be greater than that of beef or dairy. Deer also contribute high inputs of microbes, especially where their wallowing areas are connected with waterways.

Wild animals also shed faecal indicator bacteria, and contact recreation standards have been breached in small headwater streams flowing out of pine and indigenous forest (Donnison et al. 2004). The effect of wild animals on annual faecal microbe exports is generally overshadowed by livestock sources in intensively-farmed land, where farm management is not in place to reduce these impacts. However, wildfowl sources can have an impact at base flows and where bird habitat areas are connected with a waterway. Work is ongoing to establish the significance of wild sources in disease incidence, but it has been reported that pathogens from these sources are less closely linked to human illness than are ruminant types (Mullner 2009; French et al. 2010).

Microbes from livestock faeces reach waterways through direct deposition, effluent discharges, surface run-off, or sub-surface flow. Surface run-off from farmland during rainfall carries large numbers of microbes, although much of the faecal material deposited on the land never reaches the water. On pastoral land, the length of time lapse between a grazing event and rainfall has a strong effect on the concentration of microbes in run-off. Mechanisms that reduce survival and transport include exposure to UV light, desiccation, entrapment in vegetation, and soil filtration. Faecal indicator bacteria can survive in bed sediments, and re-suspension of this sediment during floods is a significant factor in peak *E. coli* concentrations. However, some pathogens (in particular *Campylobacter*) are more susceptible to die-off in sediments than the indicator *E. coli*.

Where stock can access or cross waterways and defecate in the water, there are no opportunities for entrapment or die-off. This results in direct contamination of water and bed sediments with pathogens that are in a highly viable state. Dairy cows are more likely to defecate when crossing a stream than when walking on a lane (Davies-

Colley et al. 2004). When cattle have free access to streams, a small percentage of faecal matter (typically 1-2%) is deposited directly in the water (Bagshaw 2002; Wilcock 2006). However, because there are so many microbes in stock faeces, and these include human pathogens, any deposition in a small waterway can create a health risk and cause a breach of contact recreation standards (McKergow and Hudson 2007). Direct deposition of faeces occurs at base flows, when the greatest downstream water use is expected.

Studies in the Waikato dairying catchment of Toenepi estimated that 95% of the annual yield was exported during storm flow events (Davies-Colley et al. 2008). Contamination during high rainfall events is more difficult to address than that occurring at base flows. However, it also carries less risk of human illness through contact recreation or eating commercial shellfish (as contact recreation is less likely and aquaculture harvest is suspended after heavy rainfall). Some other uses do continue at higher flows, though, such as non-commercial shellfish harvest and untreated stock- or drinking-water takes.

Transport pathways for faecal microbes differ, depending on catchments. In fine porous soils (such as pumice and ash), soil filtration is effective and there may be little run-off. Riparian buffers are less important in these circumstances, although some run-off can still occur in storms. Poorly drained soils promote run-off; catchments with these soils tend to have lower water quality (Collins 2002a). Also, where soils have blocky structure or cracks, or are artificially drained, microbes can move down with the soil water through these preferential flow-paths to reach surface water via sub-surface flow.

Mitigation options include stock exclusion, crossings and riparian filter strips. Overseas studies have shown stock exclusion at a sub-catchment scale can reduce microbial concentrations in streams by one- to two-thirds (Line 2003; Meals 2001). Stream *E. coli* concentrations were halved following the installation of bridge crossings for dairy herds over the Sherry River near Motueka, but this reduction was not sufficient to meet contact recreation standards (Young et al. 2008). Where run-off occurs in sheet flow rather than channelised flow, grass riparian strips can also filter large proportions of microbial contaminants. Conversely, buffer strips are unlikely to make much difference at high flows, or where drainage occurs through the soil rather than as overland run-off. Even where buffers trap a high percentage of microbes, large numbers of indicator bacteria can remain in run-off from farmland because the total catchment load is so high (millions of microbes per square metre). Furthermore, some of the microbes trapped in buffers can be released in subsequent events.

Protecting shallow wetlands from stock access is important as these sites are attractive to cattle, and faeces are often deposited in or near seepage areas (Collins 2002b). Microbes from this source can be transported in surface or sub-surface flow and therefore contribute ongoing contamination to base flows as well as being flushed through in rain events.

Dairy effluent is an obvious source of faecal microbes, and although there is some die-off in treatment ponds, discharge from ponds or poorly managed land application can affect water quality, especially at base flows. Further research is required to clarify the typical contribution of effluent treatment ponds at a farm scale. At a catchment scale, a reduction in pond numbers in Toenepi from 22 to 9 was not accompanied by any significant decline in faecal indicator bacteria in the stream (Wilcock et al. 2006). Furthermore, the comparative study of Waikato catchment data by Collins (2002a) found no correlation between the number of ponds in a catchment and microbial water quality. This suggests that overall, ponds are less important than other factors in Waikato stream faecal contamination patterns.

Land irrigation offers filtration and die-off opportunities, as long as application is at low enough rates to avoid run-off or drainage through the sub-soil. This is particularly important on sloping, heavy soils, or where there is artificial drainage, or soils have high “by-pass” flow properties. Avoiding dairy effluent contamination requires a suitable system for the soil conditions, and adequate storage so that effluent application can be deferred in wet periods. Where this is in place, contamination of waterways from dairy effluent will be minimised.

Constructed wetlands at the end of artificial drains can remove large numbers of microbes from concentrated effluent sources (such as spillages). They may not have a marked effect when in-flowing concentrations are low, as wetlands are attractive to wildlife species that also deposit faecal microbes. Management of open drains to retain drain vegetation promotes the removal of microbes from drainage water, but these may survive in sediments and be remobilised.

Other areas where faeces can build up, such as laneways and stand-off areas, contribute to contamination where run-off or sub-surface drainage waters carry microbes to waterways. This risk can be managed through location, design and maintenance measures. Grazing heavy soils near waterways during prolonged wet weather should also ideally be avoided.

As yet in New Zealand, there have been no comprehensive trials showing that implementing best practice at a sub-catchment scale can achieve water quality standards in intensively farmed areas. Studies in the “best dairying catchments” of Waiokura and Toenepi over ten years show that stock exclusion and effluent management changes have not yet achieved contact recreation standards. There is incomplete data on the extent of riparian protection and effective stock exclusion in these catchments over the study period, although some increase in percentage of streams fenced has been reported. In Waiokura, faecal indicator counts have decreased (Wilcock et al. 2009) but in Toenepi, there has been no significant change (Wilcock et al. 2006). Stocking intensity in Toenepi increased over the same period, but there is also evidence that wild sources contribute to faecal indicator counts in the stream, especially at low flows (French et al. 2010). Achieving contact recreation standards throughout farmed catchments at all flows is challenging because catchment loadings of faecal indicator bacteria are so high and there are so many sources in the environment. Excluding stock from waterways, seeps and riparian areas, and careful management of dairy effluent offer the best scope for achieving these standards at base flows. Modelling at Toenepi suggested that no suite of management practices was likely to achieve the standards at storm flows in that catchment (Muirhead et al. 2008).

Certain practices to improve water quality across all water bodies in the region are identified in the draft Regional Policy Statement (RPS), released in December 2009. These focus on areas near waterways, and could be expected to have a beneficial impact on direct deposition of faeces and in-flow from riparian areas. Areas near waterways may also be more likely to have heavier soil types with poor drainage, which are associated with waterway contamination. However, if “waterways” are defined in a similar way to the Dairying and Clean Streams Accord, then smaller streams and seeps will be overlooked and the benefits of fencing them will not occur. The draft RPS focus on the effects of stock, including intensive grazing, is supported by this review, and there is evidence that this should include sheep as well as cattle and deer. Sheep do not enter water in the same way as cattle, but their faeces have a high concentration of microbes and are deposited in the riparian area.

Minimising the removal of riparian vegetation is targeted by the draft RPS as it is detrimental to aquatic and terrestrial habitat values. If this policy wording includes avoiding grazing riparian grass, then it could be expected to have some positive effect on microbial contamination. Potentially, replacement of woody riparian vegetation with grass could enhance filtration and die-off due to exposure to UV light, but there is little literature on the effects of different vegetation on microbial contamination.

Other activities not covered by the policies in the draft RPS include the location of "hotspot" areas and effluent management; however there are Permitted Activity rules and RMA provisions that preclude discharges to water. Also, when catchments across the region were compared by Collins (2002a), the number of effluent discharges to land was not strongly correlated with median *E. coli* in rivers, suggesting that irrigated effluent is less important overall in this region than faeces deposited by stock.

In addition to the activities described above, policies to manage pests and to reduce flow rates during rainfall events (e.g. by vegetating headwaters, or by protecting and extending wetlands) could potentially also be beneficial in reducing deposition and transport of faecal microbes. However, there have been no studies to confirm this, and further research is required to ascertain the actual impact and associated risk that wildlife sources pose to human health.

1 Introduction

This report summarises information regarding faecal contamination of waterways arising from rural land. A brief introduction is given regarding the state of knowledge about this issue. This includes the relationship between animals and human pathogens and the applicability of information on sediment movement to faecal organisms. Then, key factors affecting faecal contamination of waterways are examined. These include sources and survival in the environment, transport mechanisms and comparative risk factors. This is followed by information about management practices to reduce faecal contamination. Finally, possible policy directions intended to improve water quality are examined in the light of this review, with a focus on activities in the draft Regional Policy Statement released December in 2009.

1.1 State of knowledge

Faecal microorganisms in rural environments are not as well studied as other contaminants of waterways such as sediment or nutrients. Local conditions produce variable results in terms of survival and transport of microbes, and as yet there are few New Zealand studies from which to draw generalised principles. Relevant international literature is also limited.

A series of reports and research studies on faecal microbial contamination from agriculture was commissioned by the Ministry of Agriculture and Forestry (under the Pathogen Transmission Routes Research Programme, also known as Pathogen Pathways, 2002 - 2005). An outcome of this programme was a publication summarising best management practices (Collins et al. 2007). There are also several recent reports commissioned by regional councils. One field that is still advancing is the development of techniques to ascertain the origin of faecal microbes found in water, and the source attribution for human disease cases. Source attribution is the process of determining the proportions in which the various pathways and sources contribute to the total disease incidence (French and Marshall 2009).

1.2 Gastrointestinal disease

Gastrointestinal illnesses are caused by a range of microorganisms (Table 1), most frequently *Campylobacter,* but also *Giardia*, *Salmonella*, *Cryptosporidium* and less commonly some strains of *Escherichia coli* known as VTEC/STEC - verotoxin or shiga toxin-producing *E. coli.* Rates of gastrointestinal diseases in New Zealand are among the highest in the developed world, particularly for campylobacteriosis (Orchard et al. 2000). High rates also apply to the Waikato region; for example, in 2009 rates for the diseases listed in Table 1 were higher for Waikato than the overall New Zealand average (ESR 2010, p. 60). Although VTEC/STEC *E. coli* caused fewer cases of illness in 2009 than the other pathogens in Table 1, 94% of these VTEC/STEC *E. coli* cases were due to *E. coli* O157:H7, a strain that can cause very serious illness (ESR 2010). Most *E. coli* shed in faeces are harmless but the occurrence of pathogenic VTEC/STEC strains is disproportionately high in rural New Zealand (Till and McBride 2004).

Table 1: Notified gastrointestinal diseases in New Zealand in 2009 (from ESR 2010).

Gastrointestinal disease	Total cases reported in 2009
Campylobacteriosis	7176
Giardiasis	1640
Salmonellosis	1129
Cryptosporidiosis	854
VTEC/ STEC (*E. coli*)	143

In 2009, consuming untreated water and contact with farm animals were among the risk factors for a range of gastrointestinal diseases (see Table 2; ESR 2010). While recreational water contact was a less commonly reported factor, it was still a risk factor across the range of common gastrointestinal illnesses. The Surveillance Report notes that often more than one risk factor is reported and also that a reported exposure was not necessarily the infection source (ESR 2010).

Table 2: Percentage of cases of common diseases that answered 'yes' for selected risk factors (from ESR 2010).

Risk factors (multiple mentions possible)	Gastrointestinal disease			
	Campylo-bacteriosis	Giardiasis	Salmonellosis	Crypto-sporidiosis
Consumed food from retail premises	44.4	26.8	41.9	18.0
Contact with farm animals	43.2	30.8	34.6	55.1
Consumed untreated water	26.4	36.9	28.5	31.7
Recreational water contact	13.6	30.9	18.0	28.8

In a New Zealand study that assessed the relative importance of different possible sources of infection for both *Salmonella* and *Campylobacter,* Mullner et al. (2009) attributed all reported cases of salmonellosis to a food source. For campylobacteriosis the pattern was different. Poultry was the source of 80% of the cases of campylobacteriosis, but the remaining cases were considered to be more likely to have arisen from environmental or occupational exposure than from consumption of food. In New Zealand the *Campylobacter* species responsible for most of the cases of campylobacteriosis reported to the Department of Health is *C. jejuni.* However, when *C. jejuni* is classified at a sub-species level (i.e. by typing), not all types are associated with human disease (Carter et al. 2009).

Campylobacter is frequently found in New Zealand waterways. For example, in one study by Savill et al. (2001), *Campylobacter* was found in 60% of river waters and 75% of shallow groundwaters sampled. A national freshwater microbiology research programme identified that *Campylobacter*, *Cryptosporidium* and *Giardia* were more prevalent in rural waterways than those in urban catchments, suggesting a link with animals (MFE/MOH 2002). The same report noted that *Campylobacte*r was the animal-sourced pathogen most likely to cause human waterborne illness in recreational freshwater users. Genetic typing of *Campylobacter* shows that types present in both cattle and sheep are indistinguishable from those associated with human disease (Gilpin et al. 2008a; Moriarty et al. (in prep (b); French and Marshall 2009).

Establishing the role of waterborne infection in campylobacteriosis illustrates the difficulties in determining pathways and defining risk. Unlike many other enteric pathogens, there is apparently limited spread of *Campylobacter* in families and the main reservoir of the organism is animals. The most common transition route is from poultry meat, but contact with animals, water or drinking unpasteurised milk are also risk factors (Lake 2006; see also Table 2 above). French (2008) used several models to study transmission in Manawatu and attributed poultry as the cause of 52-75% of cases, with contact with cattle the cause of 17-23% of cases. Smaller contributions were estimated to come from sheep, wild birds and environmental water. French also found that most of the ruminant cases could be attributed to direct contact with animal faeces. Epidemiological risk assessment has suggested that about 5% of NZ cases can be attributed to contact recreation in freshwater (Till et al. 2008). But water may also play a role in the cycling of infection among animals and therefore the risk of infection from animal faeces.

In a source attribution study using *Campylobacter* data from the Manawatu region collected over four years, French and Marshall (2009) looked at a total of 624 human isolates, and 766 source isolates grouped into four source categories: Poultry, Sheep, Cattle, and Environmental. These were collected from poultry meat, beef and lamb as well as cattle and sheep faeces, and water collected at natural freshwater swimming locations. The 'Environmental' source refers to the isolates found in the water, some of which may have been attributable to wild birds such as ducks and geese. They used a 'modified Hald model' for source attribution and then estimated the number of cases per month attributable to each of these sources (see Figure 1). The dip in poultry attribution in 2008 corresponded with industry changes, but does not appear to have been sustained fully. If improvements in the poultry industry occur in future, the relative significance of ruminant animal sources would increase (B. Gilpin, ESR, pers.comm. July 2010).

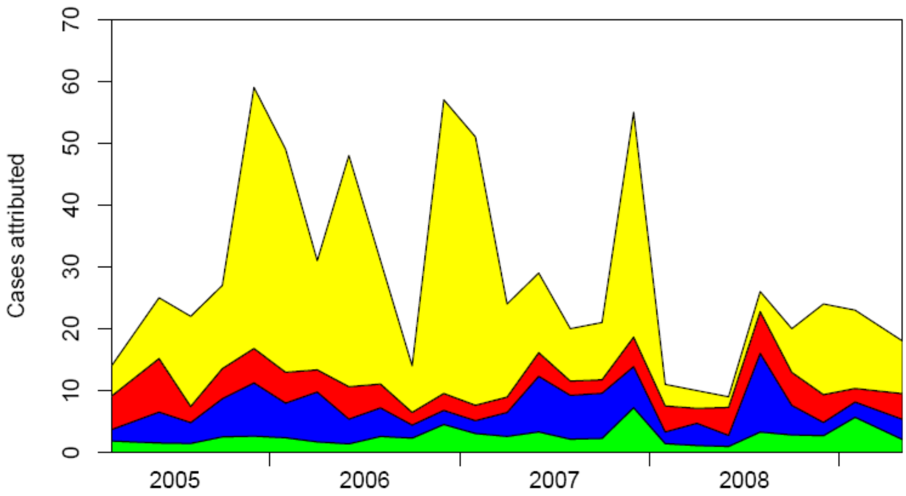

Figure 1: Estimated number of human cases per month in the Manawatu attributed to each source from the modified Hald model – yellow = poultry, red = cattle, blue = sheep, green = environmental (includes fresh water and wild sources) (French and Marshall 2009, p19).

In summary, sub-species types of *Campylobacter jejuni* have been identified from cases of human disease, and identical types have been isolated from sheep and cattle, and less frequently from wild sources. A small proportion of campylobacteriosis cases are currently attributed to waterborne *Campylobacter*. However, contaminated water may also play a role in cycling the disease among animals.

1.3 Water quality microbial indicators

Escherichia coli is a commonly used indicator species for faecal contamination in freshwater as well as in food. There is a long history of use and much data based on this indicator. The properties of *E. coli* that make it suitable for an indicator include its widespread occurrence in the faeces of all warm-blooded animals. This is in contrast to pathogens which are only present when shed by an infected individual, with the presence of one species not being predictive of any other. It is also useful that the numbers of *E. coli* in faeces are high enough to facilitate detection by relatively inexpensive methods.

In the current Waikato Regional Plan, the *E. coli* standard for freshwater is that a median of samples taken over the bathing season shall not exceed 126 MPN *E. coli*/100 ml and no single sample shall exceed 235 MPN *E. coli*/100 ml in a designated bathing area (MPN stands for Most Probable Number). Water quality indicators are also reported on the Environment Waikato website (www.ew.govt.nz). Under these guidelines, excellent water quality is defined as water with <55 E. coli/100 ml, satisfactory 55-550/100 ml, and unsatisfactory >550/100 ml.

While *E. coli* is an indicator of faecal contamination, it behaves differently in the environment to other organisms that cause gastric disease, such as *Campylobacter*. For example, while *E. coli* can readily survive and even grow in the environment, *Campylobacter* cells die off relatively rapidly in water (Sinton et al. 2007a) and in faecal pats of a range of species (Sinton et al. 2007b; Gilpin et al. 2008b; Moriarty et al.; in prep (a); (b)). For this reason detection of *Campylobacter* demonstrates recent faecal inputs, whereas *E. coli* is a general indicator of faecal contamination, which may or may not have been recent. However, not all pathogens die off as rapidly as *Campylobacter*. For example *Salmonella* has similar environmental survival to *E .coli* (Sinton et al. 2007b), and enteric parasites such as *Giardia* and *Cryptosporidium* that form environmentally resistant cysts persist for much longer.

While survival rates of pathogens vary, the comparative survival of *Campylobacter* and the indicator *E. coli* is critical, as *Campylobacter* is the animal-sourced pathogen most likely to cause human waterborne illness in recreational freshwater users (MfE/MoH 2002). Because *E. coli* are more persistent than *Campylobacter* in the environment, they provide a somewhat conservative indicator of risk from this pathogen. However, a moderate correlation has been found between concentrations of *Campylobacter* and of *E. coli* in natural freshwaters, being stronger where *Campylobacter* concentrations are high (Till et al. 2008). The critical value for *E. coli* as an indicator of increased risk of infection from *Campylobacter* is in the range of 200–500 *E. coli*/100 ml (ibid). This is reflected in the threshold for unsatisfactory water quality published on the EW website.

1.4 Comparability of transport patterns with sediment transport pathways

Reported work on sediment and particulate nutrients can provide some guidance for microbial contamination pathways, as some microbes travel attached to soil or dung particles (NIWA 2006a). Collins (2002a) concluded that turbidity can be used as a 'surrogate' measure for microbial contamination within a site, but will not be accurate when comparing sites across a region, as highly erodible soils may not correlate with stock grazing.

