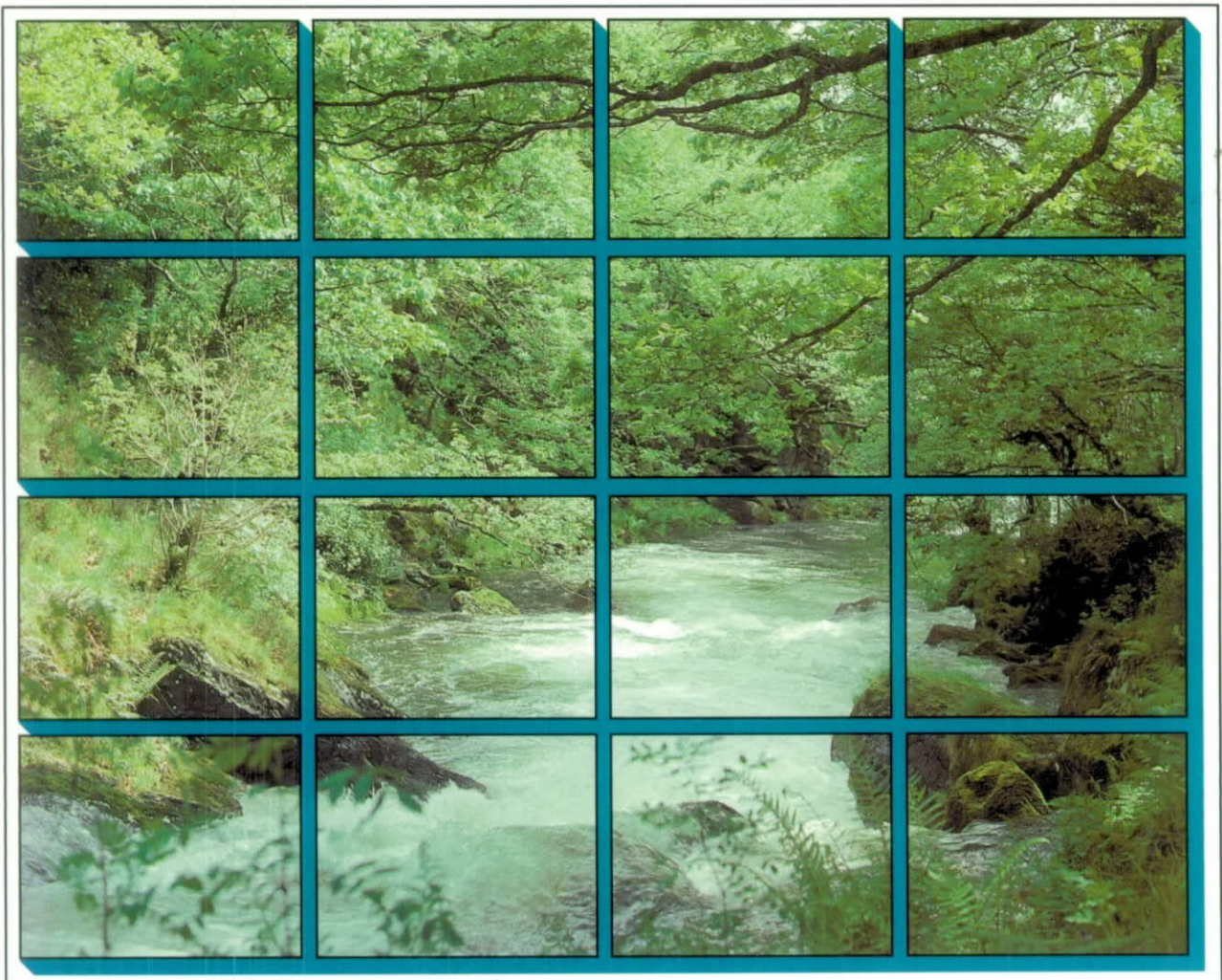# Report No. 127

Modelling faecal coliform concentrations in streams

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Modelling faecal coliform concentrations in streams

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October 1995

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Abstract

This report examines the mechanisms of faecal coliform transport and die-off on the land and in the aquatic environment. Equations derived from these investigations are used to develop a mathematical model of faecal coliform dynamics in streams and rivers. Process equations describing the key die-off mechanisms in the aquatic environment are derived from published data. A novel approach is used to describe the entrainment of organisms which have settled to the river bed: this is based on field observations during flow releases from reservoirs. A dead-zone approach is used to model mixing and dispersion mechanisms, as well as simple flow routing equations and terms specific to faecal coliform dynamics, including settlement and entrainment, and die-off due to sunlight, temperature and pH. The model accurately reproduces the field data and is successfully applied to routine monitoring data for the River Exe in Devon. Climate change scenarios are examined for the driest and wettest predictions for the year 2050: these result in a 5.6% reduction and a 1.6% increase in faecal coliform concentrations, respectively. A regional model, using GIS techniques to relate faecal coliform concentration to land-use, shows a strong correlation between agricultural land-use and faecal coliform concentration. The regional model could be used to describe diffuse inputs to the river model. Applications for this work include water quality planning to examine the effects of land-use change, changes in sewage effluent discharges, water supply protection, estimation of faecal coliform loads to the coastal zone, and assessment of health risks associated with recreational use of freshwaters.

Acknowledgements

Contents

Executive Summary

This report is based on the results of research carried out under contract for the Department of the Environment during the period 1991-1994*. The objectives of this work were to ascertain the key processes by which faecal coliforms are transported through river catchments, to suggest land-use impacts on stream faecal coliform concentrations, and to develop an integrated predictive model of faecal coliform concentrations from point and non-point sources. The work is of direct relevance to other research on bacterial water quality and will help to assess the risks associated with exposure to waters containing faecal contamination.

The key influences on the survival of faecal coliforms in streams and rivers include light and turbidity, temperature and pH. Sunlight is probably the most important single driving variable with regard to faecal coliform die-off in streams and rivers. A 90% reduction of an initial population in a few hours might be expected in bright sunlight whereas in darkness the organisms may persist for many days. The effect of solar radiation is reduced in turbid waters because light penetration is reduced and the organisms are shielded by an envelope of small particles. Temperature and pH play a lesser role in determining faecal coliform survival. Faecal coliform survival is extended in sewage contaminated or oxygen-stressed waters.

Examination of the processes of faecal coliform transport within catchments suggests that the significance of different processes and sources of faecal contamination changes with location. In headwater areas the supply of organisms is dominated by non-point sources: organisms are transported from the catchment surface by a combination of surface run-off and non-matrix throughflow in the subsurface zone during rainfall events. Further downstream the emphasis changes, as point sources and channel storage interactions become more significant in the supply of contaminant organisms.

Previous models for faecal coliform dynamics have used a range of approaches. Multivariate statistical approaches relate the bacterial concentration to a number of driving variables using simple statistical relationships. Simple deterministic first order decay functions have been used for describing the exponential die-off of a bacterial population and, in application to rivers, have been combined with equations to describe fluid mixing processes and flow hydraulics. These

models all lacked the necessary structure to describe the process of bacterial transport in rivers adequately. Only the model of Jenkins (1984) sought to describe the transfer of organisms to and from storage within the channel.

The new model presented in this report (Section 5) uses a mass balance structure similar to that adopted by Jenkins and can successfully reproduce the faecal coliform time-series produced during the field experiments described in Section 5.

The model structure and operation incorporates the following assumptions:

- The channel-store is distributed across the entire channel.
- The regions of storage respond sequentially to rises in flow.
- Any given rise in flow will produce entrainment of organisms from the channel.
- At any quasi-steady flow the active supply area of organisms will become depleted.
- No further entrainment can occur once the flow recession commences.
- Further higher flows will still release organisms from storage.

The model incorporates terms for the effects of environmental influences derived from data in the literature, describing the effect of sunlight and turbidity, temperature and pH on faecal coliform survival in the water column.

The model was successfully applied to a reach of the River Exe in Devon for the years 1990 and 1991. The model operated well for extended periods of data, the numbers of organisms in the channel store remained stable and were, in effect, self regulating. Seasonal effects were modelled with a simple cosine function accounting for die-off changes resulting from solar radiation and temperature, overcoming the need for data for these variables and reducing the number of parameters needed to calibrate the model. No previous model has given a satisfactory description of faecal coliform river dynamics. The model applied here not only gave a good fit to the observed data but also has scope for application to other water quality determinants.

Analysis of faecal coliform concentrations in 13 upland Welsh catchments, together with data from ADAS land-use maps and those derived from surveys of stocking practices and fertiliser use

showed that catchments with higher proportions of improved agricultural land (with higher fertiliser use and livestock densities) produce higher geometric mean faecal coliform concentrations than forested catchments. This finding reflects the higher loadings of organisms from livestock in agricultural catchments.

Examination of a further 12 catchments in England, Scotland and Wales, representing a broad range of size, land-use and faecal contamination, demonstrated behaviour consistent with the Welsh study. It was found that agricultural land classes and groupings of classes, perhaps relating to the lowland nature of the catchment, produce more faecal coliforms than more upland catchments with non-agricultural land-uses. The results suggest that the proximity of faecal coliform supply areas to the stream channel will be significant to the concentration therein.