Single, unattached bacteria may be similar in size to clay particles (which range from 0.3-2 micrometres, µm) but some bacteria are smaller in size, as are all viruses. Protozoa are larger, including *Cryptosporidium* (4 - 7 µm) and *Giardia* (7 - 10 µm). Even where bacteria fall into the same size category as clay particles, they are less dense and therefore may not settle out in the same way. Their electro-chemical properties also differ, both from soil particles and also from each other, resulting in different degrees of attachment (Schinner et al. 2010).

Attached or clumped bacteria are more likely to behave like particulate nutrients or coarser sediment in terms of settling. However, many bacteria travel unattached. Unattached bacteria are neutrally buoyant in water and less likely to be filtered by vegetation than larger particles (Muirhead et al. 2006a). They are only likely to settle on the surface or be filtered by soil if infiltration occurs. One study found the percentage of attached *E. coli* cells in the run-off from cowpats ranged from 2-26% with an average of only 8% (Muirhead et al. 2005). Later experiments (Muirhead et al. 2006a) confirmed only 9% of *E. coli* were attached to dense particles, and 80% of the *E. coli* cells passed through a 20 µm filter.

As living organisms, microorganisms have biological dynamics that are not shared by inorganic sediment, such as die-off, survival, and even growth in the environment (see Survival and reservoirs, below).

2 Sources of faecal organisms

The information in this section looks at the sources of faecal contamination in waterways, including grazing livestock, which deposit large numbers of faecal microbes in pastoral catchments. Livestock faeces may reach waterways through direct deposition where stock can access the water, through effluent pond discharges, or through transportation by water from paddocks following grazing or effluent irrigation. The information in this section shows that there is a correlation between livestock density and microbial water quality, but that other factors also have an influence, varying from site to site. All warm-blooded animals shed *E. coli*, and wild animal inputs can sometimes cause breaches of water quality standards as measured by this indicator. Because waterfowl defecate directly into the water, they can have a marked effect where they are found in high numbers. Human sources may also be significant in some localities (e.g. where there are poorly-functioning septic tanks or leaking infrastructure). These are not reviewed in this report.

2.1 Direct deposition to water

Direct deposition may occur where stock freely access streams, or at herd crossings. Direct deposition can also occur from wildlife, for example, possums and birds (Donnison et al. 2004).

Dairy cows are more likely to defecate while the herd is crossing a stream (e.g. to the farm dairy) than while on a normal race. In Sherry River in Tasman District, dairy cows defecated 50 times more per metre of stream crossing than they did elsewhere on the raceway (Davies-Colley et al. 2004). There is little published data on defecation frequency for dairy cattle freely accessing water. Collins et al. (2007) reported unpublished data from Bagshaw indicating that while dairy cattle spent only 0.1% of their time in the stream, 0.5% of their defecations occurred there. Overall it appears that dairy cattle defecate at a higher rate when in water than when on land. This is in contrast to the observations made by Bagshaw (2002) of beef cattle freely accessing streams, where the cattle were found to defecate in proportion to the amount of time they spent in streams. In Bagshaw's (2002) study, the beef cattle spent an average of 4% of their time in or within 2 m of the stream, depositing 4% of their daytime average faeces in these areas. Half of this was deposited directly to water (i.e. 2% of total defecation) and half (a further 2%) within the riparian zone (defined as 2 m on each side). Wilcock (2006) reports the figure of 1% is widely accepted for modelling faecal loads in New Zealand, as an estimate of the proportion of total daily faecal material deposited directly to streams where cattle have direct access. Collins and Rutherford (2004) attributed a figure of 8% to direct deposition when modelling the effect of stock exclusion at Whatawhata; however this included defecations on seepage wetland zones. In summary, the few reports available indicate that when dairy cattle can freely access water, they defecate at a higher rate than when on land, and this is more pronounced at herd crossing points. However, beef cattle freely accessing water have not been found to defecate at a more frequent rate in water than on the paddock. Collins et al. (2007) discussed the variation in reported frequency of deposition to streams as being potentially attributable to a range of factors e.g., stream size, ease of access, and the characteristics of the stream bed. In addition to direct deposition to the water channel, faeces are deposited in riparian and nearby seepage areas (high-risk zones for transport to waterways) because stock are attracted to grazing there.

The Sherry River cows caused a peak *E. coli* concentration of 52,000 cfu/100 ml of stream water during their crossing, compared with 300 cfu/100 ml upstream of the crossing (cfu stands for colony-forming units). Similar loads were deposited by the herd on the way to and from the farm dairy, even though there was more pressure and bunching up of the herd on the way to be milked. This suggests that the stress level of the animals, sometimes cited as a cause of increased defecation (DEC 2006), did not influence the water quality impact in this study.

McKergow and Hudson (2007) found that even a single cow pat deposited in small streams could result in recreational water quality guidelines being exceeded. These workers calculated the effect of deposition of a cow pat on different sized streams. For a stream flowing at 100 l/sec, one cow pat could result in an instantaneous concentration of 4500 *E. coli*/100 ml; or in a larger stream of 1000 l/sec flow, a concentration of 450 *E. coli*/100 ml. (This calculation did not account for settling or die-off over time and it assumed complete mixing. Neither did it consider the effect of cattle stirring up and remobilising bed sediment with its faecal reservoir).

These authors also attempted to define a threshold number of cattle accessing streams that would contribute enough faecal coliforms to breach recreational standards. This threshold was dependent on the flow rate of the stream and the concentration of faecal organisms in the cowpats (which is highly variable). Figure 2 shows that at low flow rates (10 l/sec), between 9 and 200 cattle needed access to a stream channel to cause a mean increase in the average 12-hour *E. coli* concentration equivalent to the contact recreation standard of 126 *E. coli*/100 ml (Department of Health 1992). Again this accounted for no settling, stirring or die-off, and assumed the background level was zero. Their conclusion was that "at the *reach scale*, it is very difficult to allow any access to small streams (with consequent low dilution) without exceeding guidelines, at least sporadically" (McKergow and Hudson 2007, p28, emphasis in the original). At a larger (catchment) scale, they concluded that at times, it would be possible to meet guidelines for contact recreation at low stocking rates, in base-flow conditions. However they pointed out the shortcomings of using a simple stock number or stocking rate to predict stream contamination, given the flow dynamics of waterways.

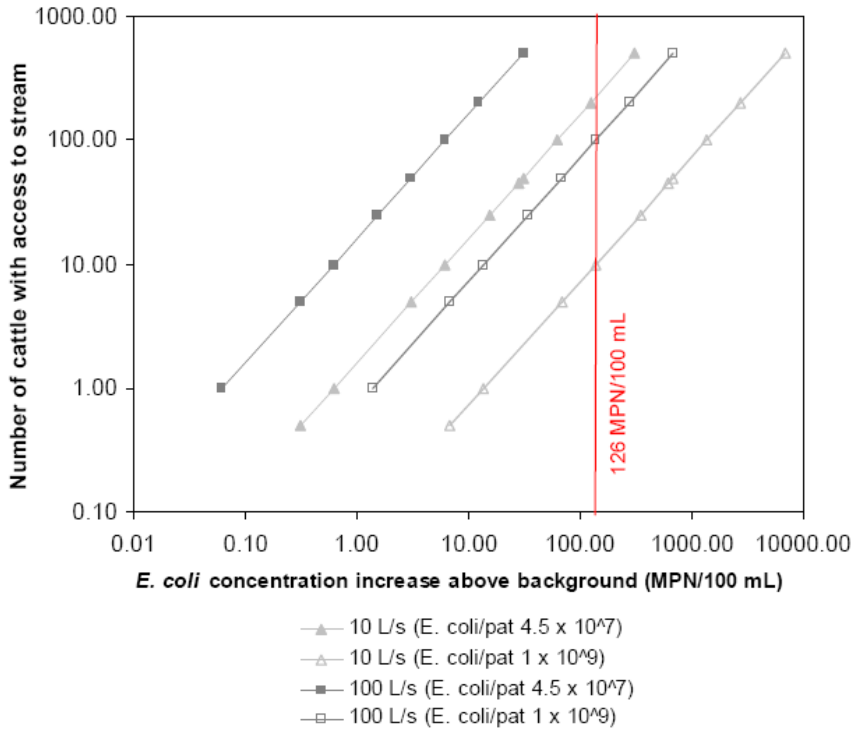

Figure 2: Relationship between *E. coli* concentration and number of cattle with stream access at different flow rates and high and low *E. coli* in cowpats (McKergow and Hudson 2007, p27). Red line is the Department of Health (1992) contact recreation standard, included by these authors for comparison.

McDowell (2006) also reported that direct access had a marked impact on water quality. He found that ten calves given access to an Otago stream created a peak in *E. coli* that was greater than any run-off effect from winter forage crop grazing.

Deer wallows connected to catchment waterways have been found to have a marked effect on water quality, including faecal contamination (McDowell 2009). Davies-Colley and Nagels (2002) sampled *E. coli* upstream and downstream of two deer farms in the Piakonui catchment, Waikato. The *E. coli* concentrations were 2-10 times higher downstream of the deer farms than in upstream reaches which were already impacted by two dairy farms.

When direct deposition occurs at base flows, some faecal material will be removed from the water column through settling. However, in modelling work at Toenepi (Muirhead et al. 2008), access of a dairy herd to the stream at base flow was estimated to result in ten times more input to the stream than the load that would be removed by die-off and sedimentation. They concluded that stock exclusion was essential in order to maintain an acceptable base-flow load to remain within contact recreation standards. Regular stock crossings would also create unacceptable loads. Furthermore, any stock access would cause stirring up of stream sediments, releasing additional microbial contamination to the water column.

2.2 Faeces deposited during grazing

Grazing of pasture by cattle and sheep is linked to faecal contamination loads and increased concentrations of faecal indicators in waterways during rainfall events.

Both cattle and sheep are significant sources of faecal microbes (Wilcock 2006). While all livestock faeces contain microbes, there is considerable variation in the concentrations of individual species including *Campylobacter,* for which shedding varies between individual cows and also over time from an individual cow (Massey University 2007; Gilpin et al. 2008a). Gilpin et al. (2008a) found a *Campylobacter* prevalence of 51% in cows and 65% of calves in their study of 36 Matamata-Piako dairy farms. Moriarty et al. (2008) sampled cattle faeces for *E. coli*, enterococci and *Campylobacter* over four seasons from four New Zealand dairy farms. They found few seasonal or regional patterns, except that average *Campylobacter* counts were consistently higher in spring.

Grazing animals deposit large numbers of faecal organisms, forming a reservoir available for transport during rainfall events. Loads from grazing animals have been estimated based on yields from a heavy run-off event. Experiments at Whatawhata (Collins et al. 2003; Collins et al. 2005) simulated an 8-year return storm event on an 18° slope grazed by sheep. The total *E. coli* load in the outflow collected by a flume showed a contribution equivalent to between 9 x 10^5 and 5 x 10^8 MPN *E. coli*/m^2 of catchment. Repetition of the experiment gave very similar catchment yields of 2 x 10^5 – 6 x 10^8 MPN/m^2 (NIWA 2006a). Peak run-off concentrations ranged between 5 x 10^3 and 7 x 10^6 MPN/100 ml of run-off (Collins et al. 2005). By comparison, total export of *E. coli* from the Toenepi catchment near Morrinsville has been estimated at 1 x 10^{13}cfu/km^2/yr or 1 x 10^7cfu/m^2/yr (Davies-Colley et al. 2008), indicating that one large storm from the hill catchment produced a yield in the same range as the total yield in one year at Toenepi.

Loads and concentrations in run-off are closely associated with grazing history, and have been shown to decline exponentially with time since grazing due to die-off, irrespective of intervening rainfall - see Figure 3.

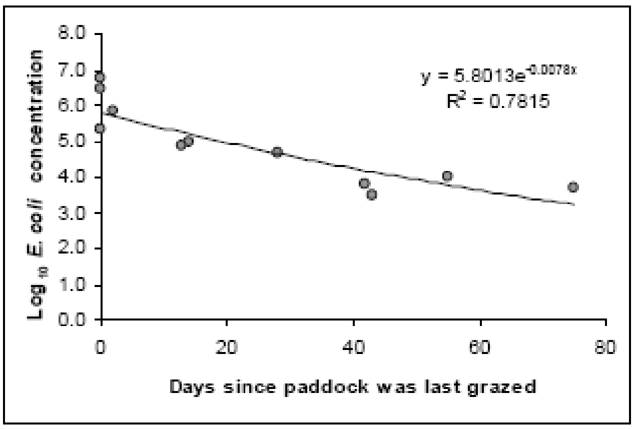

Figure 3: The decline in event-mean concentration (*E. coli*/100 ml) with the time elapsed since the last grazing period - note log scale (NIWA 2006a, p7).

This work confirmed earlier experiments at Whatawhata showing that *E. coli* concentration in outflow can decline 90% after 4-7 days, explained by die-off on the catchment surface (Collins et al. 2003). Results from this sheep-grazed catchment are in Table 3.

Table 3: Decrease in *E. coli* concentration with increasing time since grazing (from Collins et al. 2003)

Time between grazing and rainfall event	Mean concentration of *E. coli* in catchment outflow (MPN/100 ml)
Immediately after grazing	3×10^6
2 weeks after sheep removal	1×10^5
7 weeks after sheep removal	1×10^4
8 weeks after sheep removal	6×10^3

Time elapsed since grazing was the strongest predictor (r^2 = 0.78) for event mean concentration of *E. coli* in a series of eleven of these rainfall simulator experiments on sheep-grazed pasture at Whatawhata (NIWA 2006a). Time elapsed since grazing was also a significant predictor of total *E. coli* load (r^2 = 0.60); while adding total flow to the time-since-grazing predictor slightly increased the r^2 to 0.67. Other variables tested did not contribute to predictive strength for either concentration or load. The variables tested were the number of animals grazed, duration of grazing, and 3-day antecedent rainfall (which varied from 0-70mm).

In their fortnightly samples from Whatawhata streams, Donnison et al. (2004) also found a poor correlation between concentrations of *E. coli* in streams and rainfall in the 24 hours before sampling. They concluded that substantial increases in *E. coli* numbers under high flow conditions may only occur after recent grazing events. In the absence of stock, *E. coli* may progressively decrease with successive high flow events as the catchment sources are depleted (Donnison and Ross 2003). Monaghan and Paton (2004) found that areas where deer had been pacing fencelines had significantly more *E. coli* in run-off than other pasture areas one day after grazing, but not six weeks after grazing.

Even though loads and concentrations decline following grazing, some organisms will survive and persist (see Survival and Reservoirs, below).

Faecal deposition during grazing can also be a source of contamination of sub-surface water. Measuring sub-surface water in a wetland at Whatawhata, Collins (2002b) found a sharp peak in *E. coli* in soil water (down to 80 cm depth) after a grazing event. Collins concluded that faecal contamination of soil water is likely under all pastoral land, and that this may represent an important source of stream contamination in base-flow conditions.

Deposition patterns are not uniform. Collins (2002b) observed that cattle at Whatawhata were attracted to wetlands for summer grazing, and estimated that 75% of the observed faecal material in a paddock was deposited around a wetland. In a subsequent study at Whatawhata (Collins 2004), bulls grazing around the same wetlands in winter showed a similar pattern of excretion. In a neighbouring wetland, though, which was larger and deeper, the bulls did not enter the wetland. Instead, they walked around the perimeter, excreting a disproportionate number of pats within 2 m of the wetland edge. A survey of water quality in 18 wetlands as part of Collins' (2004) study showed that higher concentrations of *E. coli* (over 1000 MPN/100 ml) were associated with recently deposited cowpats on small, shallow wetlands. Other small wetlands had low concentrations of bacteria (5 MPN/100 ml). This indicates that the relative attractiveness of these wetlands to cattle grazing could determine their importance as a source of bacteria to streams.

2.3 Effluent irrigation

Irrigation of liquid effluent may represent a more immediately transportable source of effluent than a cowpat deposited during grazing. In an experiment comparing cowpats and liquid effluent (Collins et al. 2003), microbial concentrations in surface runoff were found to be at least an order of magnitude lower from a cowpat treatment than from a liquid effluent treatment. Over a one-hour simulated rainfall event, a lower percentage of microbes applied in cowpats was recovered compared to those applied in liquid effluent (see Table 4).

Table 4: Recovery rate of applied microbes (%) in surface runoff from 5 m long plots from Ruakura experiments using two different types of faecal material over four trials (Collins et al. 2003, p12).

	Type of faecal material applied	
Microbes recovered	Liquid effluent	Cowpats
Campylobacter	13-89%	<5%
E. coli	16-62%	2-4%

While the initial concentration recovered from the cowpat treatment was low for the first 10 minutes of simulated rainfall, it continued to increase over the 60-minute trial as the cowpat became saturated and broke down. By contrast, liquid effluent rates declined after an initial peak 10-15 minutes into the simulated rainfall (Collins et al. 2003). Therefore, while irrigated effluent provides an immediately transportable form of faecal contamination, manure deposited during grazing is more likely to provide an ongoing source of contamination.

Irrigated effluent can be a particularly important source of contamination on soils underlain by artificial drains, or on poorly drained soils on sloping land (Monaghan and Houlbrooke 2005). Soil properties as they affect microbial transport are considered in detail under Soil types, below.

2.4 Effluent ponds

Wilcock (2006) reports typical treatment pond effluent concentrations of 50,000 *E. coli*/100 ml. Donnison et al. (2008a) measured the concentrations of *E. coli* and *Campylobacter* in discharges from two effluent ponds over a milking season. The average concentrations in the final effluent were 8.1×10^4 *E. coli*/100 ml and 1.5×10^2 *Campylobacter*/100 ml and the average discharge rate was 0.223 l/sec. Bacterial discharge rates from the ponds were calculated as 2×10^8 *E. coli*/cow/day and 2×10^6 *Campylobacter*/cow/day, indicating that ponds discharge substantial numbers of faecal bacteria. As the summer progressed, the discharge of effluent became intermittent and ceased altogether for one pond. Wilcock et al. (1999) also observed that many ponds do not discharge during summer periods of low rainfall.

Wilcock (2006) calculated that ponds discharging to Toenepi stream could create an average annual incremental change in the stream concentrations of 240 *E. coli*/100 ml. Modelling by Monaghan et al. (2008) suggested that two-pond treatment systems used in the Toenepi catchment could contribute approximately half of the faecal bacteria load emitted from those farms. This was a higher figure than that given by Muirhead et al. (2008), who used a different model and estimated a pond system accounted for only 9% of total losses from a typical Toenepi farm. The wide variance in these numbers suggests that a generally accepted figure for the contribution of dairy farm ponds to waterways has not yet been determined. Even with their smaller estimate, Muirhead et al. still found that pond discharges exceeded by 10-fold an acceptable load at base flows to maintain water quality by not exceeding the stream's 'self-cleaning' capacity.

While pond contributions occur at base flows, research in the Toenepi has demonstrated that 95% of *E. coli* exports occur during storm flows (Davies-Colley et al. 2008). This suggests that faecal indicator bacteria from pond discharges during base flows are stored in stream sediments until they are remobilised during storm flows. This topic is the subject of ongoing research interest (R. Muirhead, pers.comm. Sept 2010).

In addition to uncertainty around pond contributions to farm losses, there is little information on the contribution pond discharges make at a catchment or regional scale. Wilcock et al. (2006) found no change in faecal indicator concentrations in the Toenepi stream after the number of ponds decreased from 22 to 9 (a 60% reduction in numbers). It is unlikely that the effect of removing the ponds would have been negated by an increase in cow numbers discharging to the remaining ponds, as the average catchment increase in stocking rate over the same period was only 7% (ibid). This suggests that ponds were not the most significant factor affecting microbial water quality. Collins (2002a) compared Environment Waikato's regular water sampling results with a range of catchment characteristics and found no correlation between numbers of ponds or point source discharges and faecal contamination of rivers in a range of Waikato sub-catchments. This supports the argument that ponds are not a major contributor regionally. At a local or catchment scale, the significance of impact would be influenced by the nature of the receiving waterway, the extent of pond systems, the type of pond and its management.

2.5 Wild animals

New Zealand waterways contain faecal material from wild sources. Recent studies have source-tracked *Campylobacter* in the Toenepi stream and found that of the 25 different strains identified in the water that was sampled, only six were also recovered from cattle in the catchment. Other isolates in the water were associated with ducks, starlings and pukeko (French et al. 2010).

Collins' (2002a) study of Waikato water quality data found that catchments with non-pastoral vegetation did not have zero *E. coli* data in rivers, and this was attributed to wild animal sources. Sampling streams at Whatawhata, Donnison et al. (2004) found that in summer, all streams (draining pasture, pine and indigenous forest) had periods when contact recreation standards were exceeded. This suggests that feral animals can have a significant effect on small streams particularly at low flow.

Wilcock (2006) reported estimates that the mass of faecal material (on a dry weight basis) from one dairy cow is equivalent to that of about 86 black swans. Figures of faecal coliform daily loads from a duck exceed some figures reported for sheep (see Wilcock 2006, Appendix 2). Faecal contribution to streams from waterfowl would be expected to vary seasonally, associated with birds' migratory patterns. Birds are a natural host for *Campylobacter*, having an optimal body temperature for their growth. Ross and Donnison (2003) found high levels of *Campylobacter* in dairy farm soils when seagulls were present on the plots but cows were absent, and attributed these *Campylobacter* to the gulls.

Moriarty et al. (in prep (c)) sampled scats from Canada geese, black swans, ducks and gulls from sites around New Zealand. There was a wide range of counts of *E. coli*, enterococci and *Campylobacter* spp. in individual bird faeces. *E. coli* were present in 95% of the samples, while *Campylobacter* spp. ranged in prevalence from 29% in ducks to 59% in gulls. The ranking for the highest average concentration of indicator organisms (*E. coli* and enterococci) in the faeces of the wildfowl was ducks > gulls > black swans > Canada geese. This order changed for *Campylobacter* spp. with the highest average concentration of the pathogen recorded in Canada geese, followed by gulls, black swans and ducks. Based on the counts in fresh faeces, these authors estimated that although black swans had the highest daily output of faeces (418 g), duck faeces had higher concentrations of the microbial indicators *E. coli* and enterococci, and were likely to have the highest daily outputs per bird for these microbes. By contrast, Canada geese were likely to have the highest daily output of *Campylobacter* per bird at an estimated concentration of 1.21 x 10^6. Because of the differing ratios of indicator species to *Campylobacter* between wildfowl species, these authors suggested that in order to evaluate the risk of *Campylobacter* infection, assessments of water quality using *E. coli* or enterococci need to take into account the particular wildfowl species present. For example, their results suggested that water containing 1000 *E. coli*/100 ml might only contain between 0.1 and 0.001 *Campylobacter*/100 ml water if the faeces were from black swans, gulls or ducks. But it could contain more than 100 *Campylobacter*/100 ml if the *E. coli* were all from Canada geese.

Wilcock (2006) reported that the practice of spreading chicken litter on pasture is receiving attention as a source of pathogens, although New Zealand data confirming this is not available. Researchers in the United States measured concentrations of up to 10^7 cfu/g *Campylobacter jejuni* in fresh poultry litter and noted these bacteria can survive for six weeks in untreated litter (Cook et al. 2006), although composting reduced pathogens to low levels (Macklin et al. 2008). It has also been reported that there is less potential for pathogen leaching if litter is incorporated into soil rather than applied to the surface (Sistani et al. 2010). Although these studies were done in the US, as New Zealand poultry is known to be a major transition route for *Campylobacter* (ESR 2001), it is likely that locally produced poultry litter also contains this pathogen.

Other wildlife may also carry pathogens. Typing of *Campylobacter* isolated from wild bird faeces in children's playgrounds in New Zealand revealed profiles which were indistinguishable from human cases in New Zealand (French et al. 2009). Adhikari et al. (2002) took samples of the rectal contents from a range of animals on a dairy farm. *Campylobacter* was found in all of the species they investigated (Table 5), and was also present in farm drinking trough samples and in urban sparrows, with common sub-types across sources (Adhikari et al. 2004).

Table 5: Prevalence of *Campylobacter* isolated from farm animals (Adhikari et al. 2002. p10).

Type of animal	% positive samples
Dairy cow	53.8%
Farm sparrow	37.7%
Farm rodent	10.8%
Farm fly	8.9%

3 Survival and reservoirs

Some faecal microbes have been found to survive and even grow in the rural environment outside of warm-blooded animals. Reservoirs include cowpats, the soil, wetlands and streambed sediments. Survival patterns differ between disease-causing pathogens. The indicator species *E. coli* and enterococci have been shown to survive and grow in faecal deposits of a range of species, whereas *Campylobacter* die off rapidly (Gilpin et al. 2008b; Moriarty et al., in prep (a)). *Campylobacter* also die off faster than *E. coli* in river water and sea water (Sinton et al. 2007a). The presence of *Campylobacter* in surface water therefore indicates recent faecal contamination.

3.1 Conditions for survival

In soil, unfavourable conditions include temperature extremes, pH extremes, drying conditions, and low nutrient or organic matter content. In water, microbes must survive temperature fluctuations, UV light and predation (Oliver et al. 2005). Bed sediments may shelter microbes from UV light and predators, and also provide a source of nutrients – conditions which may favour survival, particularly at temperatures of less than 15°C (Garzio-Hadzick et al. 2010).

Hot, dry conditions tend to accelerate microbial die-off (Reddy et al. 1981). However, Collins (2004) found that high solar radiation in winter at Whatawhata was correlated with *E. coli* survival. This was attributed to colder conditions associated with the sunny days in that study. Massey University researchers observed a faster drop-off in *Campylobacter* numbers in surface flow compared to sub-surface flow, and attributed sub-surface survival to cool, moist soil conditions (Massey University 2007). Background levels of *E. coli* have also been found to be high in soil in wet weather at Ruakura (Collins et al. 2002). Muirhead (2009) found a large increase in *E. coli* concentrations in soil coinciding with an increase in rainfall over two periods.

Seasonal variation in persistence and growth of microorganisms in faeces have been reported by ESR for cattle (Sinton et al. 2007b), sheep (Moriarty et al.; in prep (b)), and Canada geese (Moriarty et al.; in prep(a)). Enterococci (used in New Zealand as a marine recreational water quality indicator (Department of Health 1992)) showed better survival than *E. coli* in summer and winter. These workers also identified growth in some instances - for example, enterococci grew in all four seasons and were the only organisms to exhibit growth during winter (when sampled after rainfall). In contrast, *E. coli* growth occurred only in summer across the three studies. Growth of *Campylobacter* was not detected in the environment and the longest recorded persistence was nine days. However, no difference was observed between summer and winter conditions when measuring *E. coli* concentrations in a catchment outflow in a Whatawhata trial (Collins et al. 2003). There was also no difference between summer and winter *E. coli* contamination data from a range of Waikato catchments analysed by Collins (2002a).

Muirhead et al. (2005) found no correlation between any weather variable and *E. coli* concentrations measured in cowpats over 30 days. In other studies, the fluctuating moisture content of sheep faeces (Moriarty et al.; in prep (b)), and of cattle faeces (Sinton et al. 2007b) strongly influenced growth, inactivation and re-growth of the indicator microbes *E. coli* and enterococci. Temperature did not have such a strong influence on these indicator organisms. *Campylobacter* did not grow in the faeces of cattle or sheep, and numbers decreased faster at higher temperatures. Moriarty et al. (in prep (b)) also noted that *E. coli* and enterococci survival and growth occurred at lower moisture content in sheep faeces (as low as 30% moisture) than in cattle faeces (growth occurred at 80% moisture and reductions began at 70-75%), an observation that remains unexplained.

In shallow fresh and saline surface waters, the principal factor affecting the survival of faecal bacteria is the level of exposure to sunlight, although there is a contribution from factors such as starvation, protozoan grazing, temperature and salinity (Sinton et al. 2007a). In Sinton et al.'s experiments, *E. coli* inactivation was faster in sea water than river water, reflecting a combined effect of sunlight and salinity. *Campylobacter* were more susceptible to sunlight than *E. coli*, with a 90% inactivation of *Campylobacter* in 1.6 hours in winter, compared to 17.3 hours for *E. coli*.

Stream modelling work at Toenepi estimated that *E. coli* die-off in the stream could occur at a daily average rate of 19%/km (Muirhead et al. 2008). Slow-moving, shallow and clear waters allow more opportunity for die-off. Low numbers of faecal microbes in the Upper Waikato River are attributed to enhanced die-off from solar radiation together with settling in the hydrolakes (Vant 2010).

River modelling work done by Wilkinson (2008) and applied to Motueka also links die-off principally to solar radiation. Secondary components are:

- Water temperature
- Turbidity – reduces light and coats cells with protective clays
- Depth – reduces light

- Velocity – reduces residence time

There can also be a ‘false die-off’ effect when microbes settle in waterways, but are later remobilised (see Streambed sediments, below). Wilkinson (2008) suggests that in dry weather there is minimal transport, water is shallow, less turbid and slower-flowing, and die-off dominates the dynamic variations in faecal indicators. After rain, rivers are deeper and turbid, cloud cover reduces insolation and there is faster travel. In these conditions, die-off is minimised and transport mechanisms dominate the dynamic variations.

3.2 Growth and survival in animal faeces

In a New Zealand study of microbial growth and survival in cattle faeces, Sinton et al. (2007b) found that in the first 1-3 weeks, there were increases in the counts of enterococci (in four seasons), *E. coli* (three seasons), faecal streptococci (three seasons), and *Salmonella enterica* (two seasons), but there was no increase in the counts of *C. jejuni*. Thereafter, the counts decreased, giving an average ranking of the times necessary for 90% inactivation of *C. jejuni* (6.2 days from deposition) < faecal streptococci (35 days) < *S. enterica* (38 days) < *E. coli* (48 days) < enterococci (56 days).

In an American study, Wang et al. (1996) reported recovery of *E. coli* O157:H7 from bovine faeces for up to 70 days at 5°C and 56 days at 22°C. One study in the UK (Avery et al. 2004) sampled faeces/soil after intensive stocking outdoors in November (late autumn). The average survival time of *E. coli* was 134 days and the maximum recorded was 162 days. On average, a 1-log reduction (i.e. a 90% decline in numbers) took 28-38 days. Overall these findings are broadly similar to those from New Zealand.

Muirhead et al. (2005) found that *E. coli* continued to grow in cowpats. In their trials, cowpats aged for 30 days and then exposed to simulated rainfall remained a significant source of *E. coli* in run-off. In most cases, they observed that concentrations in the cowpat on Day 30 were higher than the initial concentration. *E. coli* numbers in the run-off correlated with numbers inside the cowpat. Where cowpats were repeatedly sampled over time under field conditions (Muirhead 2009), *E. coli* concentrations remained high and growth was often observed. Physical decomposition of the cowpat, especially during rainy periods, reduced the size of the cowpat reservoir. However soil background levels remained high. Muirhead concluded that mitigations designed to increase the decomposition rate of cowpats could be a suitable option to reduce the overall paddock cowpat reservoir, but would not solve the issue of the soil reservoir.

The formation of a water-resistant skin on a cowpat may delay immediate release of *Campylobacter*, but also provide conditions for bacteria to survive (Massey University 2007). Sinton et al. (2007b) suggest that sunlight initially assists bacterial replication by warming the pats to optimum growth temperatures and by forming a moisture-retaining crust on the pat. Thereafter, sunlight contributes to bacterial inactivation through pat dehydration. These authors also argue that the effect of rainfall is complex. It leaches bacteria from cow pats, but this may be inhibited by crust formation on the pat. Conversely, rainfall rehydrates pats, slowing inactivation rates and possibly causing re-growth of species such as *E. coli*. Moriarty et al. (in prep (b)) suggested that sheep faeces dehydrate and rehydrate rapidly, compared to a cowpat with a sun-dried skin, which may initially deflect rainfall.

As these studies demonstrate, animal faeces can be a significant source of bacteria for many days after they are deposited onto pasture, although *Campylobacter* are likely to die off faster.

3.3 Survival in soil

In dairy farm conditions, even as cowpat area reduces over time, *E. coli* concentrations can remain high in the soil for six months after grazing (Muirhead 2009). Muirhead calculated that the concentration of *E. coli* in the soil was only ~2 orders of magnitude lower than that in the dung, indicating that the soil was a significant reservoir at a paddock scale.

In trials at Massey, no *Giardia* or *Cryptosporidium* were found in fresh dung, but these organisms were present in outflow after simulated rainfall. This suggested the soil or old dung spots acted as a reservoir for these microbes (Massey University 2007).

Survival of *Campylobacter* and *E. coli* O157:H7 (sourced from laboratory culture) was studied under controlled conditions in two soils, a gley and a sandy loam, from the Toenepi catchment (Donnison and Ross 2009). *Campylobacter* declined faster than *E. coli,* with no differences identified for soil type.

Sukias and Nguyen (2003) investigated whether riparian soils would inactivate *E. coli* faster than non-riparian soils due to differing organic matter content and microbial assemblages. Their study did not show any statistically significant difference between the pasture soils and the riparian soils from two farms (retired for 2 years and 8 years).

Hutchison et al. (2004) studied the effect of either incorporating faecal material into the soil, or leaving it on the surface. They found bacterial decline was significantly more rapid when wastes were left on the soil surface. However, they recognised that this exposure increased the risk of run-off or spread by wildlife.

In summary, soil can retain faecal microbes and provide conditions for prolonged survival. Although survival rates will vary for different organisms, the soil is a reservoir for many species.

3.4 Streambed sediments

Settlement in stream sediments occurs during and after storm events. Bacteria can persist for extended periods and be re-suspended in subsequent events, although die-off of some pathogenic microbes (e.g. *Campylobacter*) will be more rapid than that of the indicator *E. coli*.

Sedimentation was estimated to transfer *E. coli* from the water column to the stream bed at a rate of 10%/km in modelling of the Toenepi Stream using Stokes Law under base flow conditions (Muirhead et al. 2008).

Studies by McKergow and Davies-Colley (2010) suggest that microbes that have settled out into stream sediments are the immediate source of storm pollution peaks in the Motueka River (see Transport, below). They suggested direct deposition and overland flow in smaller events would recharge streambed stores of faecal microbes.

The role of streambed sediments as a reservoir of *E. coli* was also demonstrated by Nagels et al. (2002) who released water from a dam to generate an artificial flood (equivalent to a one-year return period event) in the Topehaehae stream near Morrinsville. This artificial flood with no catchment run-off resulted in similar levels of faecal contaminants to a natural flood event with wash-in from the catchment. These authors concluded the streambed stores were the dominant source of faecal contamination in floods. Three successive floods of similar size were generated with about 60% decline in the magnitude of peak *E. coli* concentration between each flood, demonstrating depletion of microbes in in-stream sediment stores (Muirhead et al. 2004). Nagels et al. (2002) also compared two natural floods and noted that the second flood yielded 25% fewer *E. coli* despite similar water yields, illustrating depletion of catchment or in-channel stores of faecal bacteria in natural flooding events. A further point made was that because under base-flow conditions much of the faecal contamination is in the sediment rather than the water column, recreational activity which disturbs the bed could raise faecal indicator levels that would otherwise be measured as suitable.

Further work in this stream demonstrated that the highest in-stream concentrations of *E. coli* were in the fine-grained sediments associated with cattle crossings (Muirhead et al. 2004). In a United States study it was found that manure-borne *E. coli* survive much longer in sediments than in water, with the best survival in fine sediments, which have high potential for re-suspension during flood events (Garzio-Hadzick et al. 2010).

3.5 Survival in wetlands

Microbe removal in wetlands may occur through die-off caused by UV light (in open water sections). In shady vegetated wetlands, 'dark' processes of predation and settling may be more significant causes of decline in numbers (Stott and Tanner 2005). Microbes removed by entrapment, settling and filtration can be available for subsequent remobilisation.

A richer diversity of predators has been found in wetlands than in ponds in the US (Struck et al. 2006). Predators can include protozoa, amoeba and rotifers (Stott et al. 2003). In Stott et al.'s study of *Cryptosporidium* ingestion, greater prey density prompted higher rates of ingestion. Decamp and Warren (1998) found that a protozoan population of 20 *Paramecium*/ml had the potential to remove 17,760 *E. coli*/ml in 8 hours from wastewater flowing through a reed bed. Stott et al. (2001) showed that ciliates including *Paramecium* have the potential for removing up to 5,000 *Cryptosporidium* oocysts/cell/hr from wastewaters treated in constructed wetlands. The fate of organisms ingested by predators is not clear, although Stott et al. (2003) report that flocculation as they are expelled may make them more likely to settle out even if they survive.

Monitoring has occurred on water exiting from a range of wetlands constructed to treat outflows from tile drainage. Where background microbial levels are reasonably high, wetlands can effect a dramatic reduction. Sukias et al. (2007) monitored an effluent spill as it flowed through a series of two constructed wetlands built to treat sub-surface drainage waters. In-flow *E. coli* concentrations were 1.1 x 10^8 MPN/100 ml in surface effluent flow and 3.7 x 10^6 in sub-surface flow. The median out-flow concentration from the constructed wetland was 528 MPN/100 ml with a maximum of 2500 MPN/100 ml. Thus the wetland reduced *E. coli* by 5-6 orders of magnitude, protecting the Toenepi stream from this concentrated source of faecal bacteria. The authors did not investigate the means of removal, but cited the work of other authors suggesting that possible mechanisms include UV inactivation in open water sections, sedimentation and adsorption to organic matter, microbial predation and natural die-off following entrapment.

Where background levels are low there is more variability in wetland effects, and *E. coli* concentrations may be higher in a wetland out-flow than in the in-flow. Constructed wetlands have complex microbial dynamics due to inputs from wildlife, and show increases in bacterial indicator concentrations in summer (Thurston et al. 2001). Sukias et al. (2006) sampled three constructed wetlands at Toenepi, Bog Burn and Titoki and observed an apparent increase in *E. coli* at two of the sites. At Toenepi, the median concentration at the in-flow was 23 MPN/100 ml compared to 76 at the out-flow. At Bog Burn, in-flow concentrations of 30 MPN/100 ml were recorded and some decrease was observed (out-flow median 15 MPN/100 ml). Results were more variable at Titoki (Whangarei) where the land received effluent irrigation. They noted that achieving levels of *E. coli* below 100 MPN/100 ml may be difficult, as this is a common background level in wetlands accessible to wildlife. As yet no data is available (e.g. by use of genetic techniques) to ascertain whether there is growth occurring in the wetland system, or whether outputs may be from another source such as wild animals or birds (C. Tanner, NIWA, pers.comm. January 2010). Survival and growth may be favoured in conditions found in wetlands - relatively low nutrient concentrations, shade from sunlight and buffering from temperature extremes (Reeser et al. 2007). As unfenced wetlands may attract stock and provide conditions for microbial survival, they can constitute an ongoing source of faecal bacteria to waterways. In their study on a King Country farm, Ross et al. (2010) found higher counts of both *E. coli* and *Campylobacter* in waters draining catchments with extensive swampy areas.

Collins (2002b) measured sub-surface *E. coli* at 50 cm and 80 cm depth in a Whatawhata hill country wetland after grazing. A peak in *E. coli* occurred immediately after grazing (10^2-10^4 cfu/100 ml) and then declined to <100/100 ml over several days. The time taken for 90% of the original concentration to decline (T_{90}) was estimated at 8-17 days. Wetlands were considered to constitute an ongoing source of *E. coli* contamination of streams during base-flow conditions.

Comparison by Collins (2002a) of catchment characteristics with faecal bacteria data collected by Environment Waikato showed a weak inverse relationship between percentage of catchment in wetland and median *E. coli* in the river (R= -0.32). He concluded that large wetlands may be playing a positive role in trapping bacteria.

Clearly, then, both natural and constructed wetlands act as traps for a range of disease-causing microbes, especially when in-flowing concentrations are high. Indicator counts of bacteria in wetland out-flows can be elevated due to wildlife attracted to these areas. But wildlife may pose a lower health risk than livestock sources. Microbes in wetlands are exposed to predation and to UV light in unshaded areas. However, wetland environments may also provide conditions favouring survival of faecal microbes, and become a source for ongoing discharge. Overall, then, wetlands may be useful in reducing concentrated sources of microbes but have a less significant role where concentrations are low. Fencing wetlands will still be important to prevent them from becoming a greater source due to preferential grazing behaviour and defecation.

3.6 Survival in effluent ponds

Meals and Braun (2005) in assessing US effluent ponds found that storage for 30 days reduced *E. coli* counts in the pond effluent by 99%, as long as new sources of microbes were not introduced in that time. However, because bacteria numbers in the influent to a conventional effluent pond are very high, even when treatment causes a decrease of two orders of magnitude, the outflow can still represent a significant discharge (Donnison et al. 2008a).