Further studies will examine the relationships between faecal coliform concentrations and the more recent ITE land-use classification system which differentiates between grasslands used for pasture, and those used for rough grazing, etc. A greater number of sites will be used in this analysis thus enhancing results already presented.

The current version of the model is capable of simulating the changes in faecal coliform concentrations in a river network at both seasonal and storm event time scales. The potential applications of the model include: assisting the assessment of changes in effluent discharges or land-use on, for example, the health risk posed to recreators by a given river reach; assessing loadings of faecal contaminants to the marine system and hence the impact on compliance with health regulations relating to local bathing waters; and to help drinking water abstractors predict the timing, duration and magnitude of events of peak bacterial concentration in order to prevent the intake to water supplies with large faecal contaminant loadings.

1 Introduction

1.1 Background to the project

This project was funded by the Department of Environment with contributions from the National Rivers Authority. The main aims of the project were:

- to ascertain the key processes by which faecal coliforms are transported through river catchments;
- to suggest land-use impacts on stream faecal coliform concentrations;
- to develop an integrated predictive model of faecal coliform concentrations from point and non-point sources.

The work can be integrated with other research seeking to describe bacterial water quality and to assess the risks associated with exposure to waters containing faecal contamination.

The need to develop a predictive model for faecal coliform concentrations in rivers stems from the fact that existing water quality models only contain rudimentary process equations and the processes controlling faecal coliform concentrations were previously ill-defined. The ability to predict bacterial water quality and model bacterial transport has a number of benefits:

- Assessing the impact of changing land-use or effluent quality on bacterial water quality.
- Predicting loadings of faecal contaminants to the coastal zone to assist in assessing periods of poor bathing water quality.
- Determine the timing and magnitude of episodes of poor bacterial water quality associated with flow events in order to optimise the closing of water abstraction intakes to protect water supplies and reduce disinfectant dosings.

1.2 The coliform index as an indicator of faecal contamination

The coliform group was chosen as an indicator of bacterial water quality as a result of the work of Escherisch who, in 1885, identified *Bacillus coli* as being characteristic of the faeces of warm-blooded animals (Dutka, 1973). The group comprise; *Escherischia, Klebsiella, Enterobacter* and *Citrobacter* types but not all, however, are associated with faecal contamination since *Citrobacter* is not a faecal coliform.

The faecal coliform index, which is used in the field studies described in this report, was developed in an attempt to select only those types specifically of faecal origin: *Escherischia* and *Klebsiella*. Faecal coliforms or thermotolerant coliforms can be termed presumptive *E. coli* as all procedures in the incubation for the identification of *E. coli* (i.e. acid production) have been undertaken apart from the final confirmatory tests (i.e. gas production from lactose fermentation and indole production from tryptophan). The agency data used in the model application presented and in the examination of land-use relationships are also faecal coliform counts.

1.3 Sources and inputs of indicator bacteria to catchments

There are a wide variety of sources of indicator bacteria to catchments. Inputs to surface water may be categorised as either point or non-point sources (see Table 1.1). Non-point or diffuse source inputs of faecal contamination are essentially those derived from the catchment surface and represent the inputs to the river network which do not issue from a drain or pipe or other such easily defined point. The supply of organisms is determined by the land-use such as deposition by grazers, applications of farm wastes or sewage sludge, natural fauna, and recreational uses. The magnitude and significance of these sources will depend on the intensity of the stocking practice or application rate of waste and the degree of existing contamination in the receiving water. Point source inputs are localised sources which are easily identified and quantifiable. These include leachate from land drainage systems, sewage works outfalls, combined sewer storm overflows or industrial discharges.

Other less obvious sources of faecal contamination have been documented in the literature. Food processing waste waters (bacterial levels in vegetable processing wastes are attributed to the soils in which they were grown) the use of animal fertilisers, farm animals and poor quality irrigation water. In meat processing wastes, straw, blood, flesh, fat, offal and manure from the intestines of the slaughtered animals result in similar bacterial concentrations to the faeces of farm animals (Geldreich and Kenner, 1969). Standridge *et al.* (1979) found that outbreaks of various diseases on a lake-side recreational beach resulted from contamination

Table 1.1 *Sources and inputs of faecal contamination to catchments, with typical indicator bacteria concentrations*

Source of faecal contamination	Nature of input	Typical values, all in FC
1. Non-point sources	Diffuse inputs to the catchment surface, transported to surface waters by hydrological processes.	
Natural Inputs	Faeces of indigenous fauna e.g. rodents (FC per gram) [Geldreich and Kenner, 1969]	2.10^5
Avian	Inputs from nesting or roosting bird colonies. (FC per gram faeces) [Standridge, 1979]	3.10^7-4.10^8
Enhanced	Agricultural; use of farm animal wastes as fertilizers on arable and pasture land, dirty water reuse schemes for irrigation/disposal, intensive stock grazing.	10^9 per 100ml [Evans and Owens, 1972]
	Sewage sludge disposal; application to arable land and forestry as fertiliser. Recreational; directly from contact water sports, contamination of rivers draining catchments supporting enhanced recreational activity [Geldreich and Kenner, 1969].	10^7 per g human faeces
2. Point sources	Localised, easily quantifiable sources, such as direct piped discharges to a water body.	
Sewage works	Constant piped discharges of treated domestic (e.g. FC/100ml) and industrial wastes [Cohen and Shuval, 1973; Menon, 1985].	10-10^6 final 10^8 raw
Industrial effluents	Discharges from food processing industries; poultry/meat/potato plants, fruit/vegetable canneries etc. [Menon, 1985].	10^2-10^7 per 100ml
Urban runoff, storm drainage	Combined sewer storm overflows[1], storm drains[2], contamination from urban runoff[3], street gutters[4]. Storm induced inputs may result in extremely high "first-flush" concentrations due to scour of material in pipes. Storm sewerage may be outdated and operate before sufficient receiving water can be achieved. (All values per dilution by 100ml) [[1&2]Burm and Vaughn, 1966 (in [3&4]Geldreich et al., 1968)].	[1]10^7 [2]10^6 [3]10^3-5.10^4 [4]10^2-10^4
Landfill sites	Evidence exists showing that leachates contain faecal indicators and that bacteria can survive within sites for long periods [Blannon and Peterson, 1974].	10^6 per 100ml (In waste 10^5-10^7 per g)
Land drains	Discharge of bacteria in leachate from agricultural land with artificial drainage [Evans and Owens, 1972].	10-10^5 per 100ml

by a permanent water fowl population on the shore. Similarly, gull populations have the potential to lower the quality of night-roost water bodies as well as streams receiving run-off from nesting sites. Gulls may feed on agricultural land, at landfills and sewage works and breed on moorlands, often using reservoirs for safe night-roosts. One herring gull may excrete around 18.10^8 FC in a day: 19.10^8 is typical for a man (Gould and Fletcher, 1978). Sanitary landfills can pose a threat to the bacterial quality of ground and surface waters as both wastes and leachates contain high concentrations of

indicator bacteria (Table 1.1) (Donnelly et al., 1981;1982). Niemi (1985) examined the potential of fish farms as a source of faecal contamination. In a survey of effluents and water quality from various trout farms, the majority of the coliforms of faecal origin were traced to run-off from bird droppings on surrounding land. This might be expected: fish are not warm-blooded and do not have a permanent faecal coliform flora in their intestines. In remote regions their faeces rarely contain faecal coliforms and positive occurrences have been attributed to contamination from wildlife (Geldreich, 1970).

2 Relationships between environmental conditions, water quality and faecal coliform mortality

The survival of faecal coliforms outside the intestinal tract is influenced by a complex range of interacting factors. These might be physical and include the environmental conditions particular to a specific location or water body, the temperature or light intensity or moisture status. The chemical and microbiological hostility of a location is also significant, including the trophic status and degree of inorganic pollution, which in turn will be affected by environmental conditions.

Some of the studies reviewed use *E.coli* enumerations instead of faecal coliform. *Klebsiella* species, which comprise part of the faecal coliform count, will survive longer in the environment outside the gut than *E.coli*, leading to a relatively greater proportion of non-*E.coli* species in a faecal coliform count. Although this might lead to the reporting of slightly reduced survival times in such studies, evidence from microbiological laboratories analysing environmental samples suggests that between 80 and 85% of faecal coliforms in freshwater are *E.coli* (Godfree, 1994). Hence this should not detract from the overall conclusions since *E.coli* still represent the most significant group in the organisms contributing to the faecal coliform count in fresh waters.

2.1 Characterising bacterial die-off

To allow comparison of the results of bacterial survival studies, two main descriptors are used in the literature. These are the die-off rate coefficient k and T_{90} (the time taken for a population to fall to 90% of its initial value). The die-off rate coefficient k is derived from first order decay dynamics as described by Chick's Law, which describes a simple exponential decline from an initial population:

$$\frac{N_t}{N_0} = 10^{-kt}$$

where, N_0 = the initial population, and
N_t = the population at time $t > 0$

$$N_t = N_0 .10^{-kt}$$

for, $k = 0$ there is no die-off,
$k < 0$ indicates growth, i.e. 10^{-kt} is positive, and
$k > 0$ and increasing, die-off is more rapid.

The die-off rate coefficient, k, is the amount the log_{10} population falls per unit time, i.e.;

$$-kt = log_{10} \frac{N_t}{N_0}$$

$$-kt = log_{10}N_0 - log_{10}N_t$$

$$\therefore \quad k = \frac{log_{10}N_0 - log_{10}N_t}{t}$$

where, $t = t_0 - t_1$,

k can therefore be found graphically by plotting the log_{10} bacterial population against time and extracting from the y axis the two intercepts from the best straight line fit (see Figure 2.1) for one hour on the x axis. T_{90} is the time taken for the population to fall by one log_{10} cycle from the initial value. If $t=T_{90}$ or T_{99}, by definition, at T_{90}, $log_{10}N_0 - log_{10}N_{T90} = 1$.

$$\therefore \quad k = \frac{1}{T_{90}} \quad and \quad \frac{2}{T_{99}}$$

In practical applications there may be a time delay before a decline in the bacterial population occurs. In such situations k is the slope of the die-off portion of the graph, and T_{90} remains the total time from the beginning of an experiment to when the population has fallen by 90%.

McFeters and Stuart (1972) use $T_{\frac{1}{2}}$ as their measure of die-off, this being the time taken for a 50% reduction in the initial bacterial population. Assuming that simple first order decay is appropriate, $T_{\frac{1}{2}}$ values can be converted to die-off rate k;

$$T_{\frac{1}{2}}, \frac{N_t}{N_0} = 0.5 \quad \therefore k T_{\frac{1}{2}} = log_{10} 0.5$$

$$k = \frac{log_{10}0.5}{T_{\frac{1}{2}}}$$

3

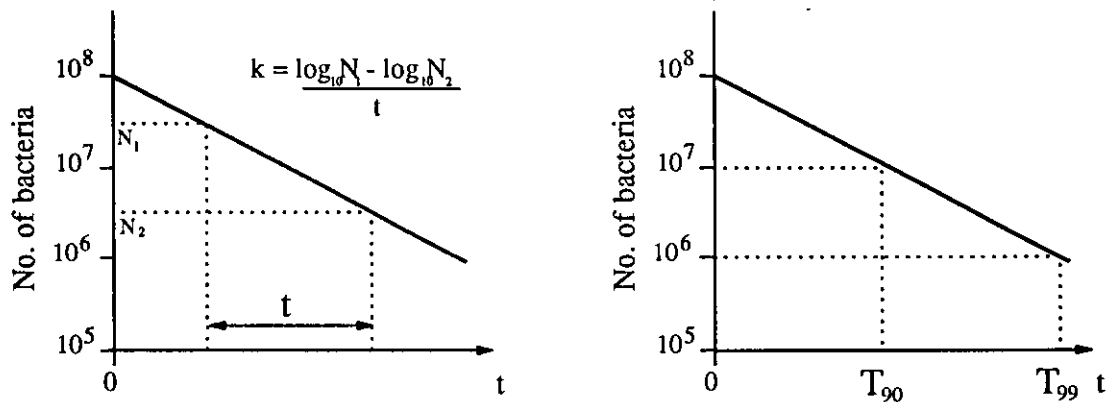$$k = \frac{\log_{10} N_1 - \log_{10} N_2}{t}$$

Figure 2.1 *Graphical derivation of die-off rate k and 90% reduction time T_{90} assuming simple first order decay dynamics*

Direct comparison of die-off rates taken from different studies should be treated with caution. Crane and Moore (1986) examined a wide range of data from studies of bacterial die-off in the environment. Die-off rates were found to be highly variable, spanning several orders of magnitude. Environmental factors were assumed to be the main cause of the differences but attempts to relate die-off rates to these using multiple regression were unsuccessful. Thus it was suggested that due to non-linearity in the effect of pH and temperature on die-off and the incomplete reporting of experimental conditions, quantitative definition of the effect of physical and climatic factors on die-off rates was not possible with the available data base. Recent papers, however, have made the development of such relationships possible (see Sections 2.2, 2.3 and 2.6).

2.2 Temperature and faecal coliform die-off rate

Faecal coliform survival is generally extended at lower temperatures. This is, however, more a consequence of the general reduction in biotic metabolism of the whole microbial community. Clearly, faecal coliforms multiply at the temperatures experienced in the intestines of warm blooded animals. Outside this environment die-off is enhanced at higher temperatures due to greater activity amongst the hostile natural biota.

Laboratory experiments using dialysis chambers inoculated with naturally occurring *E.coli* and using filtered stream water tested the change in *E.coli* population half-life over a range of temperatures. The results showed bacterial die-off to be proportional to temperature (McFeters and Stuart, 1972). A similar result was found by Kunkle and Meiman (1968) although they could not separate the effects of temperature and sunlight. Evison (1989) tested for the effect of temperature

on *E.coli* die-off on water from the River Coquet near the inlet to the Warkworth water treatment plant. This site is free from industrial pollution and suffers only slight sewage contamination. Samples were filter sterilised at 0.2μm, raw sewage was used as a source of *E.coli* and the effect of temperature was examined in the dark with samples inoculated with 0.25% raw sewage. Flint (1987) performed a similar study to Evison but compared the effect of temperature on die-off both up and downstream of a sewage effluent discharge.

The results of these studies are shown in Figure 2.2 which demonstrates a logarithmic relationship between die-off rate and temperature. Two distinct slopes occur according to whether the test water is clean or sewage contaminated. A general relationship for this type of curve is:

$$\log_{10} k_{T_2} = \theta . T_2 + \log_{10} k_{T_1}$$

thus

$$\theta . T_2 = \log_{10} k_{T_2} - \log_{10} k_{T_1} = \log_{10} \frac{k_{T_2}}{k_{T_1}}$$

therefore

$$k_{T_2} = k_{T_1} . 10^{\theta(T_2 - T_1)}$$

and the slope of the curve, θ, is given by;

$$\theta = \frac{\log_{10} k_{T_2} - \log_{10} k_{T_1}}{T_2 - T_1}$$

where, T_1 and T_2 are temperature in °C and k_{T_1} and k_{T_2} are die-off at those temperatures. In terms of model application, the slope of the temperature mediated die-off curve is more important than the actual die-off rate. Given a particular model
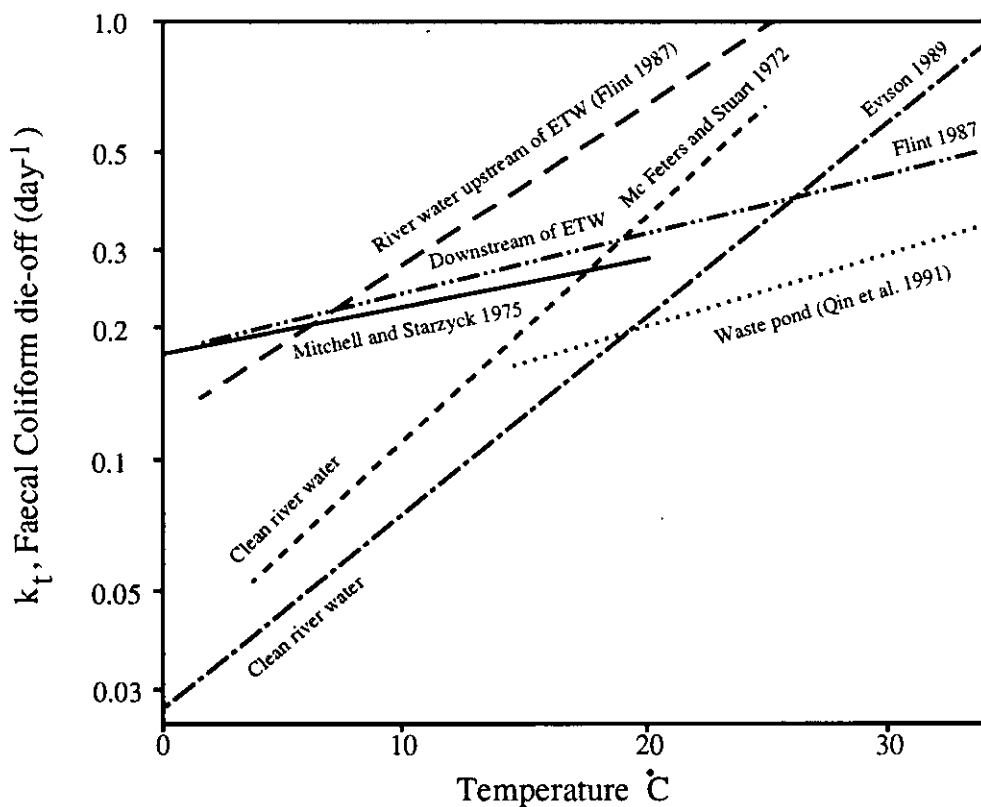
Figure 2.2 Plots of faecal coliform die-off rate against water temperature

calibration, the die-off coefficient will be set according to a range of influences and provided that an appropriate slope value is chosen, the die-off rate will self-adjust appropriately (Section 5.3).