3.7 Stand-off pads

Luo et al. (2007) studied the effectiveness of two different *Pinus radiata* wood products in trapping and holding faecal bacteria deposited on a stand-off pad. Significantly more *E. coli* were recovered in bark pad drainage than in sawdust pad drainage, a finding later confirmed in a laboratory study (Donnison and Ross 2008). Luo et al. (2007) estimated retention of *E. coli* to be 99.7% in the sawdust pad and 90.2% in the bark pad. Although no statistically significant difference was found for *Campylobacter*, the results followed a similar pattern to that of *E. coli*. Some *E. coli* (but not *Campylobacter*) continued to be present in drainage liquid up to nine months after animals were removed.

4 Transport and entrapment

Flow pathways and rainfall events are critical for microbial transport. The driving force behind pollutant transfer from land to water is the hydrology, because water provides the energy and the carrier for pollutant movement (McKergow et al. 2007). Preferential flow, both over the surface and through the soil, can rapidly move large loads of faecal microbes into waterways. The efficiency of entrapment via soil filtration or grass buffer strips varies with site conditions and flow characteristics.

4.1 Rainfall events and flow

Flow and faecal bacteria levels were positively correlated in fortnightly samples taken over a year at Toenepi and nearby Topehaehae streams (Donnison et al. 2006). However, over two years at Whatawhata, Donnison et al. (2004) did not find a strong relationship between flow and *E. coli* concentrations in streams.

Storm flows have been estimated to contribute 95% of the total faecal pollution loads in the Toenepi stream (Davies-Colley et al. 2008). Similarly, modelling in Motueka estimates that 98% of all faecal load exported to the ocean travels there during rainfall events (McKergow and Davies-Colley 2010). These authors suggest that *E. coli* are mobilised from the river bed by accelerating currents as water flows rise in a storm event. This explains why *E. coli* peaks occur before sediment and flow peaks in this river, as sediment is transported further from eroding parts of the upper catchment. In smaller catchments, *E. coli* and turbidity peaks may be more synchronised.

Stott et al. (in press) measured flow events at Toenepi and found the bacterial load transported downstream was three orders of magnitude greater under storm flows than at base flow (storm flow was defined as times following rainfall events when flow increased above the base-flow level). They also found that *E. coli* peak concentrations occurred close to the turbidity peak and ahead of the *Campylobacter* peak, which coincided with peak water flow. They attributed the *E. coli* peaking before *Campylobacter* to remobilisation of in-stream sediment *E. coli* sources, while *Campylobacter*, which does not survive well in the environment, arrived in run-off carrying material from fresh faecal deposits. *E. coli* survives well in sediments and is re-suspended by turbulence (Nagels et al. 2002). Therefore, when rainfall increases stream flow, there is an associated increase in bacterial concentration sourced from both in-stream stores and wash-in from land stores. Conversely, at base flow, faecal microbial input from pastoral animals occurs by direct deposition, from irrigated or discharged dairy effluent (Muirhead and Monaghan 2010), or through ongoing sub-surface discharges from saturated areas (Collins 2004).

4.1.1 Subsequent events

In experiments at Ruakura (Collins et al. 2002), the first simulated rainfall event after applying a source of effluent released a rapid flush of bacteria into overland flow. Further simulated events 5 and 12 days later showed a marked (3-4 orders of magnitude) decline in *Campylobacter*. However, the authors noted that the applied source was liquid effluent, and these results might not hold for a manure source which could be expected to release microbes over time. In a subsequent article (Collins et al. 2004), it was noted that the remobilisation measurements were done after a high-flow initial event, which may have left few microbes entrapped and available for remobilisation.

Davies-Colley et al. (2008) found that there was not always a consistent relationship between the magnitude of storm flow and *E. coli* exports in the Toenepi catchment. They suggested this could be due to differences in the intervening period between flood events, determining the extent of build-up of catchment stores. At Whatawhata, Donnison et al. (2004) also noted that *E. coli* numbers tended to decrease when there was repeated rainfall over a short time, which supports the concept of depletion of stream and land stores.

Management options to limit storm in-flows of faeces deposited during grazing are limited. However, to ensure that successive flood events are not accompanied by successive large peaks of microbial input into waterways, it would be desirable to remove stock from areas in close proximity to waterways during prolonged heavy rainfall, especially on heavier soil types.

4.1.2 Flow affecting entrapment

In experiments at Ruakura, different flow rates were trialled on four replicate plots to observe the effect on microbe recovery or entrapment (Table 6). At low flow, <5% of applied microbes were recovered from the bottom of a 5 m grass strip representing a riparian buffer. At high flow, there was much more variance in attenuation across the plots, with 15-100% of microbes recovered (NIWA 2006a; Collins et al. 2004).

Table 6: Recovery rate of microbes (percentage washed through four replicate 5 m grass strips) as affected by flow rate in Ruakura trials (after Collins et al. 2004, p568).

	Low flow (4.0-6.6 l/min)	Intermediate flow (10.2-10.6 l/min)	High flow (13.0-13.3 l/min)
Recovery rate of *E. coli* (%)	<1-4	16-62	41-100
Recovery rate of *Campylobacter* (%)	<1-5	13-89	15-51

A similar finding was made by Muirhead et al. (2006a), who used experimental field plots 5 m long. Subjected to a flow of 2 l/min, 27% of *E. coli* was removed after 5 m, but at higher flow rates of 6 and 20 l/min, no attenuation trend was observed. This suggests that at times of low overland flow, riparian strips 5 m long can effectively attenuate faecal bacteria, but this will not necessarily apply under high rainfall, when riparian buffers are unlikely to be effective.

Similarly, Collins (2002b) found very little attenuation of faecal microbes in sub-surface water occurred in hill country wetlands at high flows. At low flows, wetlands could attenuate faecal contamination by an order of magnitude. However, wetlands were likely to retain some bacteria and these could subsequently seep out in sub-surface flow, being slowly released to streams.

4.2 Topography

Topography affects run-off events and transport pathways, as well as riparian effectiveness.

In their review, NIWA (2006a) report on a sediment study by Dillaha et al. (1988) where slope varied and other factors were kept constant. An inverse relationship was found between slope (6-9°) and sediment entrapment (50-90%). No trial of this sort has been done for microbes on different slopes. In New Zealand studies (NIWA 2006a) of microbes on an 8° slope there was evidence of entrapment in Tirau and Hamilton at low and moderate flows, but variation in slope was not trialled.

Hill country

Steep slopes create convergent flows of high-velocity run-off and are often overlain by shallow soils with little attenuation potential. However, slopes with allophanic or pumice cover have higher infiltration and therefore higher filtering potential. This suggests that run-off may be less of an issue for free-draining soils, and preliminary laboratory research supports this (Donnison and Ross 2009).

Rolling country

Moderate slopes generate run-off, and where this is in sheet flows it is ideal for riparian filtration. Riparian efficiency will depend on soil types, flow rates, filter width and by-pass flows through the soil.

Flat country

Flat country is less likely to generate run-off and may be artificially drained, making riparian filters less useful (NIWA 2006a). However, even artificially drained soils can have surface run-off. In one study at Massey University on a Pallic soil, researchers measured 46 mm of surface run-off and 258 mm of sub-surface drainage in 2003, while in 2004 the corresponding figures were 179 mm and 388 mm respectively (NIWA 2006b). Interpreting results from the Ruakura trials on 5 m long strips, the authors suggested that such strips could have trapped most of the bacteria in the surface run-off at the Massey site.

Where flat land is artificially drained, there may be preferential flows of liquid effluent from irrigation or from rainfall following grazing. This is exacerbated by soil cracks creating by-pass flows (see Soil types, below).

4.3 Soil types

Different soils have varying capacity to filter microbes, related to infiltration and drainage properties. Important risk factors for the transport of faecal microbes include both the soil type and the underlying material in the vadose zone (the layer immediately below the soil and above the groundwater).

Poorly drained soils generate run-off and if flow rates are not excessive, there may be opportunities for filtering and entrapment in riparian areas. However, high by-pass through cracks is also likely. By-pass flow occurs when microbes carried by water are transported through continuous large pores or cracks in the soil with minimal interaction with the soil matrix (McLeod et al. 2008). Cracks may occur naturally in clay soils or be created mechanically (e.g. when installing artificial drainage (NIWA 2006b)). Poorly drained soils are correlated with degraded microbial water quality in Waikato catchments (Collins 2002a). Collins suggests that these soils not only have greater run-off and susceptibility to trampling, but they have more rapid peak flows that can release faecal material within stream channels. These soils also show less effective retention of bacteria (Donnison and Ross 2009).

Artificial drainage may speed the transport of microbes to waterways, and soils above these drains often exhibit cracks. Ross and Donnison (2003) found soils with mole and tile drainage rapidly transported faecal microbes to drainage water. In their Otago trial, when by-pass flow of irrigated effluent occurred through the mole and pipe network, concentrations of *Campylobacter* in the drainage water were similar to those in the applied effluent, indicating no filtration had occurred.

Free-draining soils have high infiltration and are less likely to generate run-off. Alluvial soils from young, loamy materials and soils from allophanic or pumice materials are effective soil filters because their pore structure encourages matrix flow, where the flow of liquid in these soils is spread throughout the soil particles, as opposed to by-pass flow through heavy soils. Pumice soils can become hydrophobic under dry conditions with potential run-off of applied effluent. However, when effluent is applied at recommended low rates (Houlbrooke et al. 2004) this is unlikely to be a problem (M. McLeod; Landcare Research, pers. comm. September 2010), with attenuation through the soil matrix likely.

While alluvial soils are generally effective filters, those of gravel or shallow sand without much silty matrix have low attenuation potential because the free draining pores of these soils permit liquid to flow rapidly downwards without effective filtering. Groundwater aquifers below these materials and below fractured volcanic rock, karst or very coarse pumice may be at risk.

Experiments were conducted at Ruakura (Collins et al. 2003) in a Hamilton clay loam soil rated high for by-pass flows due to cracks and a well-developed structure. At simulated high flows, sub-surface flow represented 4-24% of the total outflow from the plots and accounted for 1-13% of the total *E. coli* outflow numbers and 1-24% of the total *Campylobacter* outflow numbers. In total, only around 30% of the applied water was recovered, implying losses to deeper soil zones.

Comparison was then carried out at Tirau on an allophanic soil (NIWA 2006a). At the same fast flow rate, there were no conclusive differences in recovery of microbes (Tirau rates were 40, 41, 47 and 96% over four trials while Hamilton rates were 41 and 100% over two trials). However there were clear differences in flow. No sub-surface flow occurred in the Tirau soil and there was a lower total outflow, indicating liquid was held in the soil matrix (NIWA 2006a). This implies that in the Tirau soil there would be no sub-surface by-pass flow and water would flow down through the soil matrix with opportunities for microbe attenuation.

This was shown in a comparison of four Waikato soils (Aislabie et al. 2001) where farm dairy effluent was applied at 50mm/hr followed by simulated rainfall. Two of the soils (Waihou and Atiamuri) were free-draining with a uniform porous structure that created a matrix flow. These soils filtered the effluent much more effectively than the heavier soils (Te Kowhai and Netherton) which had a course structure and promoted by-pass flow. McLeod et al. (2003) found that up to 10% of applied faecal coliforms in effluent were transported into the drainage liquid from poorly and imperfectly draining soils, but <1% were lost from freely draining soils.

The importance of soil type for pathogen transport was confirmed by Donnison and Ross (2009), who studied movement of *Campylobacter* and pathogenic *E. coli* O157:H7 in two contrasting soils from the Toenepi catchment: a gley and a sandy loam. For the gley soil, large numbers of bacteria were transferred to drainage and run-off by simulated rainfall at moderate (25 mm/hr) and heavy (50 mm/hr) rates, particularly within 14 days of bacterial application. The relative rate of transfer of *Campylobacter* to drainage water decreased with time but that of *E. coli* O157:H7 did not. The authors concluded that rainfall could transfer bacteria from the gley soil for at least 28 days after deposition and that although far fewer bacteria are washed out than retained, the actual numbers in the outflow can still be large. In contrast, for the sandy loam soil, rates of retention were high, and increased over 28 days. The authors suggested that where possible, the grazing of gley soils bordering streams should not occur under wet conditions, and preferably, effluent irrigation should also be avoided on these soils.

Aislabie et al. (in press) reached a similar conclusion about the risk of effluent irrigation on heavy soils. They demonstrated rapid transport of microbes to depth in a well-structured Netherton clay loam soil following application of 25 mm dairy farm effluent at 5 mm/hr. They also reported that *E. coli* continued to leach from this soil under natural rainfall for up to three months after application of effluent, regardless of soil moisture conditions. In contrast, they found that the potential for leaching from fine-structured Manawatu sandy loam depended on soil moisture conditions. The only time *E. coli* was detected in leachate from this soil was in late winter when rainfall was frequent and the soil was likely to have been wet. This suggests that while the message to avoid irrigating wet soils holds for lighter soil types, on heavy well-structured clay soils, extreme care is required when irrigating dairy effluent in all moisture conditions if by-pass flow is to be avoided. Aislabie et al. (in press) suggest that alternative sites or systems (e.g. advanced pond systems) might be preferable to irrigation on high risk soils. If effluent is irrigated onto soils with high risk of by-pass flow then low volumes should be applied at low application rates.

Another study tested virus movement through contrasting soil types (McLeod et al. 2001). The virus was filtered out as it moved through pumice and allophanic soils, but moved rapidly through a gley and a recent soil, due to by-pass flow. Protozoa (e.g. *Giardia, Cryptosporidium*) are more readily retained in the soil matrix as they are larger than bacteria, while viruses are smaller.

Soil risk maps have been prepared for microbial contamination (McLeod et al. 2005). These are based on the following factors:

- Soil properties and run-off potential
- By-pass potential
- Transport through the vadose zone

These considerations are then combined to estimate the risk of transport to:

- Surface water – combined risk of run-off and by-pass flow
- Groundwater – combined risk of by-pass flow and flow through the vadose zone

A map showing soils at risk of preferential flow in the Waikato Region is shown in Figure 4.

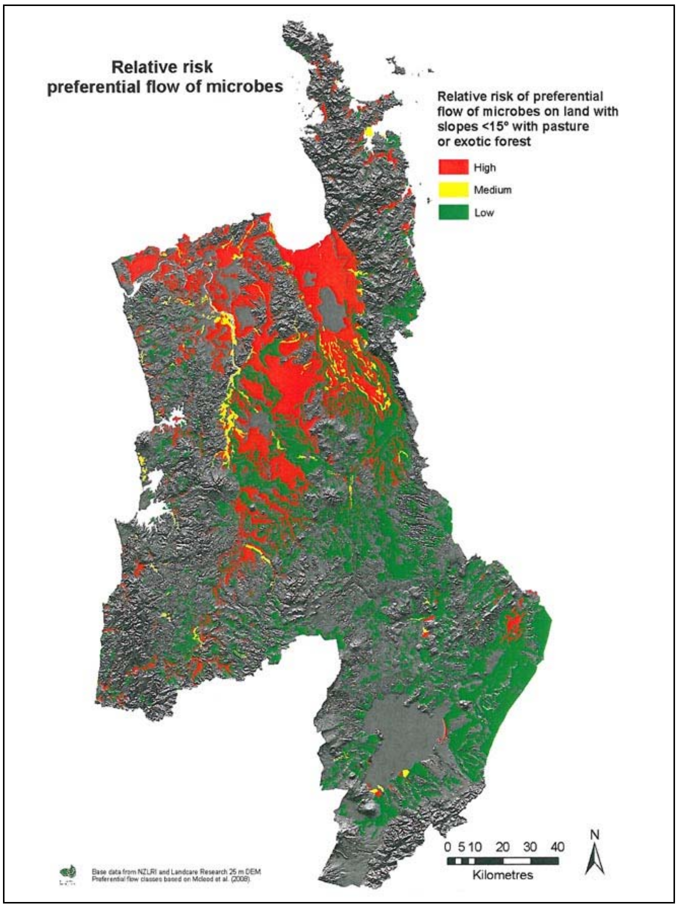

Figure 4: Relative risk for preferential flow of microbes in pastoral farm or forest land in the Waikato region (supplied courtesy of M. McLeod, Landcare Research, prepared for EW under contract). Note that areas in grey are for land of >15° slope or not in pastoral farm or forest use.

Soil treading damage can increase the risk of run-off, especially on heavier soils. In addition to soil type, treading damage depends on stock type and rate, animal behaviour, and soil moisture.

4.4 Sub-surface flow

Connolly et al. (2004) observed little difference in *Campylobacter* concentrations in drainage water compared to surface run-off at comparable sites in measurements done under wet conditions. This suggests that flow through some wet soils occurs with limited attenuation. By contrast, Ross and Donnison (2006) applied effluent to four soils with moisture levels below saturation, and found that when simulated heavy rainfall was applied 4 and 11 days after dairy effluent application, only about 1% of the applied *C. jejuni* were recovered in leachates. Their findings suggest that if effluent is applied to soil that is not saturated, a high proportion of the applied Campylobacter can be held by the soil.

Collins (2004) observed storm events at Whatawhata and found that surface run-off was apparent for only a short duration. He considered sub-surface flows were an important pathway for bacterial movement into large hill country wetlands, and therefore mitigation options are limited.

Sub-surface flow can also affect shallow groundwater and springs. Spring water at Pigeon Creek was found to have elevated *E. coli* concentrations (197 MPN/100 ml) and this was attributed to stock grazing nearby (Wilcock et al. 2007).

Similarly, elevated *E. coli* in tile drainage water on an Otago deer farm was attributed to stock congregating around a self-feeding silage pit in the paddock (McDowell et al. 2006).

4.5 Laneways, tracks and yards

Laneways result in a concentration of manure on surfaces that often have high connectivity with waterways at crossing sites.

Smith and Monaghan (2009) collected run-off from laneways on Southland dairy farms during rainfall events and measured the concentration of *E. coli*. This was found to approximate that of raw dairy effluent at sites close to the farm dairy (mean 8.5×10^5 MPN/100 ml), and was only slightly lower at bridge crossings some distance from the farm dairy (mean 1.0×10^5 MPN/100 ml). In winter, when cows were transported off-farm, there was only a marginal reduction in concentration (mean 6.3×10^4/100 ml near the farm dairy and 3.2×10^4/100 ml at the bridge crossings), indicating persistence of *E. coli* over time. On a concrete lane surface, 91% of the rainfall was collected as run-off, whereas on porous fine surfaces, 50% of the rainfall was collected. In spite of the high concentration of effluent, these authors judged that laneways on these farms would not constitute a major source of contamination, due to their small area. A field survey showed the laneways represented only 0.55% of the total catchment, typically 1.2 ha for a 213 ha dairy farm. However, it was difficult to quantify the total area of lane discharging run-off directly to streams. Their field survey assessed the laneways discharging directly to represent 4.3% of the total laneway area. They calculated that if 5% of laneway areas were discharging directly to a stream, this would represent less than 3% of the annual whole farm discharge of *E. coli*, and in a worst-case scenario if all the laneways discharged directly to streams, this would potentially represent 12% of total *E. coli* discharges from the farm.

Other hard or compacted surfaces may also be important. On Scottish farms "hardstanding" areas such as cattle yards were identified as generating runoff with high concentrations of contaminants, including faecal bacteria, even during relatively light rainfall (Edwards et al. 2008). In New Zealand conditions, Wilcock (2006) calculated the amount of *E. coli* that would accumulate on a stand-off pad and found this could reach 1×10^{12} *E. coli*/hectare of pad/day. While pads are not generally large in size, they represent a concentrated source of effluent and any drainage from a pad is a potential source of localised contamination.

4.6 Drain transport and attenuation

Both surface and sub-surface drains can be pathways for transport of faecal contaminants. Mechanisms include cattle access into open drains, run-off from paddocks of dung or irrigated effluent, and by-pass flow to sub-surface drains (Wilcock 2006).

Faecal organisms may be removed from drainage water in open drains by becoming attached to plants or settling out. Nguyen et al. (2002) introduced faecal matter into a drain and found a 40% reduction in *E. coli* concentration over a 40 m distance and close to 100% decline over 110 m of drain. The water velocity was slow (0.3 m/minute) which would favour sedimentation. Drain sediments could act as a temporary storage reservoir, with later re-release (see Streambed sediments above). However, for those drains that do not contain water all the time there could be an opportunity for bacterial die-off during dry weather.