In general, faecal coliform die-off increases with temperature, resulting from a combination of greater expiry rate due to enhanced metabolism as well as increasingly vigourous predation and competition by the naturally occurring microbial community (see Section 2.5). This relationship can be represented by a simple logarithmic relationship, the slope of which relates to the quality of the water at the target location, the effect of temperature being less marked in sewage impacted waters (Table 2.2).

2.3 The bactericidal effect of sunlight

The effect of sunlight on faecal coliform die-off is dramatic; T_{90} between one and two hours in bright sunlight and a number of days in darkness have been observed (Fujioka and Siwak, 1985). Similar results have been attained for containers suspended in a stream at shaded and exposed sites, die-off occurring in only two hours in exposed containers (Kunkle and Meiman, 1968). Van Donsel et. al. (1967) found die-off to be twice as rapid at an exposed lawn site compared to a shaded hillslope.

Light has both direct and indirect effects, the direct effect being cell damage, and indirectly through its effect on temperature or moisture.

Table 2.2 Summary of θ values taken from a range of studies

θ, slope of temperature affected die-off curve in clean and contaminated water			
Clean rivers		Sewage contaminated waters	
Source	θ	Source	θ
Evison (1988)	0.0449	Qin et.al. (1991)	0.0131
Flint (1987)	0.0381	Flint (1987)	0.0136
McFeters and Stuart (1972)	0.0511		
Mean value	0.0447	Mean value	0.01335
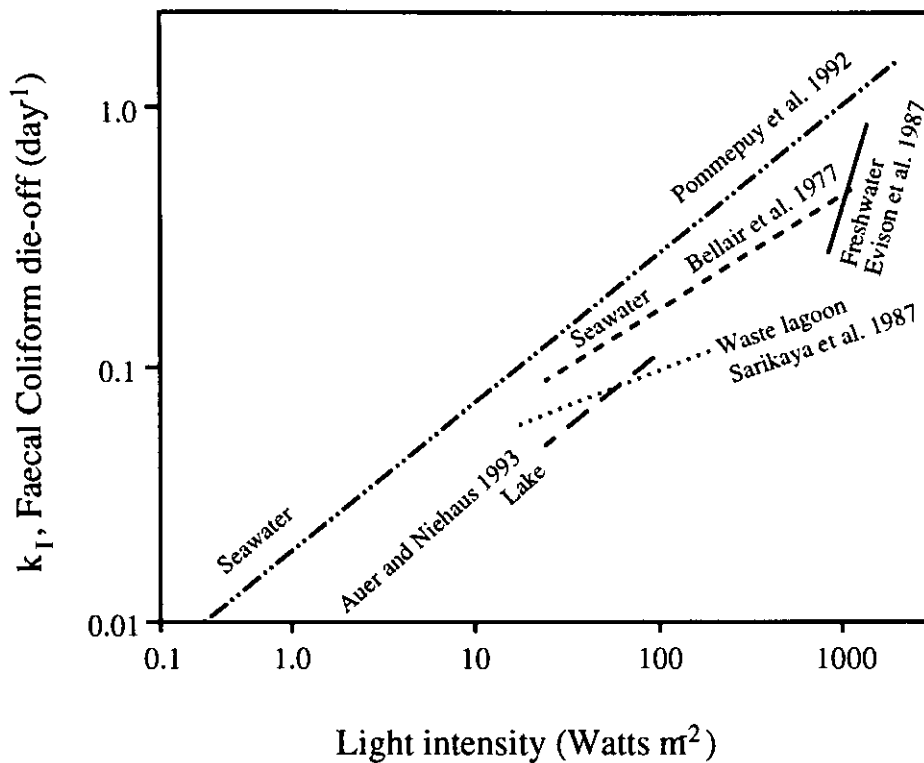
Figure 2.3 Plots of faecal coliform die-off rate against light intensity

conditions. In surface waters oxygen production and the excretion of algal toxins exerts a further stress on the contaminant organisms (Verstraete and Voets, 1972).

Figure 2.3 summarises the results of five studies examining the relationship between light intensity and faecal coliform die-off in both fresh and seawaters under both field and laboratory conditions. Light intensity values are expressed in $W.m^{-2}$ and die-off rates per day. This has required the conversion of much of the data; no common measure of light intensity has been used in these studies (Table 2.3).

Table 2.3 Conversion factors for comparison of light intensity values

Alternative Unit of Light Intensity	Equivalent Light Intensity in $W.m^{-2}$
1 $calorie.cm^{-2}.day^{-1}$	0.4845
1 $\mu Einstein.m^{-2}.hr^{-1}$	$6.3131.10^{-5}$

The effect of solar radiation on faecal coliform die-off is affected by the initial quality of the stream or water body. Verstraete and Voets (1972) showed that die-off due to solar radiation was greater in clean lake water than in heavily sewage contaminated water. Solar radiation accounted for 40-50% of the observed *E.coli* variation. In clean waters, good light penetration leads to optimal

die-off conditions through the direct lethal effect of sunlight, adequate oxygen supply to the hostile indigenous biota and production of toxins by algae. In poorer quality or perhaps deep waters, where light attenuation is significant, the benefits of sunlight will be reduced. Poor penetration, protection by adsorption to particulates and oxygen stress of the natural micro-community will result in extended faecal coliform survival. On the catchment surface, factors such as aspect, slope and vegetation cover will be important in determining the influence of sunlight on faecal coliform survival.

Auer and Niehaus (1993), studying faecal coliform dynamics in lake environments, developed equations relating depth averaged light intensity, $I_{z,avg}$ and die-off, k_I

$$k_I = k_d + \alpha \, I_{z,avg}$$

where k_d is the die-off rate under darkness conditions and α is the rate constant which might be derived from Figure 2.3.

$$I_{z,avg} = \frac{I_{0,avg}}{\eta . z} . \{ 1 - e^{-\eta z} \}$$

$I_{0,avg}$ is the mean light intensity at the water surface and η is the light attenuation coefficient. Pommepuy *et al.* (1992) demonstrate the relationship between suspended matter and light attenuation (Figure 2.4), giving the following relationship:

Figure 2.4 Plot of light attenuation factor,η ,per metre depth against suspended matter concentration (after Pommepuy et al. 1992)

$$\eta = 0.22 \, SM^{.78}$$

In fresh river water 73% of observed light attenuation was due to suspended matter; dissolved organic matter also contributed to this effect.

2.4 Soil type, moisture and faecal coliform survival on the catchment surface

In soils, and on the catchment surface, moisture is perhaps the most important factor in determining bacterial survival (Van Donsel et a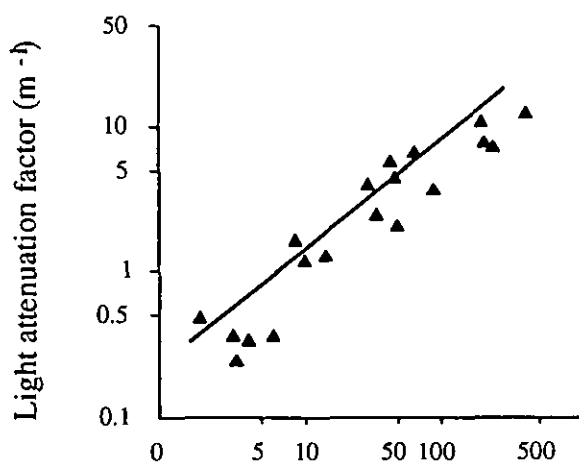l., 1967). Moist land areas contribute significantly greater numbers of faecal bacteria to surface waters than dryer areas (Hunter and McDonald, 1991a; Bagdasaryan, 1964) and survival is greatly extended during periods of water-logging (Chandler et al., 1981), see Figure 2.5.

Moisture content and retention properties of the soil are in turn influenced by other factors that may indirectly affect bacterial survival, for instance, soil type and vegetation cover (Beard, 1940; Hunter and McDonald, 1991a). Vegetation type, as an indicator of long-term soil moisture regime, could be used as a marker for zonation of a catchment into active and passive areas. Active zones include boggy/moist areas where bacterial survival is enhanced and run-off is actively generated. Reductions in bacterial inputs to surface waters could be achieved by preventing livestock access to such zones (Hunter and McDonald, 1991). Vegetation type and cover also influence soil moisture conditions. Dense sward affords protection from adverse environmental conditions, limiting the effect of sunlight, wind and
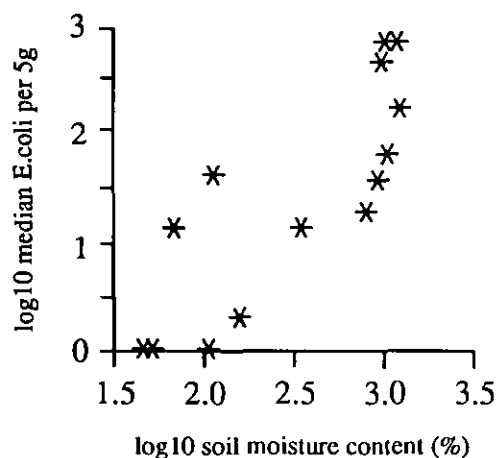
Figure 2.5 Scatter plot of log_{10} median E.coli count (per 5g) against log_{10} soil moisture content (% of dry weight) for a catchment in the Yorkshire Dales (after Hunter and McDonald, 1991)

high temperatures (Chandler et al., 1981; Zyman and Sorber, 1988). Waste application influences moisture supply and retention. Liquid animal wastes have high moisture content and good moisture retention properties (Crane and Moore, 1986) thus their application to such "active" zones might be avoided.

The nature of a soil, in terms of nutrient supply, moisture retention and particle/bacterium interactions may also be significant in determining faecal coliform die-off. In a three year study, Chandler et al. (1981) examined the persistence of faecal coliforms in soils varying from clay to loam, in areas with rainfalls from 500 to 1000 mm. Subsoil profiles were taken from plots irrigated with piggery waste at weekly intervals until contaminant organisms could no longer be detected, at which time further waste applications were made. The average concentration of the applied slurry was 3.10^5 faecal coliforms per 100 ml with mean dry matter content of 1.9%, the application rate approximating to a 30 mm depth of slurry. The overall T_{90} was 9 days for pastures and 11 days in top soil. These values were not significantly different at higher or lower waste application rates, nor was die-off significantly different six weeks after application. Faecal coliform concentration of the slurry was found to have a greater effect on soil concentrations than application rate. Of the 12 soil types tested, only one differed significantly (Chandler et al., 1981). The results suggest that such wastes not only supply vast numbers of bacteria but also, by maintaining moisture levels, provide conditions suitable for extended survival.

T_{90} values of three days and 25 days have been observed in sandy and loam soils, respectively.

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These results are attributed to the higher organic matter (OM) content and better moisture retention of the loam soil (Van Donsel et al., 1967). This effect may be a function of the relative sizes of the bacteria and particles since clay size particles (hundredths of a micron diameter) have been observed to coat the outside of the much larger bacteria (a few microns in size) affording protection from microbial predators (Roper and Marshall, 1978) (see Section 2.8).

Faecal coliform survival is enhanced in moist soils and in locations which favour the continuity of cool moist conditions, i.e. shaded, well vegetated areas in soils with good moisture retention.

2.5 Influence of natural microbial predation and competition on faecal coliform survival

The components of the natural microbial community, which have the most significant impact on faecal coliform die-off, vary between different aquatic environments. It is significant to note that the capacity for self-purification in sewage contaminated waters is reduced with respect to clean waters. The following section highlights a number of studies in this field indicating the importance of microbial purification processes in determining faecal coliform concentrations. The incorporation of such interactions into a working model would be difficult and indeed undesirable given the need to represent many relationships of uncertain causality, however, the general effects are modelled implicitly within the other die-off terms (Section 5.3).

The balance of microbial communities is maintained by a complex array of inter-relationships between populations within that community. One of the consequences of these interactions is the ability to eliminate populations of contaminant organisms including faecal coliforms (Verstraete and Voets, 1972). A number of studies have attempted to isolate the sections of the natural microbial community which cause the most rapid die-off of faecal coliforms, all showing that different microbes dominate die-off in different environments.

In tests on clean and polluted lake waters in Belgium successive filtering of water samples was undertaken to examine the effect of removing sections of the microbial community on die-off rates. Removals were carried out by filtering samples in the following sequence; algae and protozoa at 5μm, bacteria at 0.45μm,

bacteriophages[1] and other anti-microbial agents by autoclaving at 120°C for ten minutes. Samples were inoculated with E.coli cultured from natural strains and added to a nutrient preparation. The anti-microbial agents, referred to as thermolabile toxins, unstable and sensitive to heat, were shown to be largely non-diffusible and it was suggested that they could be a coliphage, a Bdellovibrio or a high molecular antibiotic. Further filtration (0.22μm), however, showed that they were not Bdellovibrios, which measure at least 0.3- 0.4μm. A dilution experiment eventually demonstrated that both coliphages and a potent colicidal toxin were present in the waters. In general, each removal resulted in successive reductions in die-off rate. There were marked differences between the clean and polluted waters. In the polluted water the greatest effect was noted by removal of bacteriophages and thermolabile toxins, while in the clean water it was indigenous bacteria (Figure 2.6). In the clean water the ability of the microbial community to reduce the E. coli population was greater than in the polluted water but the nature and origin of the agents acting in the polluted water remained unknown (Verstraete and Voets, 1972).

A more recent study discusses the role of ultramicrobacteria, identified in experiments similar in nature to those of Verstraete and Voets (1972), in destroying E.coli. In fact, this agent was found to act similarly to coliphage and the colicidal toxin mentioned above. Samples of river water from up and downstream of a sewage works outfall were successively filtered to remove fractions of the natural biota, firstly through Whatman No.1 papers, then through 0.45μm Millipore membrane filters and finally sterilisation by autoclaving. The filtrates and raw samples were then used in die-off experiments, with results expressed as T_{99} (Figure 2.1) and die-off coefficient k. Die-off was termed disappearance because, it was suggested, there was no evidence that E.coli had actually died. Little difference was observed in die-off between raw samples and those free of protozoa and suspended material, suggesting the minimal importance of these factors (Figure 2.7a and b). In clean samples, collected upstream of the sewage discharge, the removal of the majority of competing bacteria by filtration at 0.45μm had the greatest effect on survival. Downstream of the discharge removal of bacteria had less effect than the removal of phage and ultramicrobacteria (Table 2.4). These were found on microscopic examination to be very motile (Flint, 1987). The results are similar to those of Verstraete and Voets (1972) and suggest the importance of different microbial purification processes under different environmental conditions.