Deposition and subsequent re-suspension within drains is highly dependent on conditions, especially flow rate and disturbance (e.g. by stock in the water) (Entry et al. 2000). Other survival factors are also relevant such as sunlight and predation. Nguyen et al. (2002) noted that while drains could effectively attenuate the faecal material from accidental access of dairy cows to a drain, further research is required to determine the extent of attenuation under different flows, rates of faecal input and drain vegetation conditions from those in their study.

5 What are the most important factors affecting faecal contamination?

5.1 Multi-variate analysis comparing Waikato catchments

Multi-variate analysis was carried out by Collins (2002a) seeking to explain the variance in microbial contamination data from a range of Waikato sub-catchments (see Figure 5). Three key factors were found to explain 68% of the variance:

- The percentage of the catchment with poorly drained soils (Figure 5c)
- Median turbidity levels at the catchment outlet (Figure 5k)
- Cattle stock units in the catchment (Figure 5a)

There was a moderately strong linear relationship (R = 0.69) between median *E. coli* concentration and the percentage of a catchment characterised by poorly drained soil (soil drainage Classes 1 and 2). This was attributed to these soils being prone to compaction, generating more run-off, and creating higher peak flows which could re-suspend entrained faecal material in water bodies. It was suggested that artificial sub-surface drainage might also play a role, but no data were available to explore this.

Cattle density gave a slightly stronger relationship than general stock density (R= 0.58 for cattle; R =0.54 for stock in general). This was attributed to the preference of cattle for spending time in waterways. Stock density had a somewhat stronger relationship to faecal contamination data than percentage of catchment land cover in pasture (R=0.48).

The percentage of steep slopes in a catchment was *negatively* related to median *E. coli*, attributed to the fact that catchments with a high proportion of steep slopes were not in pastoral use. This gives an indication that feral animals as they occur naturally in bush have less impact than pastoral animals at average stocking rates.

The association between faecal contamination and riparian access could not be assessed due to a lack of catchment data.

Neither dairy nor non-dairy farm point source discharges were found to be strongly correlated with median *E. coli*, although one catchment with exceptionally high volumes of non-dairy point source discharges also had one of the highest concentrations of *E. coli*. The number of discharges of effluent to land (from dairy, non-dairy agriculture and treated sewage waste water) did not correlate strongly with bacterial water quality. (Note that while volume of dairy point sources was assessed (Figure 5i), data on the volume of effluent discharges to land was not available, only the number of discharges to land, limiting the ability to assess these relationships.)

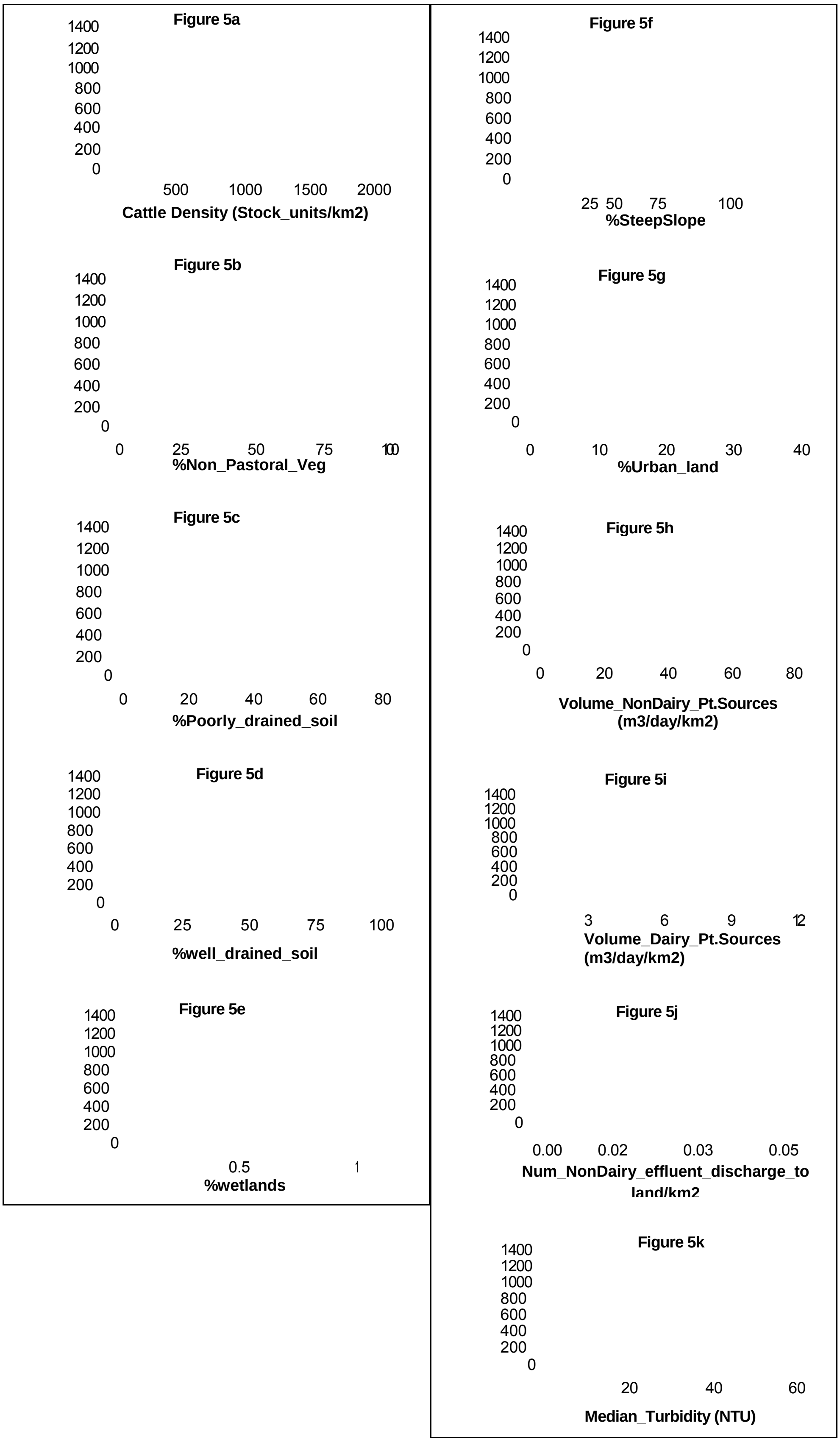

Figure 5: Relationships between environmental factors and median *E. coli* concentrations in catchment water samples. (Collins 2002a, p22)

5.2 Determining the relative importance of different sources

Wilcock (2006) used data reported across a number of studies to calculate loads of *E. coli* and compare their relative significance.

5.2.1 Stock type

Some of the data used by Wilcock (2006) to determine *E. coli* loads for cattle and sheep and the average daily loading to pasture are given in Table 7.

Table 7: Comparative faecal organism loadings from cattle and sheep (after Wilcock 2006, p4).

	Cattle	Sheep
Faeces excreted per day (wet weight)	28 kg	1 kg
Faecal microbe content (per gram)	4.5×10^4 faecal coliforms (of which ~90% are *E. coli*)	5×10^6 *E. coli*
Average daily output	1.2×10^9 *E. coli*/cow/day	5×10^9 *E. coli*/sheep/day
Average daily loading to pasture	3.6×10^9 *E. coli*/ha/day (stocked at 3 cows/ha)	2.5×10^{10} *E. coli*/ha/day (stocked at 5 sheep/ha)

The data in Table 7 indicate that while sheep excrete less weight of faeces than cattle, the concentrations of faecal organisms can be higher in sheep faeces. It should be noted, however, that studies have shown bacterial counts in cattle faeces are highly variable (e.g. Moriarty et al. 2008; Muirhead et al 2006b; Donnison et al. 2008b). Moriarty et al. (2008) measured median numbers of *E. coli* in cattle faeces at 5.9×10^6/g wet weight, similar to the figures for sheep cited above from Wilcock (2006). Muirhead et al. (2006b) reported *E. coli* counts in cattle faecal samples over a 13-month period ranging from 9.7×10^1 to 1.9×10^7 MPN/g dry weight with a geometric mean of 2.1×10^5 MPN/g. Given a water content of 88% in this study, wet weight equivalent concentrations would be an order of magnitude lower. Moriarty et al. (in prep (b)) found that concentrations of *E. coli* in sheep manure were equal to or higher than those in an earlier study of cattle faeces by Sinton et al. (2007b) of 10^5-10^6 CFU/g dry weight. The water content in the Moriarty et al. study was 57%.

Donnison et al. (2008b) measured the entire daily output of *E. coli* in a small but highly controlled study of eight cows fed on pasture silage. In this study *E. coli* were very variable and not detectable at all in 14% of samples. The authors postulated that there could be a relationship between diet and faecal bacteria shedding. Avery et al. (2004) in a trial with penned animals found that cattle faecal material sampled had significantly greater numbers of *E. coli* than sheep faecal material on some sampling days, but the pattern was not consistent. Overall, *E. coli* concentrations in freshly deposited faeces from cattle, sheep and pigs were similar.

While figures in a similar range have been reported on a per-hectare basis for typical cattle and sheep stocking rates, block dairy grazing may see short-term stocking rates of 400 cows/ha (i.e. more than 100 times higher than the farm's average rate). Wilcock (2006) calculated that based on block grazing of dairy cattle for two months of the year, *E. coli* deposited per hectare per year in a block-grazing scenario could be thirty times that of typical grazing at 3 cows/ha.

Wilcock (2006) concluded total annual loads to waterways are similar from hill country sheep and beef farms and low-gradient dairy farms (~10^{11} *E. coli*/ha/yr). He also reported various studies showing that surface run-off concentrations from hill country (sheep/beef) and dairy catchments fell within a similar range (10^3-10^7 *E. coli*/100 ml). Similar findings were reported by McKergow et al. (2008), who noted run-off from sheep and beef pasture with similar *E. coli* concentrations to that from dairy land.

It is clear from the reported studies that concentrations of bacteria are highly variable in the faeces of individual animals, both within and across animal types. However, variability between individual animals and species does not result in marked differences in catchment loadings, with broadly similar reports across stock types.

5.2.2 Wild animals vs stock

The significance of contamination from different sources is site-specific. Work has been done in some sites to trace faecal bacteria back to the source animal. For instance, studies in the Heathcote-Avon estuary found a predominance of wild bird and dog sources (Moriarty and Gilpin 2009). The lower Maitai River in Nelson was found to have both ruminant and human sources (Kirs et al. 2008). Seawater sampling in Marlborough found indicators of wildfowl, human, animal and possible possum material (Gilpin and Tiernan 2008). An investigation of the Waimoku stream in Taranaki (where duck ponds have been created on the stream) has suggested wild birds as the likely contributor to breaches of water quality standards in the stream and nearby ocean water (Taranaki Regional Council 2010).

Source tracking of faecal contaminants is a developing field. Tests can reveal the range of sources, but do not currently allow a proportion of contamination to be attributed to each source (C. Cornelison, Cawthron Institute, pers.comm. July 2010). A range of evidence needs to be reviewed in order to determine the likely significance of sources, including site surveys.

Catchments in the Waikato region without grazing stock generally have lower concentrations of *E. coli* in water samples (Collins 2002a), suggesting that wild animals occurring naturally in bush habitats produce less faecal contamination than livestock at typical densities (see Figure 6).

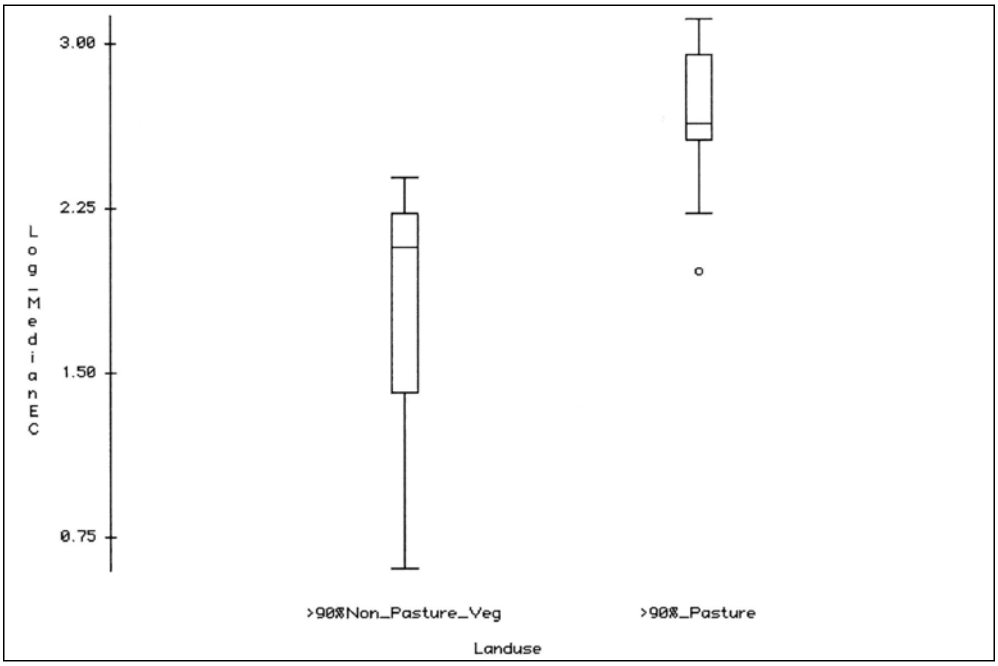

Figure 6: Boxplots illustrating log transformed median *E. coli* statistics from Waikato catchments with >90% non-pastoral vegetation and those with >90% pastoral vegetation. Outliers plotted as a circle. (Collins 2002a, p13)

Collins (2002a) concluded that the "pattern of contamination across the Waikato is dominated by the presence of grazing livestock and the highest median *E. coli* concentrations are associated with the most intensive dairy farming in the centre of the region. Conversely, the lowest median values are found in forested catchments, although *E. coli* concentrations are always measurable, indicating contamination by wild animals" (ibid, pg i). However, in Collins' analysis, livestock density only accounted for around a third of the variance in microbial water quality across Waikato catchments, as shown by the R^2 values of 0.29 for stock density and 0.34 for cattle density (R^2 indicates the percentage of variance explained by a certain factor). This suggests that in addition to livestock density, other factors also have an important influence on water quality.

Young et al. (2005) compared water quality in sub-catchments of the Motueka River with different land uses and found that pasture catchments had significantly higher *E. coli* and *Campylobacter* concentrations than either native forest or pine (Figure 7). Horticultural catchments had the highest levels of *Campylobacter*, attributed to septic tank leakage from horticultural workers' accommodation.

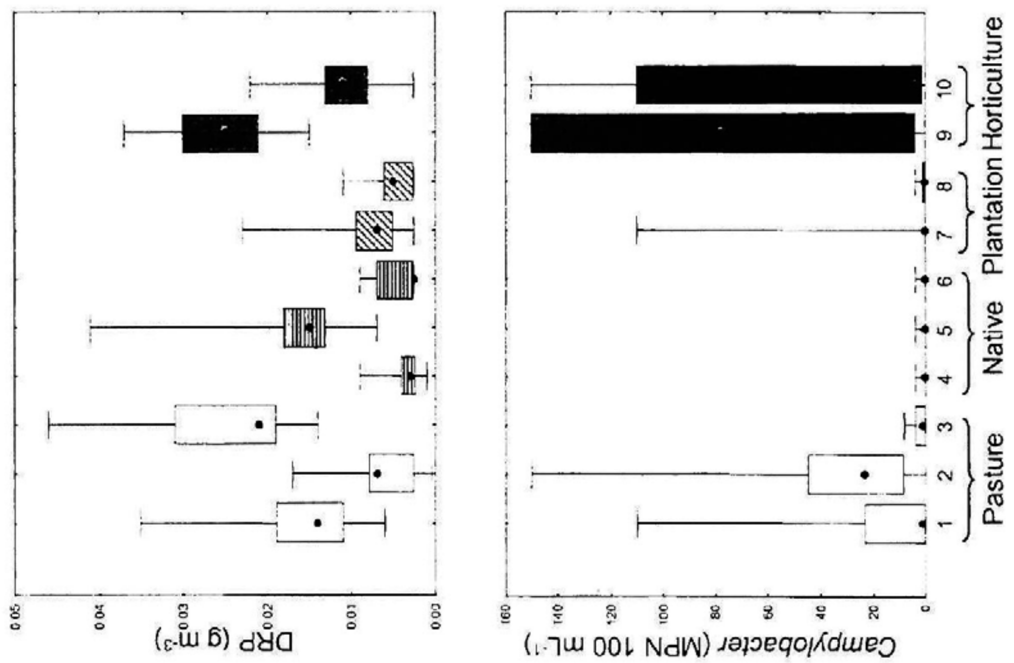

Figure 7: Comparative concentrations of *E. coli* and *Campylobacter* under different land uses in sub-catchments of the Motueka River (Young et al. 2005, p812)

Donnison et al. (2004) studied catchments at Whatawhata under different land uses. They found that unfenced pastoral streams consistently failed to meet stock drinking water guidelines, and also that indigenous and pine forest streams failed to do so in summer. Of their fortnightly samples over two years, contact recreation standards were not met in 28% of pastoral samples, 25% of indigenous forest samples, 14% of 7-year pine samples and 5% of new pine samples. This indicates that wild animal sources can cause breaches of standards in small catchments even without the presence of stock. They found no significant difference between *E. coli* concentrations in the 7-year pine and the indigenous forest, suggesting that feral animal populations might have been similar across these two forest types.

Source tracking of *Campylobacter* at Toenepi (French et al. 2010) suggested that the relative frequency of ruminant versus wildlife strains varied according to flow. At base flow the types were predominantly of wildlife origin, whereas during high flows there was more evidence of ruminant types. The authors were testing the hypothesis that ruminant strains dominate stream export only during freshes and floods, and hence the public health threat could be lower than previously thought, given that ruminant-associated types are recognised as important human pathogens in New Zealand, and wild bird types less so. Their analysis showed that while the majority of *C. jejuni* isolates recovered from the Toenepi Stream were not of cattle origin, a significant proportion of them were, indicating a health risk was present. The most prevalent type in the water has only been isolated from pukeko and is not recognised as a human pathogen. The avian contribution to campylobacteriosis in New Zealand is not yet fully understood, but some types associated with wild birds are also associated with human illness, so wild sources cannot be considered completely harmless (French et al. 2009 and see Figure 1 above).

Moriarty et al. (in prep (c)) concluded that the relative contributions of waterfowl to the microbial pollution of a water body will depend on the sizes and species of bird populations in a particular region, and their proximity to the waterway. Monaghan et al.

(2010) consider that water fowl can be an important source of faecal bacteria in rural streams when good practice, such as improved effluent management, stock exclusion and stock crossings are in place, and when large populations of water fowl inhabit stream reaches.

For *E. coli*, Wilcock (2006) calculated a possible total annual load from a swan, present over 30 days of the year, of $3 \times 10^9 – 3 \times 10^{10}$ *E. coli*/bird/year. This is in a similar order as a total *daily* load from a dairy cow of 1.2×10^9 *E. coli*. However, a large proportion of the wild fowl faeces could be deposited directly to water.

A number of studies have found a strong correlation between days since grazing and microbial contamination levels during run-off events (Collins et al. 2003; Connolly et al. 2004; NIWA 2006a; Collins 2002b; Monaghan et al. 2010). This again indicates the influence of livestock sources.

In summary, there is a general correlation between livestock density and *E. coli* concentrations in water, and livestock are likely to be the dominant contamination source in waterways draining farmed catchments, particularly after rainfall. However, wild animals can cause breaches of water quality standards in non-pastoral headwater catchments. Waterfowl tend to deposit a large proportion of their faecal material in or near the water, and in certain locations birds have been found to be the principal source of contamination. There is evidence from a Waikato dairying catchment that wild bird sources are particularly important at base flows. These may be less of a concern for human health than ruminant types; however some avian types of *Campylobacter* are associated with human disease. Overall, there is insufficient evidence to definitively quantify wild animal, including bird inputs across waterways or to be conclusive as to the level of human health risk from these sources. This is likely to be the subject of ongoing research and debate.

5.3 Direct deposition vs run-off

Direct deposition from livestock to streams accounts for only a small percentage of total annual catchment faecal deposits and catchment loads to streams. For example, using data for the Toenepi, Davies-Colley et al. (2008) estimated that direct deposition accounted for only about 0.23% of the total annual *E. coli* ‘production’ from the catchment streams (assuming 46% of stream length was fenced and without accounting for die-off). Similarly, calculations using data reported for the Waikato region by Moriarty et al. (2008) indicate that direct deposition into a typical stream would not produce a measurable change in the concentration of *Campylobacter* when considered on an annual contribution basis (R. Muirhead pers. comm. Sept 2010).

However, while direct deposition is only a small proportion of total *annual* yield, under base-flow conditions when there are few other inputs, direct deposition was still considered by Wilcock (2006) to be the most important source of faecal contamination from livestock. This potential is further illustrated through calculations for *Campylobacter* deposition (R. Muirhead pers. comm. Sept 2010), indicating that a total load of 1.7×10^7 *Campylobacter*/ha/day could be shed on an 80 ha Waikato farm with 230 cows. For a stream with unimpeded cattle access that received 1% of the daily faecal matter shed (Wilcock 2006), 1.7×10^5 *Campylobacter* per ha/day could be deposited directly into the water. For *E. coli*, McKergow and Hudson (2007) considered that deposition and complete mixing of a single cowpat into a stream flowing at 100 l/sec could give an immediate increase in local concentration of 4500 *E. coli*/100 ml. Therefore, the short-term and immediate effects of direct deposition cannot be discounted, even though on an annual basis these effects are overshadowed by the larger quantities of faecal contamination carried in storm events.

The reason that direct deposition can dominate base flows but still be only a minor proportion of total annual export is that annual yields are strongly weighted towards flood events. For example, studies at Toenepi indicated that 95% of the annual yield was exported during the thirty storm flood events that occurred over a twelve-month period (Davies-Colley et al. 2008). These bacteria were derived from re-suspension of sediments as well as catchment inputs. The total exported yield represented around 6% of the expected loading from livestock in this catchment (with no allowance made for wild sources making up part of the exported yield). This suggests that the majority of the export occurs through overland and sub-surface flows (rather than direct deposition), but that much of the faecal material deposited by livestock does not reach the water by any of these pathways, being held in the soil or vegetation, or succumbing to die-off.

Modelling at Toenepi (Muirhead et al. 2008) estimated that for a "model" farm, 80% of the faecal contamination occurred through overland flow. This modelling assumed 90% of the paddock streams were fully fenced, and effluent was treated and discharged from a two-pond system. Under these assumptions, direct deposition accounted for 1% of total annual losses to the stream, and pond discharges for 9% (see Figure 8).

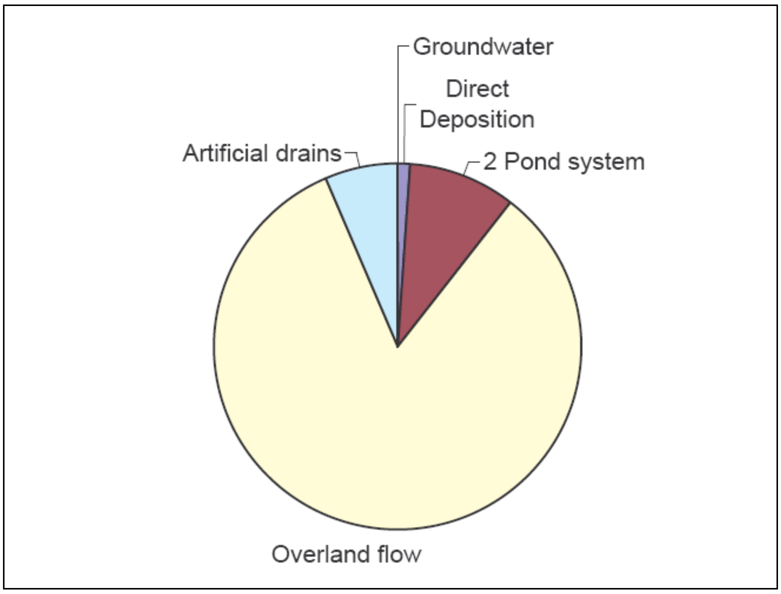

Figure 8: Estimated proportion of annual *E. coli* losses from a modelled Toenepi farm (Muirhead et al. 2008)

It is, however, important to note that although the highest faecal bacteria counts (particularly *E. coli*) occur after rainfall, these bacteria are derived from washout of faecal deposits or soil. In contrast, faecal bacteria deposited directly into waterways have not yet been subjected to environmental stresses, and at least at the time of deposition, any pathogens present would be in their most viable and infectious state.

Stock access can also serve to re-charge bed sediment stores of microbes, thereby increasing peak concentrations during rainfall events. As noted earlier, Nagels et al. (2002) found that an artificial flood event generated by tripping a dam in the Topehaehae Stream (with no run-off) had similar faecal yields to a natural event (with run-off). This finding supports the argument that microbes stored in bed sediments (including through direct deposition) will be mobilised in flood events. The relative contribution of stock access to restocking of bed sediment reservoirs is not fully established, as catchment run-off also results in restocking during smaller rainfall events and in the falling limb of flood peaks (McKergow and Davies-Colley 2010; Davies-Colley et al. 2008). Collins (2002b) has argued that another mechanism by which bed reservoirs are restocked between flood events is through sub-surface flows from wetland areas carrying microbes to streams, suggesting that stock exclusion from these wetlands is also important.

Localised impacts of stock access can be significant. The effects of run-off from grazed cropland were found to be small in comparison to stock accessing the channel upstream in an Otago study by McDowell (2006). The farmer had introduced ten calves to clean up the stream banks, and this was enough to produce a peak of *E. coli* that was higher than any effect seen from cropland run-off.

The impact of direct deposition differs for different stock types. Sheep behaviour has not been reported, but anecdotally it is known that sheep prefer not to enter waterways. However sheep deposit faeces in riparian areas, and do camp in these areas (Evans 1998). Direct deposition may be particularly important for deer. Deer tend to wallow in wet areas, and if these are connected to waterways they can be significant sources of faecal contamination. McDowell's (2009) study of an Otago deer farm found that where deer wallows were connected to streams, *E. coli* measurements were at the higher end of the range of loads that has been reported thus far for pastoral catchments in New Zealand. In catchments where deer wallows were not connected to streams, *E. coli* levels were similar to other dry stock pastoral systems. Where the wallowing sites were connected to a stream, about two-thirds of the overall *E. coli* load to the stream was contributed when deer had access to wallowing sites. *E. coli* measurements were not related to flow rates, indicating that there was considerable contribution from the wallows at base flows.

While direct deposition represents only a small proportion of the total annual export of faecal microbes, it is particularly important in summer and at base flows, when dilution potential is lower, and other sources and pathways (such as overland flow and sub-surface drainage) are less active (Monaghan et al. 2010). In a study of a King Country farm, Ross et al. (2010) found no relationship between rainfall and the concentration of faecal bacteria, with high counts in summer in the absence of rainfall. They attributed this to stock accessing the streams and wetland areas. Muirhead et al. (2008) noted that direct deposition and effluent are the dominant inputs to streams 90% of the time, when overland flow and artificial drainage are not generated. It is also in these conditions (summer/base flow) that lowland waterways are put to greatest recreational use. In addition, downstream aquaculture harvest occurs at base flows, but is suspended by commercial operators during significant rainfall events.

Therefore, while stock exclusion may not significantly influence *annual* catchment exports of faecal bacteria, arguably it can make a difference during the *most critical times* for public exposure to health risk (i.e. at base flows). However, although stock exclusion could result in better water quality at base flows, there are some risks to water users which would not be addressed. Firstly, faecal microbes stored in bed sediments may be re-suspended by recreational users themselves. Secondly, when water is sourced but not properly treated for domestic human use, its quality may be adversely affected both at base flows and during rainfall events. Thirdly, members of the public may not be aware that they should not harvest shellfish after rainfall. These users will still be exposed to risk from run-off sources. The role of water for stock drinking and the cycling of pathogens between livestock also need to be considered.

The information in this section suggests that direct deposition is particularly important at base flow, which is also the time of higher recreational and commercial aquaculture use. Direct deposition is less significant in terms of total annual loads and storm contamination peaks, although it contributes to the latter by restocking bed sediments with faecal microbes between rainfall events. Stock exclusion would help to protect water quality at times of high use by preventing direct deposition and by avoiding the stirring up of sediment by stock. Further evidence of the effectiveness of stock exclusion is presented in the section on Mitigation, below.

5.4 Comparative loadings

Wilcock (2006) compared major sources of faecal matter in the Waikato region for land and water (see Figures 9 and 10).

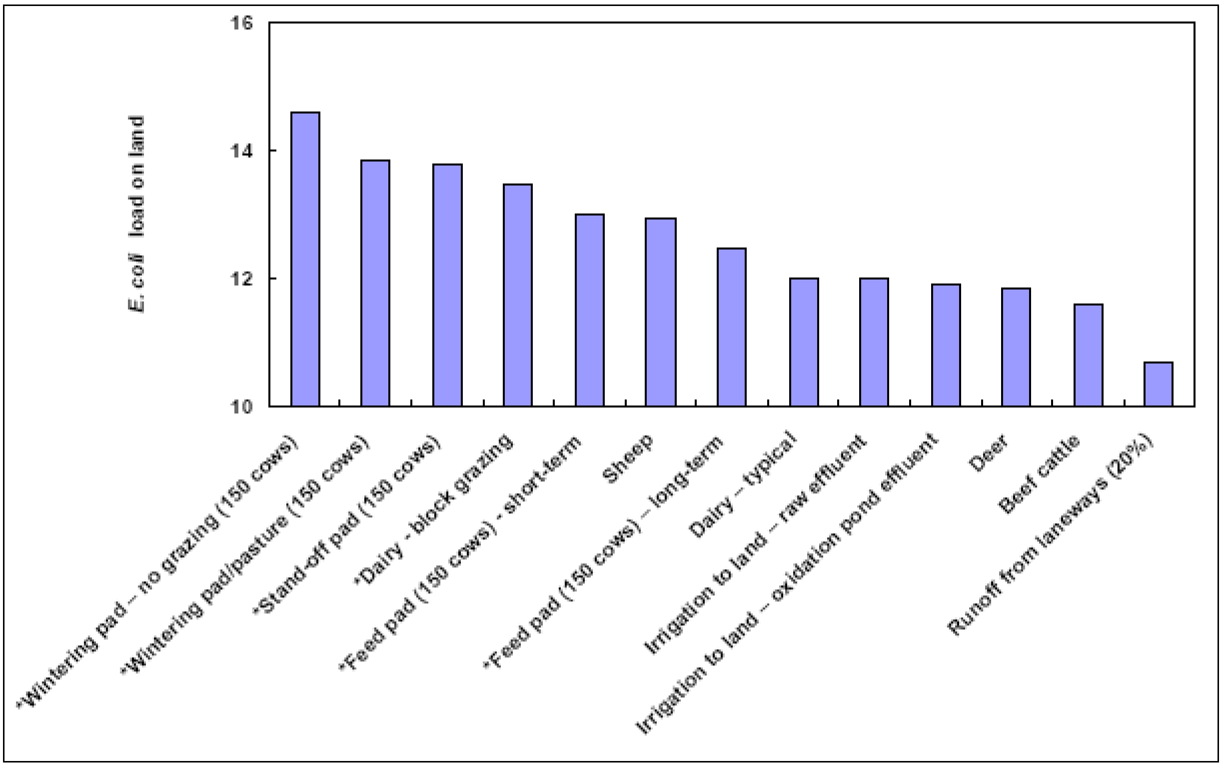

Figure 9: **Land loadings, log10 (*E. coli*/ha/yr), for major sources of faecal matter in the Waikato region. Note that some areas (e.g., feed pads, block grazing) are small compared to whole farm grazing. These smaller loading areas are marked (*). (Wilcock 2006, p22)**

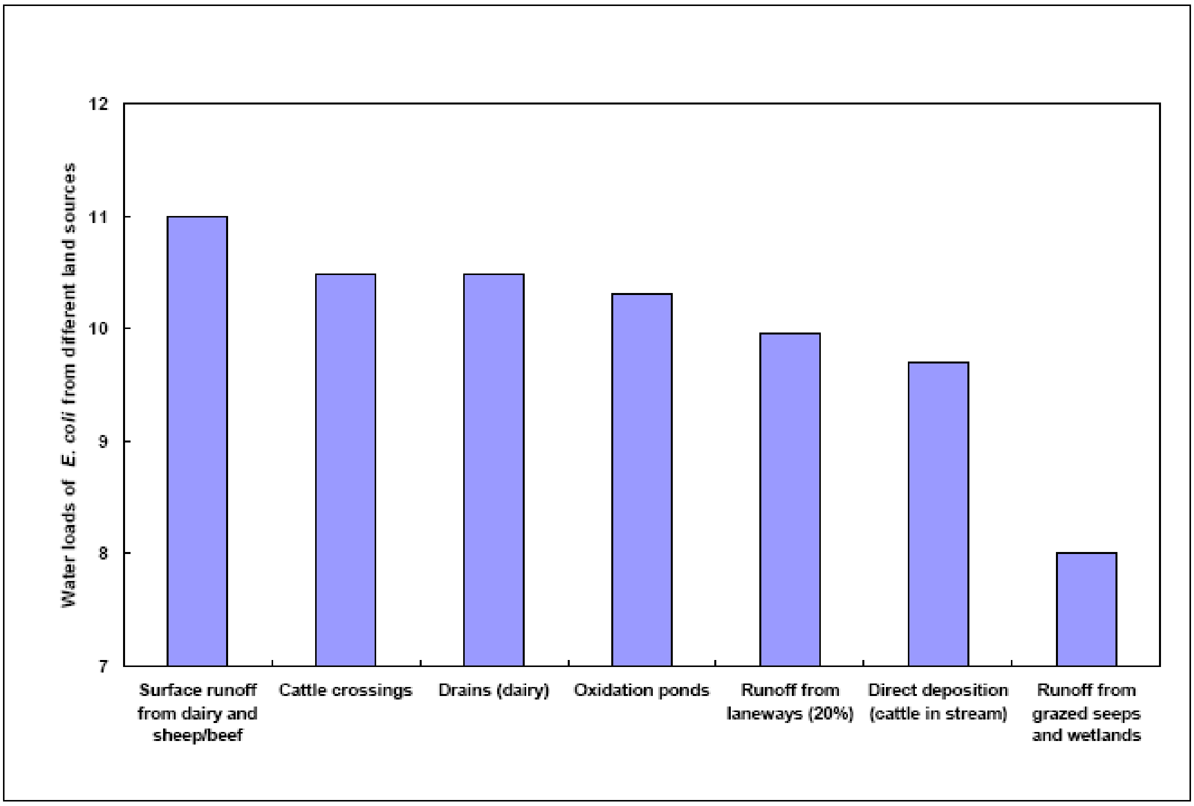

Figure 10: **Waterway loadings, log10 (*E. coli*/ha pasture/yr), for major sources of faecal matter in the Waikato region. (Wilcock 2006, p23).**

In Wilcock's analysis, land loadings from sheep were higher than those from dairy, deer or beef cattle. In terms of contamination reaching waterways, surface run-off sources were judged to be the highest source, but stock access, and particularly crossings also contributed to the overall load reaching the water (see Direct deposition vs run-off, above). Figure 9 also shows that there are 'hotspots' of concentrated land loading, such as pads, stand-off areas and intensive block-grazing. Although these sites typically only occupy a small area, and hence do not make a large contribution to waterway loadings overall (Figure 10), if they have high connectivity to waterways they can cause localised contamination, as is the case also for laneways.

Drains and effluent ponds can exhibit high concentrations of microbes, although it has been reported that many ponds cease to discharge during summer periods of low rainfall (Wilcock et al. 1999), and increasingly more farms irrigate pond effluent to land. As noted previously, modelling estimates for Toenepi of the contribution of effluent ponds to the total faecal bacteria export from farms using these ponds range from 9% (Muirhead et al. 2008) to 50% (Moynihan et al. 2008). However, catchment-scale studies have not found that ponds significantly influence microbial water quality (Collins 2002a; Wilcock et al. 2006).

Effluent irrigated to land may also make a significant contribution to faecal loads reaching waterways in some circumstances. Monaghan and Smith (2004) recorded irrigated effluent contributing 86% of the total faecal load draining from a dairy pasture plot underlain by sub-surface drainage in Otago. However, they cautioned that these results were likely to be particular to those soil and climate conditions. Modelling for the Bog Burn catchment (Monaghan et al. 2007) suggested that mole-pipe drainage under effluent irrigation might account for 78% of the total faecal load in the stream, compared with 16% from surface run-off and 0.1% from direct deposition. Notably, this catchment is relatively flat, with extensive artificial drains and stock already excluded from 84% of the stream length. In a non-irrigated site, it was estimated that 66% of the yield was derived from surface run-off with the remainder coming via mole-pipe drainage (Monaghan et al. 2010). It can be concluded that where effluent is irrigated onto soils with mole-pipe drainage or very coarse structure, by-pass flow of effluent may dominate the faecal export to streams. Wherever possible, irrigation onto these areas should be avoided, or only carried out at low rates and low volumes per pass.

Clearly, situational factors determine the relative importance of different faecal contributions to land and water loadings. It can be concluded that sheep loadings to land can exceed those from typical dairy stocking, but intensive stocking of dairy cows and hotspot areas have much higher loadings than typical stocking rates of either sheep or cattle. While intensive uses occur over smaller areas, if there is connectivity to waterways these sites are a potential source of contamination. Surface run-off is the dominant pathway for waterway loadings, but herd crossings and direct deposition also contribute directly, with no opportunity for attenuation. Dairy effluent is a source of contamination when discharged from ponds or to land with high risk of by-pass flow. While surface run-off dominates storm-flow loadings and overall annual exports, at base flows, dairy effluent and stock accessing waterways are important loading factors. These can be managed to reduce the risk of directly contaminating water, and also of restocking bed sediments with faecal bacteria reserves that can be mobilised in storm flows.

5.5 Assessing critical source areas in a landscape

Critical Source Areas (CSAs) are those places in a catchment where pollutants, including faecal microbes, are likely to be most susceptible to transport. A model developed by Collins and Rutherford (2004) created a delivery index for areas of a catchment, based on three components:

- Stream proximity
- Slope
- Flow accumulation (the force of run-off water when it reached that part of the catchment)

Srinivasan and McDowell (2007) compared a range of models for identifying CSAs. They found that the occurrence of porous soils in steep valleys with seeps and springs strongly affected run-off dynamics and the models did not give accurate findings in these situations. A catchment with these characteristics was found to quickly convert rainfall into run-off via seeps and springs, meaning traditional models to predict surface run-off underestimated the resultant volume of run-off. Where these characteristics were not present, models based on areas beside waterways being active CSAs during rainfall were reasonably accurate, as were models that looked at soil transmissivity and predicted saturation and infiltration-excess run-off.

Analysis of spatial and time-series data over 18 months (Srinivasan and McDowell 2009) indicated that during dry seasons (below-average rainfall periods), the majority of storm flow came from direct precipitation, wet areas adjacent to the stream and semi-pervious areas such as animal tracks. During wet periods (above-average rainfall), flow from these areas accounted for only 10–70% of total storm flows (i.e. run-off flows were contributed by larger parts of the catchment). Since most saturation-excess surface run-off during small storms occurred within a short distance either side of the stream channel, the authors concluded mitigation strategies could target compacted areas like gateways and water troughs as well as stock exclusion from near-stream areas.

Although the above studies were not specifically focused on faecal microbe transport, overland flow is a significant pathway by which faecal microbes reach waterways (e.g. Wilcock 2006). Therefore flow characteristics and proximity to waterways will have a considerable effect on faecal contamination. Management strategies for reducing contamination could therefore focus on siting effluent irrigation blocks away from identified critical flow areas, and avoiding the grazing of these areas when heavy rainfall is expected.

In addition to transport pathways, identifying "hotspot" source areas where effluent accumulates through stock presence is also an important aspect of assessing CSAs for effluent contamination. Collins (2004) found that small wetlands in hill country pasture could be a critical source of bacterial input to streams, given the preference of cattle for grazing these areas, the large fraction of flow from the catchment that goes through them and their proximity to streams. Other hotspots can include yards, lanes and races and stand-off or sacrifice areas (Wilcock 2006) as well as grazed fodder crops (McDowell et al. 2005).

6 Mitigation

Land use change away from livestock will reduce faecal loadings, but obviously has major economic implications. Within grazed catchments, mitigation measures for microbial contamination of waterways can address both direct and indirect pathways to effect improvements in water quality.