[1] A bacteriophage is a virus whose host is a bacterium. E.coli is known to be host to a number of bacteriophages (Singleton and Sainsbury, 1981).
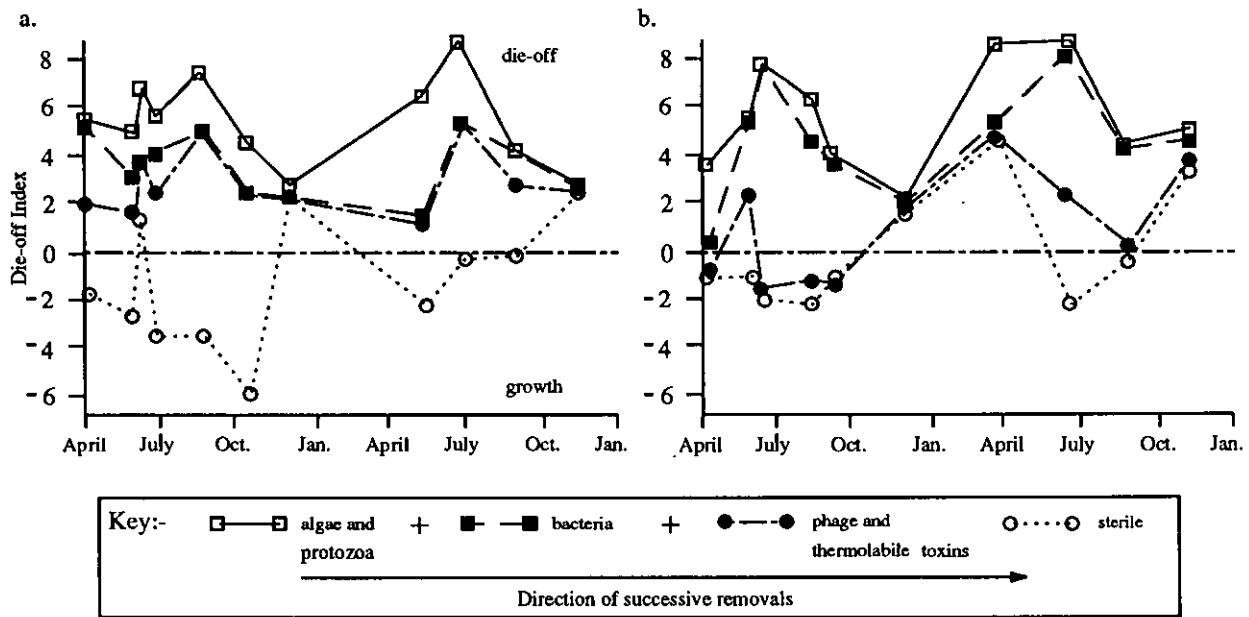
Figure 2.6 Die-off of *E.coli* in (a) sewage contaminated and (b) clean water bodies, following removal of protozoa and algae (dashed line); protozoa, algae and bacteria (dot-dash); filtration at 0.45μm and autoclaving (dotted line); raw samples (continuous). After Verstraete and Voets (1972).
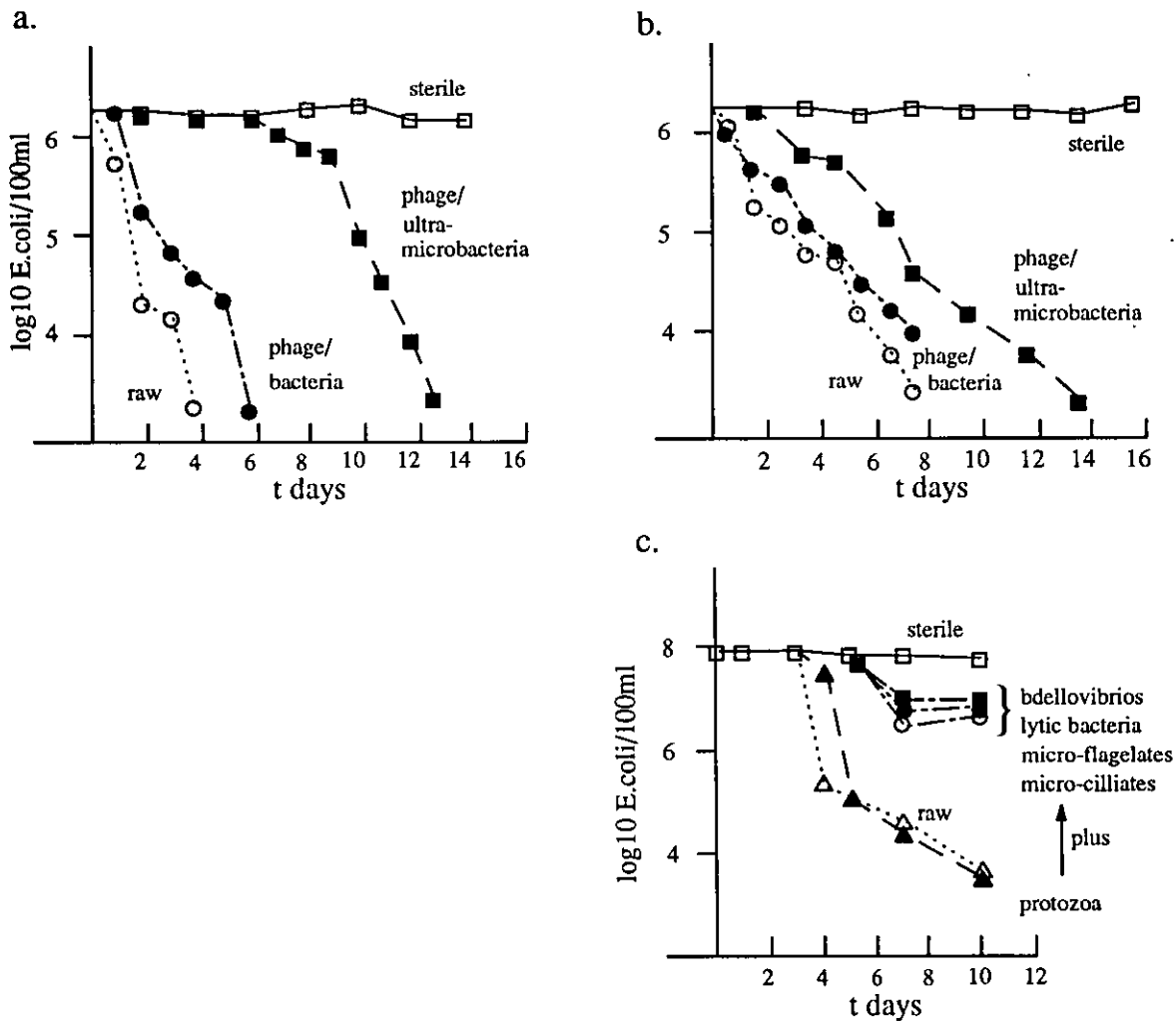
Figure 2.7 The influence of successive removal of sections of natural microbial communities on *E.coli* die-off in (a) "clean" river water, (b) sewage contaminated river water (after Flint, 1987) and (c) estuarine water (after Enzinger and Cooper, 1976)

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Table 2.4 Die-off rate k (d⁻¹) and percentage reduction in die-off rate (in brackets) in clean and sewage contaminated river water following serial removal of natural biota by filtration, temperature=15°C, after Flint (1987).

	Unfiltered	Protozoa and SM removed	Removal of bacteria	Autoclaved
Clean river water	0.47 (0.0%)	0.44 (7.0%)	0.0286 (94.0%)	0.0 (100.0%)
Downstream of sewage discharge	0.25 (0.0%)	0.2 (20.0%)	0.105 (58.0%)	0.0 (100.0%)

Enzinger and Cooper (1976) reviewed a number of early studies finding that *Bdellovibrio bacteriovorous*[2] had been associated with increased die-off of *E.coli* in natural waters. This was thought to be unlike phages which require an actively growing host, a requirement not usually met by *E.coli* when released into the hostile natural environment. Protozoan reduction of *E.coli* in sewage was thought to result from the action of motile strains or *ciliates*. In experiments with estuarine water, Enzinger and Cooper (1976) tested the effect of removing protozoa and bacteria on *E.coli* die-off. In separate tests, protozoa were removed from samples by membrane filtration and indigenous bacteria were removed by antibiotics. Antibiotic resistant *E.coli* strains were used as the test organism in the latter test. *E.coli* die-off was most rapid in the presence of the complete indigenous population and related to the development of predator populations. Die-off following treatment with antibiotics to remove the naturally occurring bacteria was almost as rapid. Filtration to remove protozoa had a much greater effect, reducing die-off considerably (Figure 2.3).

The natural biota were fractionated further by filtration at the following pore sizes: with 0.22μm, filtrates were generally free of natural organisms and *E.coli* numbers in these samples remained near constant; with 0.45μm, only *bdellovibrios* were present in filtrates causing a slight increase in die-off rate. Results for 0.8 and 1.2 μm filtrates were similar; these contained *bdellovibrio* plus one or two other bacteria capable of *E.coli* destruction. The 1.2 and 3.0 μm filtrates both contained numerous *micro-ciliates* and *micro*-flagellates. Only the 3μm filtrates contained protozoa and die-off was most rapid in these. The time-lag preceeding the onset of die-off in these experiments was probably due to the time taken for the protozoan population to reach sufficient density to effect a detectable removal of *E.coli*. Protozoa thus exerted the greatest influence on survival in these samples of estuarine water and may also exert pressure on bacterial populations in other environments (Enzinger and Cooper, 1976).

Brettar and Höfle (1992) studying lake mesocosms found that a range of grazing organisms flourished in a series of stages after the introduction of *E.coli* to that system. During the first three days of the experiments, *E.coli* reduction, it is suggested, was dominated by motile protozoa referred to as flagellates. These are organisms which can move around of their own free will. The *E.coli* cells were free-living, i.e. not settled or attached to particles, during this period. An assessment of the target particle size for flagellates showed that they graze on particles of about the size of free-living *E.coli*, 0.8 to 1.2 μm. Following the elimination of the majority of the free living *E.coli* by flagellates, a population of larger, multi-cellular rotifers, *K.cochlearis* and *Conchillus unicornis*, developed. These organisms graze on particles of about the size of *E.coli* upto 20μm. After ten days macrozooplankton of the *Daphnia* species became dominant. *Daphnia* feed on particles of up to 35μm. These organisms eliminated the remaining largely particle associated *E.coli* numbers. Phage was not found to be important in determining *E.coli* reduction in this instance. This study showed the rapid removal of free-living *E.coli*. Particle associated organisms, being eliminated more slowly, benefit from advantages resulting from nutrient availability on particle surfaces and the ability to settle into the favourable environment of the bed sediments (see Section 2.8).

Clearly, natural microbial communities can exert considerable stress on *E.coli* populations. Different organisms will be "survival limiting" in different environments. In clean freshwaters natural bacteria cause the most rapid die-off. In sewage-contaminated waters phages, ultramicrobacteria and heat sensitive toxins cause the greatest die-off whereas in estuarine waters faecal coliform die-off may be dominated by protozoan predation. In polluted waters a reduction in competition might result from the abundance of nutrients or increased survival may result from the stress exerted on the natural biota by high numbers of alien microbes and the reduction in the physical and chemical quality of the environment. It is likely that similar

[2] A parasitic bacterium whose hosts include *E.coli*.

behaviour occurs on the catchment surface and sub-surface however, the relative impact of different indigenous microbes on bacterial survival in these environments is not well documented.