Possible mitigation measures for faecal contamination are described in detail in the following sections. These are:

For farm or catchment scales
- Land use change

For direct pathways in pastoral land
- Stock exclusion and installing stock crossing structures

For indirect pathways in pastoral land
- Riparian buffers and grass filter strips, including around wetlands
- Managing run-off from farm tracks, lanes and other 'hotspots' (sites of manure accumulation)
- Avoiding practices that are high-risk on some soils and locations
- Effluent management
- Drain management, settling ponds and constructed wetlands.

6.1 Land use change

Pastoral land typically generates higher levels of faecal contamination than forested land. When a catchment was retired from stock and planted to pines at Whatawhata, *E. coli* concentrations in the stream fell rapidly and remained at levels lower than those in streams exiting from either indigenous or 7-year pine forests (Donnison et al. 2004). After conversion to pines, the percentage of stream samples in these authors' "satisfactory" category (<200 *E. coli*/100 ml) increased from 30% to 89%. The higher concentrations from the older pine and indigenous forest compared to new pines was attributed to having more undergrowth and canopy shelter attractive to wildlife, and more favourable conditions for bacterial survival. This could be partly addressed by pest control in forest blocks, which could be expected to lower the *E. coli* exports from these areas. There remains some uncertainty around the human health risk from wildlife inputs, with indications that they are of lower risk than microbes associated with ruminants, but can still have an association with human disease (Mullner 2009; French et al. 2010; French and Marshall 2009) (see Environmental sources in Figure 1 above).

6.2 Stock exclusion and crossings

Davies-Colley et al. (2004) found that crossing cattle through a stream in the Tasman District increased the stream *E. coli* loading significantly, and extrapolated their findings to calculate that over the day, these cattle crossings were responsible for quadrupling the background concentration upstream of the crossing (where there were no other dairy farms). While a relatively small proportion of the total daily defecation of the herd was deposited into the stream during the four crossings each day (calculated by Wilcock (2006) as 3.6%), the fresh microbes would be at their most viable and infectious. Sampling after bridges had been installed showed that *E. coli* concentrations at base flow had halved at sampling point 3, below three former herd crossings which now have bridges (Figure 11). Although the graph shows that concentrations were lower at all sites in the post-bridge monitoring period, including those upstream of the crossings, the highest absolute reductions were downstream of the bridges (Figure 11, sampling points 3 and 4). In spite of the crossings being installed, water quality samples below the crossings failed to meet the guideline of 126 *E. coli*/100 ml. Further studies of this nature would be useful to confirm whether these findings are generally applicable.

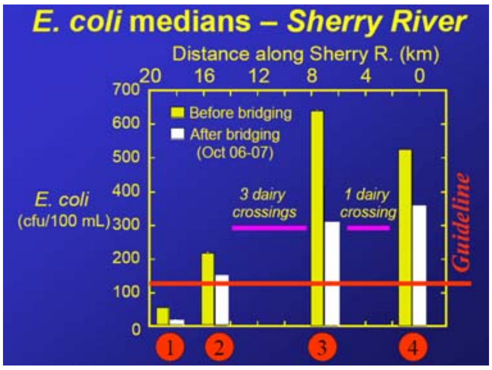

Figure 11: Reduction in *E. coli* in Sherry River after bridges were installed (Young et al. 2008)

There are few comparative studies relating in-situ riparian protection and microbial water quality in New Zealand. The complexity of microbial sources, survival and movement, and local effects at sampling points makes it difficult to discern the influence of riparian protection in paired field sites. Parkyn et al. (2003) compared areas with riparian buffers against paired reaches without riparian buffers but their findings were inconclusive - microbial water quality was better in some of the areas with buffers, but worse in others. McDowell et al. (2006) tested water quality in three streams on a Southland deer farm and reported that *E. coli* levels were lowest in a fenced stream and highest in an unfenced stream, while a partially fenced stream showed intermediate levels (exact values were not presented).

McDowell also conducted a ‘before and after’ study, where provision of wallowing areas for deer that were not connected to streams (‘safe wallows’) reduced loads of contaminants by up to 90% (McDowell 2009). In the Waikato region deer farming is a minor land use, so while deer exclusion would be expected to make localised gains it is unlikely to significantly change the overall water quality patterns.

There are few other New Zealand studies which clearly document the effects of stock exclusion. In Waiokura (Taranaki), in one of the “best-practice dairying catchments”, *E. coli* concentrations were reduced from 1500 to 900 *E. coli*/100 ml (or 40%) over six years from 2001-2007, with an increase in riparian protection from 40 to 52% of total stream length recorded between 2001 and 2004 (Wilcock et al. 2009). While the recorded change in riparian protection appears small in relation to this observed fall, there may have been further changes in unreported riparian practices over the whole period. Some pond discharges were also removed during this time.

In a stream in Vermont where a 340 m length of riparian exclusion fencing was installed, Line (2003) found a reduction of 65.9% in faecal coliforms and 57.0% in enterococci. Meals (2001) reports that bacteria counts declined by 29% - 38% after fencing 726 m of stream in one Vermont catchment (being 49% of the stream length). Similar water quality changes were observed in a neighbouring catchment, but the impacts of farm expansion reversed those improvements. While these changes were significant, Meals and Braun (2005) reported that in that area stock exclusion and riparian buffers alone were unlikely to achieve sufficient reduction to result in the relevant local water quality guidelines being met. Although there has been no comparable study done in New Zealand, the Vermont studies were done outdoors under pastoral grazing conditions, which are the conditions that apply in New Zealand.

Some New Zealand authors have modelled the effects of stock exclusion. McKergow et al. (2007) estimated a 20-35% reduction in *E. coli* through introducing a 2 m fenced margin alongside a typical stream. Collins and Rutherford (2004) estimated a 15% drop in *E. coli* levels if stock were excluded from streams and seepage (wetland) areas. Their assumptions (based on earlier observations at Whatawhata by Collins (2004)), estimated that 8% of cattle excretion was direct to a stream and that 20-40% of cow pats were deposited on or adjacent to seepage zones.

Where stock exclusion is impractical, the suggestion has been made that alternative water or shade might encourage animals away from waterways. However, Bagshaw (2002) studied the effect of an alternative water source, shade, season, field size and pasture availability, and found these had no impact on the number of defecations in the riparian zone. In Vermont, Line (2003) found that installing trough water did not significantly affect bacteria levels. An American review (Agouridis et al. 2005) also found little evidence that off-stream water, shade or supplementary feed improved stream water quality.

McKergow and Hudson (2007) were asked by Environment Canterbury to determine stocking rate thresholds at which stock exclusion was justified due to a ‘significant effect’. They found little published literature documenting effects of stock access on water quality. However, they made calculations relating numbers of faecal microbes in cowpats to stream flow and concluded that in small streams, even a single cow could cause water quality standards to be breached.

The Clean Streams and Dairying Accord calls for stock exclusion from significant wetlands and from streams that are wider than a stride, deeper than a gumboot and flowing all of the time. However, smaller tributary streams are a high proportion of total stream length, so where these are unfenced their contribution to the total water body can be high (Ross et al. 2010). One study in Western Australia showed that 80% of total stream length was low order streams (1 or 2) and these had the worst riparian condition scores (Weaver et al. 2001).

There is evidence that ephemeral waterways, seepage areas and small wetlands benefit from stock exclusion (Collins 2002b; Collins 2004; Parkyn 2004), and extending fencing upslope along flow paths to exclude stock from wetland seepage areas is recommended (NIWA 2006a). Collins (2002b) observed that cattle at Whatawhata were attracted to wetlands for summer grazing, and estimated that 75% of the observed faecal material in a paddock was deposited around a wetland. He sampled sub-surface water in wetland areas and in drier riparian areas at Whatawhata. He found that the sub-surface water was not affected by grazing in the drier riparian soil, but that there were significant peaks in *E. coli* in sub-surface soil flows in wetlands following grazing. Exclusion from small wetlands was considered a more significant mitigation opportunity than wider riparian buffers near streams, since high flow rates in hill country catchments tended to by-pass riparian vegetation. In a later study, Collins (2004) found that a larger wetland was not similarly attractive to cattle and exclusion from this deeper wetland would not have the same mitigation effect. Overall these findings suggest that for management of faecal contamination of streams, fencing of deeper wetlands is of lower priority than fencing shallow or ephemeral wetlands, to which stock are particularly attracted. The benefit would be determined by the amount of time stock spent in or near the wetland. However, there may be other reasons for stock exclusion from the margins of large wetlands, such as enhancing habitat.

6.3 Riparian buffers and grass filter strips

Riparian buffers provide the following mitigation effects (Parkyn 2004; Collins et al. 2007):

- preventing deposition on the banks as well as the beds of waterways
- physical filtering of run-off by rank vegetation
- slowing of run-off due to roughness of vegetation, promoting entrapment and settling, infiltration and filtering effects
- reduced compaction of soil as stock are excluded from these zones, further enhancing the potential for infiltration through the soil, which will provide a mechanism for removal by entrapment and die-off.

Research reported in this section indicates that riparian buffer effectiveness is strongly dependent on flow characteristics. Relatively speaking, at slower run-off flows, and where sheet flow rather than channelised flow is generated, riparian buffer zones are at their most effective in promoting infiltration, settling and entrapment (see Flow affecting entrapment, above).

New Zealand trials with 5 m grass filter strips have reported rates of 27% to 95% entrapment (Collins et al. 2004; Muirhead et al. 2006a). Overseas, a 64-87% removal rate of faecal coliforms was measured by Fajardo et al. (2001) by a 30 m buffer strip at high rainfall on a 4° slope. Even a 3 m grass strip on a 9° slope achieved a 43-74% removal rate in a study by Coyne et al. (1998) during a simulated 1-in-10 year rainfall.

Little is known of the ultimate fate of microbes trapped in strips and whether they wash out in subsequent events. One trial at Ruakura (Collins et al. 2004) showed outflow concentrations of *E. coli* and *Campylobacter* were 2-3 orders of magnitude lower in subsequent events generated 5 days after the initial run-off event. However they noted that this occurred after a high flow initial event, which may have left relatively few microbes available for remobilisation. Also, the original source was liquid effluent, and not clumped material, which might have shown greater retention and later mobilisation. Meals and Braun (2005) suggest that filter strips can become reservoirs for sediment-bound faecal organisms which can be persistent, and Muirhead et al. (2006a) consider that buffer strips can be both a source and a sink for faecal microbes. This is similar to observations for drain and stream sediments (see Streambed sediments, above).

Factors which influence riparian buffer strip effectiveness are described below and include: flow characteristics (a combination of topography, soil type and rainfall), buffer width and vegetation.

6.3.1 Flow characteristics

Collins et al. (2005) suggested that a vegetated buffer strip would have had minimal impact on faecal bacteria in overland flow in their simulated high (35 mm/h) rainfall event at Whatawhata. This event was equivalent to an 8-yr return storm event and generated peak flows between 1.75 and 6.5 l/sec. This was almost 3 times higher flow than the top flow trialled at Ruakura (Collins et al. 2004), which showed low attenuation of microbes by a 5 m strip at 13 l/min (2.2 l/sec). They concluded the hill country flow would be too fast for bacteria to settle out in large storm events, but that riparian buffers could be effective in lower magnitude rain events. In the intermediate flow trials at Ruakura (10 l/min or 0.17 l/sec), entrapment rates ranged between 38 and 84%, while at the lowest flow trialled (4 l/min or 0.07 l/sec) entrapment rates reached 95% (Collins et al. 2004). This was one eighth of the flow trialled at Whatawhata, where minimal entrapment resulted. Muirhead et al. (2006a) found no attenuation over 5 m at flows of 6 l/min (0.1 l/sec), but when this flow was reduced by two thirds (to 2 l/min) they achieved removal of 27% of the *E. coli* in the overland flow. This result, on a Pallic soil with impeded drainage, was a lower rate of entrapment than the 95% achieved by Collins et al. (2004) at double the flow (4 l/min), on a clay loam at Ruakura, reflecting different behaviours of soil types.

Where soils are highly porous (e.g. pumice or light ash) and minimal surface run-off is generated, filter strips are unlikely to be an important mitigation tool (McKergow et al. 2008).

Collins (2002b) found no effect of an 8 m riparian buffer on the level of contamination of sub-surface seepage water in areas characterised by soil saturation at Whatawhata. This was attributed to shallow channels in the riparian strip allowing run-off to bypass riparian vegetation.

Parkyn (2004) suggested that where slopes induce channelised flow, riparian buffers would need to extend along these flow channels, upstream of the main stream channel. McKergow et al. (2008) have experimented with the use of grass buffer strips away from stream channels, located on contours at mid-slope locations. These are designed to trap pollutants closer to source, before run-off becomes channelised. Their initial trials in Rotorua lakes catchments did not show a significant reduction in *E. coli* through the use of the strips, possibly because of the long duration of the rainfall events exceeding the capacity of the strips to trap or retain microbes. Further study of this mitigation mechanism in the Waikato region is required to assess its usefulness. However, initial evidence suggests that grass filters can be effective in smaller rainfall events in areas where sheet run-off is generated.

Researchers at NIWA (2006b) related run-off characteristics to topography and soil type, and predicted the consequent efficiency of riparian buffer strips (see Table 8).

Table 8: Estimated optimal width and efficiency for riparian buffer strips with respect to faecal bacteria. Buffer width is given as a percentage of hill slope length. Buffer efficiency is expressed as a percentage reduction, and represents a 'best-case' estimate of average efficiency (NIWA 2006b, p8).

Slope	Soil Drainage Rate	Bacterial Attachment	Buffer Width	Buffer Efficiency	Notes
Flat to Undulating	Low <1-4 mm/h	High: ≈ 90%	1	95	On well-drained soils, RBS may not be warranted since vertical movement of water and microbes dominates. In addition, such land often has artificial subsurface drains. High intensity rainfall, however, can generate significant surface runoff on poor or imperfectly drained soil, even with artificial drainage.
		Medium: ≈ 70%	5	90	
		Low: ≈ 40%	9	80	
	Medium 5-64 mm/h	High	1	95	
		Medium	2	90	
		Low	4	80	
	High 65->250 mm/h	High	1	95	
		Medium	1	95	
		Low	3	85	
Rolling to Moderately Steep	Low	High	2	90	Generally, these are the most appropriate slope angles for RBS since sufficient surface runoff is generated, and as spatially diffuse sheet flow rather than concentrated in rivulets or channels.
		Medium	7	70	
		Low	15	50	
	Medium	High	1	95	
		Medium	4	80	
		Low	11	55	
	High	High	1	95	
		Medium	2	85	
		Low	4	60	
Moderately Steep to Very Steep	Low	High	5	45	RBS efficiency can be limited by topographical convergence of surface runoff, causing channelised flow. Buffers may need to extend some distance upslope, following flow pathways. Exclusion of stock from critical source areas (e.g., wetlands, flow pathways) is an important mitigation measure.
		Medium	15	30	
		Low	30	20	
	Medium	High	3	60	
		Medium	7	50	
		Low	13	35	
	High	High	3	75	
		Medium	4	70	
		Low	11	50	

6.3.2 Buffer zone width

International work has found a linear decrease in coliform concentration in outflows with increasing length of buffer strips (0-25 m) (Young et al. 1980). This is consistent with a series of studies on sediment comparing multiple buffer widths in the same location (cited in NIWA 2006a). These showed that sediment entrapment increased from 53% to 98% with increasing buffer width from 4.6 m to 27 m.

Experiments at Ruakura (Collins et al. 2002) showed that the length of a grass buffer strip affected outflow rate and the timing of *E. coli* and *Campylobacter* peaks. The rate of water outflow was faster for 1 m than 5 m plots and concentration peaks of *E. coli* and *Campylobacter* occurred earlier on the 1 m plots. As well, the *Campylobacter* concentration was higher in the outflow from the 1 m plots than the 5 m plot, in spite of similar numbers of these bacteria being applied to 1 and 5 m plots. This difference was not observed for *E. coli* concentration, but the authors noted that due to a high background of *E. coli* in the soil it was difficult to accurately account for the fate of effluent *E. coli*.

In contrast to overseas studies finding a linear relationship between distance and microbe removal, Muirhead et al. (2006a) found a logarithmic relationship, with over half of the total removal over 5 m occurring in the first 1 m. This contrast could be due to a difference in how many bacteria were attached to larger particles. Muirhead et al. surmised that any bacteria attached to sediment would settle out quickly, with the remaining bacteria transported as small particles, and having little interaction with the soil matrix under saturation-excess conditions. Gharabaghi et al. (2002) found that the first 5 m of filter strip was critical for sediment removal as most of the larger size particles settled out as flow slowed down. However, studies have found that as few as 8% of microbes travel attached (Muirhead et al. 2005). Gharabaghi et al. (2002) said that such unattached particles may only be removed if infiltration is achieved, requiring longer (>10 m) buffers.

In summary, there is a lack of data with which to make definitive recommendations on buffer zone width for microbe trapping. Generally, buffer widths will need to widen as the slope length, angle and clay content of the adjacent land increase and as soil drainage decreases (Parkyn 2004).

6.3.3 Vegetation

Little work has been done specifically on grass length and microbe entrapment. One study at Ruakura (Collins et al. 2002) found no difference in entrapment efficiency between 30 cm grass length and 7-10 cm grass length in 1 m and 5 m strips. However, both grass plots had a dense vegetation mat on the soil surface, which could have promoted entrapment. Also, the rainfall rate simulated in this trial represented a ‘worst case scenario’ and lower flows were not tested; nor was there any comparison with a bare ground or hard-grazed scenario.

Muirhead et al. (2006a) assessed the attenuation of *E. coli* by grass and cultivated 5 m strips and found the removal was significantly greater in the cultivated strips (41% removal) than the grass strips (27% removal). This was attributed to a greater infiltration rate in the cultivated plots (due to the tillage) which promoted a greater volume of flow to pass through the soil matrix, providing the opportunity for filtration and adsorption of microbes. However, the use of cultivated strips in riparian areas could have other water quality effects, such as increasing sediment transport, so it is unlikely to be a useful strategy in practice.

One overseas study looked at microbial attenuation under tree cover in riparian areas (Entry et al. 2000). This study compared strips that were 20 m grass + 10 m forest vs 10 m grass + 20 m forest. The strips did not behave differently in terms of total faecal coliform entrapment, suggesting that at least with these long riparian strips, vegetation type was not critical.

The practice of lightly grazing riparian areas to manage vegetation is likely to have a detrimental effect, as faecal material will be deposited in the stream and the riparian area during grazing (Bagshaw 2002; McDowell 2006). Parkyn (2004) notes that in some situations it may be possible to mow riparian vegetation for hay.

6.4 Management of lanes, tracks and other ‘hotspots’

Minor earthworks to create regular cut-offs on tracks and lanes can prevent water building up and forming preferential flow channels as it runs down the track. Using this technique, faecal material flowing on tracks can be redirected (via cut-offs) to areas of rough grass or wetlands before it reaches waterways. This is especially important on approaches to crossings where downhill tracks can direct flow into waterways.

Attention to run-off from tracks and lanes, especially near crossings, has been highlighted as a practice to improve water quality in a Taranaki dairying catchment (Betteridge et al. 2005). These authors point out that with 107 culvert and bridge crossings in just one catchment, there is high potential for contamination. They recommend that where farm tracks cross the stream, well-formed cut-offs should be diverted to a simple detention storage dam (2-3 m^3) close to the stream, to allow sediments to settle before the run-off flows to the stream. Sediment removal is required to maintain effectiveness of these traps. Control of run-off from feed pads, silage pits and cowsheds located near streams was also recommended. However, no quantitative studies of the effect of these practices on microbial contamination are available for New Zealand so while the benefits seem logical, they cannot yet be quantified (see Laneways, tracks and yards, above).

6.5 On-off grazing of crops

Winter-grazed fodder crops can be a concentrated source of run-off. McDowell et al. (2005) found that unrestricted grazing increased *E. coli* in overland flow relative to ungrazed treatments on a South Otago Pallic soil. Average concentrations were significantly higher in run-off from unrestricted grazing treatments (in the order of 4 x 10^3/100 ml) compared to restricted grazing treatments (in the order of 2 x 10^1/100 ml). This study found that concentrations in run-off from restricted (3 hour) grazing areas were not significantly different to the ungrazed treatment. This suggests that restricted grazing can be an effective mitigation strategy compared to unrestricted foraging of winter crops, especially on soils prone to compaction and run-off. Winter-grazed crops are less common in Waikato than in southern dairying regions but if used, then restricted grazing would be recommended on wet soils.