2.6 The influence of acidity (pH) on faecal coliform survival

Extremes of pH have been observed to enhance faecal coliform die-off in soils and freshwaters. Yates and Yates (1988) noted that pH3-4 has a detrimental effect on bacterial survival in soils and water. Acid mine waters, approximately pH3, cause rapid faecal coliform die-off. In in-situ tests, reductions from 10^6 to 100 FC.100 ml^{-1} were observed in two hours, which approximates to a T_{90} of half an hour compared with neutral waters in which the reductions observed were negligible (Carlson-Gunnoe et al., 1983). Similarly, in acid peat soils of pH2.9-4.5, die-off within a few days was observed, as opposed to several weeks in limestone soil. Low pH may not only affect faecal coliform survival ability but also nutrient availability and the action of antimicrobial agents (Cuthbert et al., 1955). Cohen (1922) found that optimum survival was promoted at around pH5-6.4. McFeters and Stuart (1972) examined E.coli die-off on either side of neutral pH. Using dialysis chambers in tests conducted at 10°C the greatest survival was observed between pH5.5 and 7.5, die-off increasing sharply beyond these values (Figure 2.8).
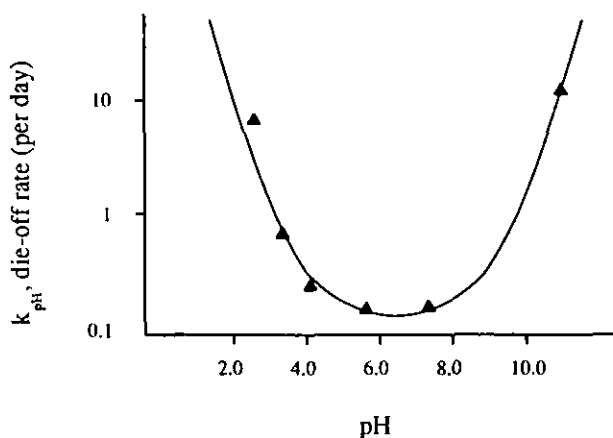
Figure 2.8 Plot of observed (after McFeters and Stuart, 1972) and modelled (solid line) of E.coli die-off rate per day against pH

There are several mechanisms of cell damage resulting in enhanced faecal coliform die-off under extremes of pH. These include hydrogen ion absorption to cell walls under acidic conditions, the inhibition of cation (Na$^+$, K$^+$, Ca^{2+}) replacement within cells and the leakage of potassium and other compounds essential to normal cell function. In alkaline situations the

binding of heavy metals to the cell membrane may also cause death or injury by inhibiting the transfer of a variety of metabolites (McCalla, 1964; Singleton and Sainsbury, 1981).

These observations suggest that a general relationship between pH and faecal coliform die-off can be represented by a simple hyperbolic cosine function of the form;

$$k_{pH} = k_{min} \cosh \{a \, (pH_{k_{min}} - pH) \}$$

Where, pH_{kmin} is the pH at which die-off is at a minimum (k_{min}) and a is a constant of proportionality. In the case of McFeters and Stuart (1972);

$$k_{pH} = 0.135 \cosh \{ -0.445 \, (6.5 - pH) \}$$

and values for k_{pH} derived from this equation are plotted (solid line) in Figure 2.8.

2.7 Dissolved oxygen content

It has already been suggested that the trophic status of a water body can influence faecal coliform survival (Section 2.5). High nutrient loadings lead to oxygen stress in the natural microbial community resulting in extended faecal coliform survival (Figure 2.9). In well aerated water die-off is enhanced (Zerfas, 1970). Verstraete and Voets (1972) found die-off to be positively correlated with dissolved oxygen content in sewage contaminated waters. Similarly oxygen depletion resulting from ice-cover has been shown to enhance bacterial survival (Hirn et al, 1979; Gordon, 1972; Davenport et al., 1976).
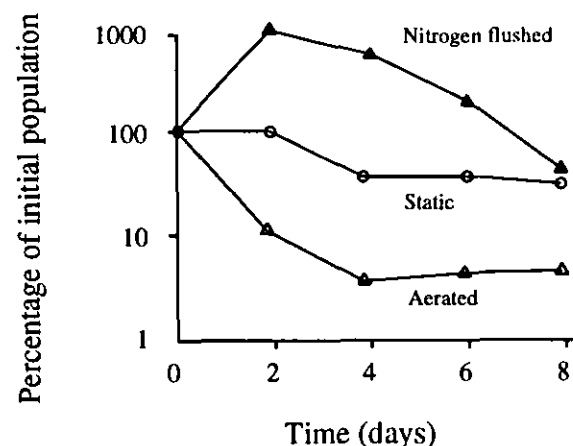
Figure 2.9 Survival of faecal coliform in raw river water at 20°C in aerated, static and N$_2$ flushed flasks (after Zerfas, 1970)

These studies show that faecal coliform die-off is more rapid in healthy and oxygen rich waters, however, there is to date no data available in the

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Table 2.5 Bacterial survival in different sediments (Burton et al., 1987; *Sherer et al., 1988)

Clay/silt/sand ratio	% Organic matter content	Total Kjeldahl Nitrogen (ppm)	Total Phosphorous (ppm)	E.coli die-off rate k(hr⁻¹)
75 : 25 : 0	14.8	3.18	9.98	0.126
28 : 55 : 12	6.2	16.32	6.90	0.236
25 : 51 : 24	5.2	13.97	14.30	0.167
12 : 76 : 11	9.0	24.10	7.70	0.330
2 : 0 : 98	0.7			0.319
34 : 34 : 32				*$0.666 \cdot 10^{-3}$
14 : 12 : 4				*$0.791 \cdot 10^{-3}$

literature suitable for the development of a general mathematical relationship between dissolved oxygen and die-off.

2.8 Suspended matter, sediments and nutrients and other effects

Suspended matter, sediments and associated nutrients play a significant but complex role in determining faecal coliform survival and transport dynamics. The role of suspended matter in attenuating light propagation through the water column and the ability of clay minerals to form an envelope around a bacterium and thus afford protection from predation has already been discussed. Bacterium/particle interactions are important in determining transport within river channels (Section 3.1); sedimentation of clumped or particle associated organisms is responsible for apparent die-off in the water column. Settled organisms accumulate at the water/channel-bed interface forming a "channel-bed store" where relatively stable environmental conditions suit extended survival and later disturbance may cause resuspension into the overlying water and transport downstream. There has been much study of water column/channel bed sediment relationships but faecal coliform survival in channel bed storage is poorly understood.

The main limitation of many studies examining survival in channel storage is the lack of representative conditions through the use of sterile samples and inoculation with effluents which are unlikely to contain the range of organisms that would be expected under field conditions. In the absence of the natural microbial community, growth of enteric bacteria has been observed at temperatures above 10°C in low nutrient mountain stream water and more rapidly in nutrient extracts from the stream bottom

sediments. It has been suggested that extensive growth may occur in bottom sediments. This is an environment less sensitive to diurnal effects where nutrients can be in high concentrations relative to free flowing water (Hendricks and Morrison, 1967; Verstraete and Voets, 1972).

Allen (1953) found that coliform survival in different mud samples was quite uniform and that the organisms were concentrated in the top 5 cm of sediment. Jenkins (1984) sampling for E.coli in the River Washburn in the UK showed that conventional grab sampling of stream bed sediments for bacterial analysis diluted the surface concentration. By means of a suction method, the organisms were shown to be concentrated at the water/channel-bed interface. Recent studies have attempted to address the problem of faecal coliform mortality in sediments. The die-off rates from the two studies are approaching an order of magnitude difference (Table 2.5). In neither study do conditions approach those that might be encountered in the field. Sherer et al. (1992) mixed cow manure with their sediment samples, introducing the liquor and organic matter from the manure to the sediment, as well as distributing the organisms evenly throughout the test samples. It is likely that if cow manure actually entered a stream in a raw form the various components would separate, the liquid component would become diluted into the flow and the solids would disperse and perhaps settle on the surface of existing deposits. The die-off rates observed in this study were very low, the experiments were undertaken in darkness at 8°C. Given that the addition of manure provides an environment derived from the parent faecal material, rich in nutrients and lacking indigenous stream biota, the slow die-off rates might be expected. E.coli concentrations have been found to correlate with phosphorous, nitrogen, and organic nutrients (Hirn et al., 1980). The results of

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Burton et al. (1987) are at the other extreme. This study attempted to mimic the physical and chemical conditions in different freshwater sediments using a continuous flow laboratory microcosm. The requirements for adequately simulating field conditions were, however, only partly met. The water used to circulate over the sediment samples, although reconstituted to the major ion concentrations observed in the field, was sterile. The sediment samples were stored at 4°C for up to two weeks before use, the test organisms were initially distributed evenly throughout the sediment samples and no reference was made to the lighting conditions used during the tests. The only sediment characteristic for which there was an apparent relationship with die-off was particle size; *E.coli* surviving longer in sediments with at least 25% clay content (Table 2.5). Stephenson and Rychert (1982), suggested that organic matter content may have a critical influence on the survival and/or multiplication of the bacteria in sediments. This suggestion was not supported by the results of Burton et al. (1987), who proposed that the lack of a relationship was due to the variable nature of organic matter and other influences on survival.

Roper and Mitchell (1978) demonstrated the protection of *E.coli* from *bdellovibrio* by montmorillonitic clays; electron-microscopy revealed that *E.coli* cells became enveloped in a thick layer of clay capable of excluding *bdellovibrio*. Colloidal montmorillonite offered less protection than crude montmorillonite which may form a more complete envelope around the bacterium. The experiments were undertaken using seawater diluted to 695μS conductivity, which is typical of many UK rivers. It is possible that this coating effect had some influence on the results of Burton et al. (1987).