6.6 Avoiding high-risk practices on some soils and locations

High-risk activities include the spreading of effluent (except with low application systems), intensive block-grazing, unrestricted grazing of forage crops, and the use of sacrifice or stand-off areas.

There are two key risk factors that could be considered in selecting locations for high-risk activities: proximity to waterways and soil drainage properties (related to surface run-off or sub-surface drainage risk).

When catchments are compared across the region, there is a strong correlation between poorly drained soils and microbial contamination (see Soil types, above). This is because these soils promote run-off rather than infiltration of rainfall. Heavy (clay) soils are also at risk because of large cracks and pores which promote by-pass flow, rather than filtering through the soil matrix. By-pass flow also often occurs in artificially drained soils. Therefore, ideally, irrigation of effluent should be avoided on very heavy soils or artificially drained soils, or if irrigation is the preferred option, then only systems capable of applying very low volumes and rates should be used (see Effluent management, below). Donnison and Ross (2009) recommended that effluent irrigation should be avoided on the gley Topehaehae soil in Toenepi catchment, which is found adjacent to streams, and that grazing should also be avoided, if possible, when either repeated showers or heavy rain is expected. Other strategies identified to reduce faecal contamination were fencing drains and seeps and removing stock to stand-off areas in winter.

6.7 Effluent management

Switching from traditional effluent treatment ponds to land irrigation or advanced pond systems can reduce the faecal load entering waterways. Modelling in Southland by Monaghan and Houlbrooke (2005) estimated that adding a K-line sprinkler system to a 2-pond system and irrigating to land instead could reduce farm *E. coli* losses to water by 12%. On artificially-drained land, K-line sprinklers could cut *E. coli* losses by 52% compared to travelling irrigators.

The impact of best management practices will vary depending on the risk of land application on each farm based on soil and landscape features (Houlbrooke and Monaghan 2010). As outlined above, risk factors include soils with a high degree of preferential flow due to artificial drainage or coarse structure, and soils with low infiltration or on sloping country where run-off and ponding can occur (Monaghan and Houlbrooke 2005). In these conditions, effluent may be a significant local source of contamination. Conversely, on free draining, porous soils with matrix flow characteristics, effluent applied to land is less likely to run off and the soil can be an effective filter of microbes.

The most appropriate mitigation in risky circumstances is to apply effluent at lower rates, and only when soil moisture conditions are favourable for application to land (deferred irrigation). Houlbrooke et al. (2004) established that when dairy effluent was applied to a mole-drained Manawatu soil near field capacity, approximately 40% of the applied effluent left the soil profile as mole and pipe drainage and 30% as surface runoff. Under deferred irrigation, only 1.1% of the applied effluent was lost. Deferred irrigation is reliant on having sufficient effluent storage capacity and being able to assess soil moisture conditions.

In the Houlbrooke research, effluent lost under deferred irrigation was attributed to uneven spray patterns of travelling irrigators and high instantaneous application rates. In Otago, Monaghan and Smith (2004) also observed a pronounced non-uniform pattern of effluent application beneath a travelling irrigator being run at typical application rates. Areas to the outside of the irrigator run effectively received double the average application depth, resulting in effluent loss to the mole drains below on occasions after recent rainfall. Although the volumes of effluent transported by the sub-surface drainage system were relatively small, the concentrations were high and loads of *E. coli* bacteria in the resulting drainage were a large proportion (81%) of the total annual yield from all sources from that plot. They recommended deferred irrigation, and speeding up the irrigator. Technology that can deliver more even spray patterns and lower rates and depths of effluent (such as sprinkler technology) can minimise these issues.

The choice of location for irrigation can also help to avoid run-off or sub-surface transport of effluent. Setbacks from waterways and drains, and avoiding areas with artificial sub-surface drainage are appropriate measures.

Soil and hydrological analysis can help to identify the best areas for effluent application, for example using electromagnetic conductivity assessment to identify sub-surface soil conditions (Eastwood et al. 2002).

6.8 Drain management, settling ponds and constructed wetlands

Wilcock (2006) suggested that stock exclusion from drains could be a significant mitigation measure, but gave no quantitative estimates for effectiveness. Managing open drains to retain vegetation may assist in entrapment and in slowing water flow so that microbes settle out (Nguyen et al. 2002). However, microbes may be subject to remobilisation in high flows, so the overall effectiveness of vegetated drains is unclear.

Constructed wetlands at drain outlets have been trialled with some testing of effectiveness for removing faecal organisms (Sukias et al. 2006). This has shown that at low levels of contamination, these wetlands may not effect any improvement. However, contaminant levels in shock loadings of effluent can be considerably reduced by a constructed wetland (Sukias et al. 2007). Overall, the effect is likely to be positive where there is a significant concentration of faecal microbes in the influent.

Water quality monitoring on an Otago deer farm showed reduced *E. coli* levels below a pond partway down a stream (McDowell et al. 2006). This was attributed to sedimentation and possible UV effects. However, if the microbes are stored in bed sediments there is potential for re-suspension, although pathogens such as *Campylobacter* are short-lived compared to faecal indicator bacteria. McKergow et al. (2007) also point out that creating ponds in landscapes can have detrimental environmental effects, such as barriers to fish passage and interruption of flow regimes, elevated water temperatures and reduced dissolved oxygen. Water fowl attracted to the ponds may also constitute a new source of faecal contamination. However, as noted above, pathogens originating from birds are less frequently associated with human illness than are ruminant pathogens.

6.9 Can mitigation practices achieve water quality standards?

Muirhead et al. (2008) modelled inputs and die-off of *E. coli* for a Toenepi "model" farm and concluded that if the farm were to implement stock exclusion and constructed crossings, and to switch from pond discharges to well-managed land irrigation, the impact of livestock faeces on microbial water quality would be minimal at base flows. (Ongoing sub-surface discharges between rainfall events were not included in this model.) During heavy rainfall events, though, these authors considered that the above practices would not be sufficient for the stream to meet recreational standards. Indeed, they concluded it is unlikely that any suite of mitigation practices would achieve this, as heavy rain would inevitably cause wash-in of part of the catchment load.

Overland run-off increases faecal contamination of waterways, and replenishes sediment stores of *E. coli*, although under high flow conditions faecal bacteria will also be flushed out of waterways. However, it is likely that much of the wash-in comes from old cowpats and land stores (if stock are not grazing close to waterways when heavy rainfall occurs), and could be dominated by hardy faecal indicator bacteria rather than pathogens like *Campylobacter*. Furthermore, some, but not all, water uses are less likely at high flows (commercial shellfish harvesting and recreation in particular). Therefore, if recreational standards can be met at base flow conditions through managing stock access and effluent, this would be a useful achievement.

The five "best practice dairy catchments" in various parts of the country have promoted catchment-scale adoption of mitigation practices over a ten-year period, with monitoring of water quality changes. At Toenepi, a completely pastoral catchment, reduction of faecal contamination to meet contact recreation standards was one of the catchment targets. However, the faecal indicator bacteria remained (mostly) above the 126 MPN *E. coli*/100 ml contact recreation target value for *E. coli* for the whole nine-year monitoring period , with no indication of change (Wilcock et al. 2006). There were some increases in farming intensity during this period, including an overall stocking rate increase of 7%. Milk production also rose 9% between 2001 and 2003, but figures for other years are not available. At the same time, pond discharges decreased in number from 22 to 9 in the catchment, and one of these is an advanced pond system (APS), known to significantly reduce faecal bacteria (Craggs et al. 2003). Although 46% of the main stream was fenced in 2002, Wilcock et al. (2006) noted frequent grazing of stock within the riparian zone even where there were fences. This suggests that switching a large proportion of effluent ponds to land irrigation and fencing without achieving full exclusion of stock from the riparian zone has not been sufficient to deliver water quality within guideline values in this catchment, as stocking intensity increases. This is confirmed by more recent data, showing that despite a further drop in pond numbers (with only 6 now remaining), median concentrations of faecal bacteria in monthly sampling at the lower Toenepi Stream from July 2009 - June 2010 were 464 *E. coli*/100 ml and 26 *C. jejuni*/100 ml (Donnison - author's unpublished data). It is possible that many of the *C. jejuni* at base flows are of wildlife origin. French et al. (2010) identified that the majority of *Campylobacter* isolates recovered from the Toenepi stream at base flow were sourced from birds, with a greater proportion of types sourced from ruminants at high flows. However, they also noted that although the majority of types recovered from the stream were not of cattle origin, a significant proportion of them were, including types that are known to cause human disease.

Another of the "best practice dairy catchments" is Waiokura in Taranaki. Before 2001, about 40% of Waiokura Stream had livestock excluded permanently. A fencing survey carried out in 2004 found that 61% of the stream length was fenced (i.e., both sides) but that 9% was occasionally grazed within the riparian zones, so that the net livestock exclusion from stream banks was 52% (Betteridge et al. 2005). A further 5 km of fencing and planting occurred from 2003-2008 (Wilcock et al. 2009). Since 2004, pond discharges have also decreased by 25-30%.

Analysis indicated that over the period from 2001-2008, median *E. coli* concentrations declined at an average rate of 116/yr, or a decrease from 1500 to 900 MPN/100 ml, being about a 40% reduction (Wilcock et al. 2009). It appears that in contrast to Toenepi, a concerted effort to improve stock exclusion and reduce pond discharges has had an effect at Waiokura. However, the absolute value is still considerably higher than in Toenepi and well above the recreational water quality standard.

The question therefore remains as to whether *E. coli* standards will continue to be breached even with sound mitigation practice in place, as *E. coli* is shed by all warm blooded animals including wildfowl and feral species. Furthermore, there is some debate as to whether *E. coli* counts accurately reflect human health risk in small rural streams, because they include indicator bacteria sourced from wild animals, yet the *Campylobacter* sourced from wild animals have been reported to present a lesser risk for human infection (Mullner 2009; French et al. 2010).

In summary, mitigation practices may not always be reflected in the expected reduction in *E. coli* due to the multiple sources in rural catchments and the complexity of routes of contamination. However, ruminant-associated pathogens are linked to human disease, so it is important that mitigation strategies are implemented to reduce the risk of introducing these pathogens into waterways. While it is difficult to control catchment wash-in during flood events, at base flows there are fewer pathways operating, so inputs are more manageable. Preventing direct deposition is particularly important because fresh faecal matter is likely to contain viable pathogens. Stock exclusion from streams and the use of constructed stock crossings are therefore key mitigation options. Other useful strategies to reduce base-flow contamination include stock exclusion from seeps and wetlands, and managing dairy effluent (deferred irrigation, low applications, avoiding high-risk soil types, or using advanced pond systems). In small rainfall events, riparian buffer areas can also be effective where run-off occurs in sheet flows.

7 Relevance of practices in the draft Regional Policy Statement

Certain practices have been identified in the draft Regional Policy Statement (RPS), released in December 2009, on the basis that they would have beneficial effects on water quality if adopted across the region. They have been identified for their potential benefit not only for faecal contamination, but for water quality and aquatic habitat generally. The draft policy suggests that the following practices should be avoided:

i. heavy stock and vehicles in and on the beds and banks of water bodies
ii. intensive grazing near water bodies particularly when soils are saturated
iii. riparian vegetation removal, and forest harvesting, tracking and earthworks near water bodies.

In Table 9, information is presented relating the findings of this review to these practices.

Table 9: Review findings in relationship to the practices for water body protection identified in the draft RPS (released December 2009).

Finding of this review	Relationship to the practices in the draft RPS
• Stock density affects *E. coli* numbers in waterways. • Cattle density is slightly more closely related to median *E. coli* levels than general stock density, but this is unlikely to constitute a significant difference in practice. Sheep faeces are an important source of faecal material, and dry-stock catchment loads can be similar to dairy.	This supports the inclusion of stock as a policy focus, including stock in and on the beds and banks of waterways and intensive grazing near water. The policy does not define what 'heavy stock' means. This review suggests that as a source of faecal microbes, cattle (including calves) and deer contribute both through run-off and direct deposition. Sheep are less likely to enter the water but contribute concentrated faecal material through run-off, including from riparian areas. No information was found on other livestock types.
• Transportation pathways by which microbes reach water are important. Direct deposition due to stock access contaminates waterways directly, while deposition on banks and in riparian areas is also highly likely to contaminate waterways after rainfall. Deposition in seepage areas continues to discharge to waterways through sub-surface flow.	This supports the focus on stock exclusion from streams, and from areas near streams that have high connectivity to the waterway and little attenuation opportunity. It also suggests that the inclusion of small order and ephemeral streams or wetland seepage areas would be technically justified, although it is acknowledged that this may pose greater practical difficulties.
• Direct deposition is a minor percentage of annual catchment yields, but is important because it occurs at base flows, when there is less dilution, and when downstream use is more likely. It also delivers viable pathogens directly to water, with no die-off effects. Catchment run-off during storms dominates annual yields and is more difficult to manage.	The focus of the draft RPS policy on areas near water can be expected to have some effect in most situations, and particularly at base flows, but catchment run-off and sediment microbe remobilisation will still occur in rainfall events. The inclusion of intensive grazing near water extends the focus beyond direct deposition and can address some critical source areas with poorly drained soils and high connectivity to water. In addition to preventing direct deposition in or near water, exclusion from the banks of water bodies could establish a riparian buffer zone. Retirement of this area might promote greater riparian filtration due to less soil compaction and thicker vegetation, depending on the location of the fence, width of buffer and catchment flow characteristics. The policy does not specify what constitutes "near water bodies". Wider riparian strips are likely to provide more microbial attenuation but come at a greater cost to the farmer, especially since occasional grazing of these areas is not recommended.
• Soil drainage characteristics affect	This supports indirectly the focus on areas

microbial contamination. This is not directly reflected in the draft RPS provisions, but poorly drained soils may be more common near streams or wetlands and these areas are targeted by the policy.	near waterways. Compaction would contribute to poor drainage conditions, so avoiding intensive grazing and vehicle traffic in areas near streams could be expected to have a beneficial effect.
• Riparian soils and vegetation can trap and retain faecal matter at low flows. Some die-off is expected, although the ultimate fate of specific microbes is not known - some faecal organisms can survive for long periods in the soil, but many pathogens are affected by exposure to sunlight and dehydrating conditions. Faecal indicator concentrations in outflows generally decline with time since grazing, showing that die-off is occurring.	This generally supports a focus on establishing riparian buffers to trap faecal matter and promote opportunities for faecal microbes to succumb to die-off. However riparian buffers are unlikely to be effective at high flows or where flow is channelised, or artificial drainage or by-pass flow through the soil predominates.
• Dense grass may be more effective at trapping and filtering than riparian trees and may offer more exposure to sun; however there is little data available to confirm this. One study showed cultivated soil to be more effective than grass at trapping microbes; however cultivating riparian areas would be expected to have detrimental effects on sediment run-off.	This may mean that practice III above (avoiding vegetation removal) may not enhance microbial trapping if it only refers to woody vegetation rather than grass (because grass may be a more effective trap than tree litter). The policy wording regarding “riparian vegetation removal” could potentially cover grazing of riparian grass, but this is not clear. However, retaining woody riparian vegetation has other beneficial effects on riparian and aquatic habitat.

Other factors and considerations arising from this review but not covered in the policy above include the following:

- Attention to the run-off of effluent from laneways and tracks, particularly near waterways or crossings, can be expected to benefit stream water quality, at least in a localised area.
- The use of pond treatment, and the location and management of sites for effluent irrigation and stand-off areas affect the likelihood of microbes in liquid effluent reaching waterways. Permitted Activity rules in the Waikato Regional Plan require that effluent applied to land does not reach waterways and the RMA precludes discharges to waterways from other activities without consent or permitted activity status. There are no specific rules stating required setbacks from waterways for effluent irrigation, or requiring particular management on poorly-drained soils or artificially drained soils, although best practice guidelines discuss this. However, when catchments across the region are compared, the number of effluent discharges to land is not strongly correlated with median *E. coli* in rivers, suggesting that irrigated effluent is less important overall in this region than faecal material deposited during grazing.
- Wild animals, including birds are a source of faecal material in the rural landscape, and pest control could be expected to have some effect on faecal indicator counts, although there is little published research on this. There is also some evidence that wild sources of *Campylobacter* are less likely to be associated with human disease than ruminant types.
- Other policies to reduce flow rates during rainfall events (e.g. by vegetating headwaters, or by protecting and increasing wetland areas) could also be beneficial in reducing transport and re-suspension of faecal microbes. However, no studies of the practical outcomes of these effects were found.

8 Conclusions

The following conclusions can be drawn from this review:

1. Faecal material from animals is a significant contributor to microbial contamination of the region's waterways. The density of stock is one of the factors most closely correlated with poor catchment microbial water quality. Dairy and dry-stock types (including sheep) deposit similar loads of faecal microbes per hectare. Ruminant types of *Campylobacter* are closely linked with human illness. Livestock deposit very large numbers of faecal microbes onto pasture and also deposit faecal material directly to waterways and riparian areas where stock exclusion is not in place.

2. Wildlife (feral animals and wildfowl) may also cause breaches of contact recreation standards in some locations. Activities to control pest animal numbers could be expected to have beneficial effects on faecal indicator counts, but this remains untested. Wild birds may be important sources of faecal organisms, particularly at times of base flow. Wild sources of *Campylobacter* are less closely linked with human illness than ruminant types. While there is an influence of wild sources in waterways, in terms of annual loadings this is mostly overshadowed by livestock sources.

3. Only a small proportion of faeces deposited on pasture ends up in waterways. However, standards for stock water, shellfish gathering and even contact recreation require low numbers of microbes, which are regularly exceeded in many pastoral waterways.

4. Most contamination occurs after rainfall as microbes are re-suspended from bed sediments and also transported from pasture and soils. The magnitude of the storm-associated peak concentration is often more closely related to the time elapsed since grazing than to flow or rainfall rates.

5. There are few mitigation practices that can address major storm flows. However, careful grazing of poorly drained soils near waterways, stock exclusion from riparian areas and attention to the drainage of "hotspots" of effluent accumulation can help manage the store of faecal material that is most likely to reach waterways, particularly in smaller events.

6. Stock exclusion and crossings are critical because direct deposition affects contamination at base flows, when contact recreation and commercial shellfish harvesting are likely to occur downstream. Direct deposition also introduces fresh, viable faecal microbes with no opportunity for die-off or entrapment. This is particularly important for the common pathogen *Campylobacter* which is susceptible to die-off when deposited on land.

7. With careful management of stock exclusion and effluent irrigation, there is some evidence that ruminant sources of contamination can be reduced at base flows. Eliminating direct deposition that occurs through stock access to streams and herd crossings can give significant (up to two thirds) reductions in contamination. However, this may not always be enough to comply with water quality standards (which are also affected by wild sources). In addition to exclusion from the main waterway, excluding stock from attractive areas with high connectivity to waterways (such as farm drains, deer wallows and seeps/small wetlands) is beneficial.

8. Riparian filter strips can be effective at trapping microbes, depending on factors such as length of strip, slope and topography of the surrounding land, soil types and flow characteristics. Riparian filters are most effective at low flows, and where drainage characteristics create run-off that is not channelised, but spread out as

sheet flow. Little is known of the effects of different types of vegetation on riparian filtration efficiency.

9. Point source discharges or the number of effluent discharges to land are not closely correlated with catchment microbial water quality. Some authors have suggested that effluent ponds are a significant source of microbes, but there is no clear Waikato evidence that supports their importance at a catchment or regional scale. Run-off to waterways from dairy effluent irrigation or sites of effluent accumulation should be avoided. Irrigating effluent over artificial drains or on heavy-structured soils with cracks and large pores carries the risk of sub-surface transport, which can be significant in areas with these types of soils. The risk of run-off or sub-surface flow to waterways can be reduced through adequate storage to avoid irrigating wet soils, and the use of low application systems. Constructed wetlands at the end of drains can also effectively remove microbes where they enter at high rates (e.g. from effluent spills).

10. This review generally supports the focus in the draft Regional Policy Statement on stock effects in and near water bodies, including access to the beds and banks of waterways and intensive grazing near water, particularly when soils are saturated. These practices are associated with the deposition of faecal material in and near water, and also a high risk of transport through run-off or sub-surface pathways in poorly-drained soils. This focus would be more effective if it included smaller waterways and ephemeral seepage areas, although it is recognised there are implications for farmers in implementing this.

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