Findlay et al. (1990) observed a self-protection mechanism in estuarine waters. In the absence of sediment particles *E.coli* were found to aggregate forming an inner core protected from the osmotic stress caused by salt water.

Tests with sterile seawater have shown that the provision of adequate nutrients or suspended solids enhance *E.coli* survival and that these effects are mutually exclusive. The same nutrient concentration was used in all of these experiments, which were carried out at 20°C and 5°C. At 20°C maximum die-off was observed at suspended solids (SS) concentration of 12.5 mg.l^{-1}. At higher or lower SS concentrations survival was extended. The suggested explanation for this behaviour was that at low SS concentration good nutrient availability enhanced survival. High SS concentration also enhanced survival as a result of an adsorption-protection mechanism. At the point

of maximum die-off the available nutrients were preferentially adsorbed to the available particle sites, lowering both nutrient availability and protection affordable by adsorption. At 5°C the addition of SS to samples had a more marked effect. Peak die-off occurred at 5 mg SS.l[-1], this may have been due to slower bacterial metabolism and diminished ability to utilise the available nutrients at the lower temperature (Milne et al., 1991).

Table 2.6 *E.coli* die-off rate in filter sterilized water from the River Coquet containing different proportions of sterile sewage effluent (after Evison, 1989)

% Sterile sewage concentration	0.025	0.25	2.5	25
Die-off rate (d[-1])	0.136	0.316	0.279	0.043

Table 2.6 shows the results of Evison (1989) demonstrating extended *E.coli* survival under high and low nutrient concentrations. The sewage effluent nutrient supplement used in these tests was autoclaved and the ratio of particulates to nutrients can be presumed to have been the same in each test. This perhaps rules out a nutrient/particle relationship with die-off, the observations resulting from a metabolic effect, whereby at low nutrient concentrations the organisms survive longer thus reducing their metabolism. With the high nutrient levels the organisms can multiply sufficiently well to maintain their populations for longer. At the intermediate concentrations the organisms maintain a normal metabolism without multiplying sufficiently and hence die-off more rapidly.

Brettar and Höfle (1992) found that extended long-term survival in samples with added growth medium resulted indirectly from the development of algal particles offering survival niches for *E.coli* cells. This effect was thought to be more significant than any protection afforded by the organic matter provided by the growth medium.

E.coli cells can, under conditions of nutrient starvation, enter a state of dormancy whereby they cannot be detected by culture enumeration methods but can be shown to remain viable by direct counts and capable of returning to a culturable state when conditions are appropriate (Brettar and Höfle, 1992; Roszak and Colwell, 1987).

The effects of particulates and nutrient supply on faecal coliform survival are complex, the most tangible of these being the reduction of light penetration into the water column, the shielding from light and predators afforded by a coating of fine particles and the enhancement of settling

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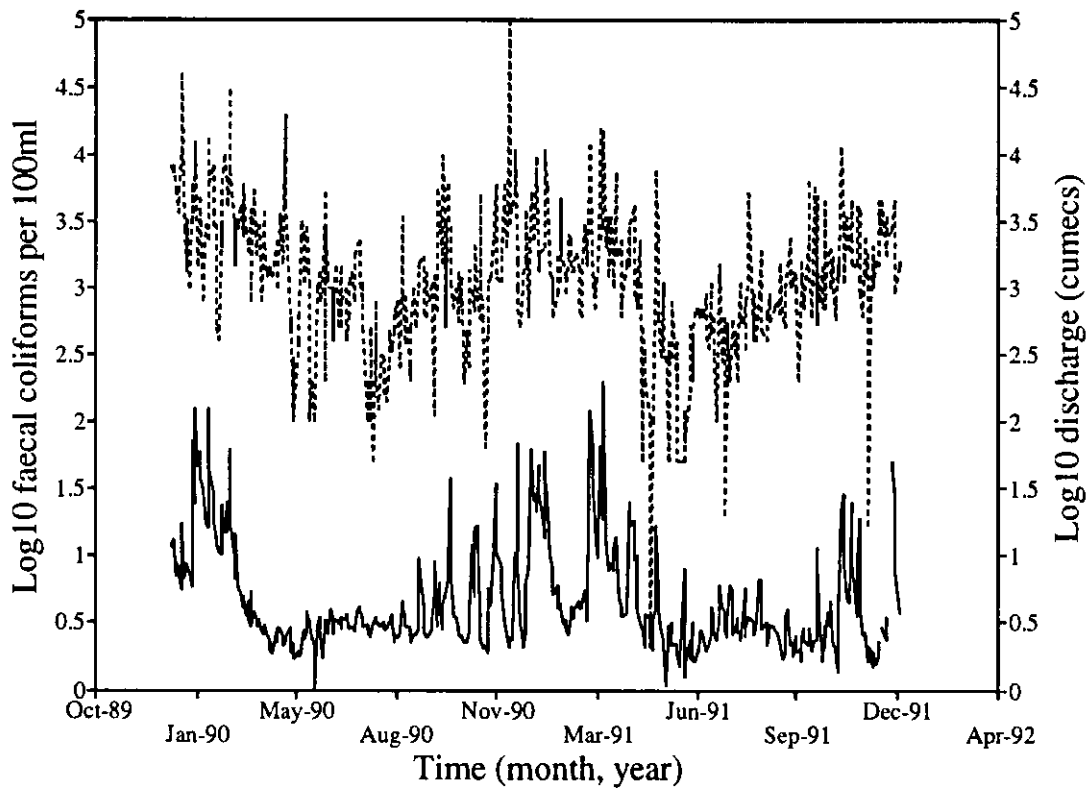
Figure 2.10 Plots of $\log_{10}$ faecal coliform concentration (dashed line) and $\log_{10}$ discharge (solid line) for the River Dee at Huntington showing seasonal variation between the calendar years 1989 and 1992

properties. Survival at the channel-bed/water interface is greater than that in the water column; channel-bed storage represents a major source of organisms capable of lowering the quality of the overlying water when suitably disturbed. Attempts to quantify this survival have produced wide variations in die-off rate as a result of the inability to create realistic conditions in the laboratory.

2.9 Seasonal behaviour

In general the seasonal variations in faecal coliform concentrations observed in a wide range of studies correspond to the hydrological year (Davenport et al., 1976; Gordon, 1972; Hirn et al., 1979).

The highest concentrations are observed in the winter months during periods of higher flows. In the summer months concentrations are much lower (Figure 2.10). The explanation for this behaviour is relatively straightforward although the exact causality is difficult to determine given the number of factors involved. The main

difficulties are in determining whether the variation is supply/transport dominated or die-off dominated. Section 3 of this report examines faecal coliform transport processes from the catchment and in the stream channel. Given an understanding of these supply/transport mechanisms and the die-off processes discussed above it can be seen that both groups of processes will lead to enhanced concentrations during the winter months and vice versa.

The factors that will lead to extended winter survival of organisms include: fewer daylight hours, lower temperatures, moister land surface, shorter residence times in each river reach and protection from light and predation by particulates. Supply and transport factors include rapid transport from the catchment surface, more frequent operation of storm sewage overflows and frequent scouring of settled organisms. Both the die-off and transport processes act to cause higher bacterial concentrations. In the summer months the effects are reversed, die-off throughout the catchment is enhanced and low flows result in minimal transport within the catchment.

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3 Storage and transport of faecal coliforms in catchments

We now examine how the organisms are stored within and transported through a catchment, from deposition on the land-surface to transport in the riverine environment.

On the land-surface, bacteria may remain locked-up in parent faecal material for extended periods. Once released they can become adsorbed on to soil and organic matter and survive for long enough to provide a semi-permanent reservoir capable of contaminating the surrounding aquatic environment following transport by hydrological processes (Hunter and McDonald, 1991a). Stored faecal bacteria have been shown to be released up to four months after being deposited (Evans and Owens, 1972). In lowland areas, transport by water infiltrating the soil mass may account for the majority of bacteria reaching surface waters from non-point sources, although these may be insignificant compared with inputs from point sources or inputs from bacteria stored in the stream bed. In upland areas bacteria are transported by surface runoff as well as matrix and non-matrix soil through-flow. During base-flow conditions inputs may be provided by return flows and matrix through-flow. The relative importance of these pathways depends on the nature of the catchment, the antecedent soil moisture status and the occurrence of rainfall, its duration and intensity. Bacteria enter a water body with solids or free in suspension. In stream and river channels bacteria are stored in the bed, attached to particles, plants and surfaces such as rocks. Transfer to channel bed-storage is by settlement, either directly, or attached to particulates, or as flocs. Low density particulates, such as organic solids and flocs, will remain in suspension at low flow velocities as will associated bacteria. The transport dynamics are likely to be dependant on the nature of the fluvial system. In fast upland streams the transport dynamics will be dependant on flow, whereas in large and slow flowing lowland rivers the general bacterial behaviour may be more likely to be dependent on supply and die-off processes.

3.1 The process of adsorption and the attachment of faecal coliforms to different substrates

The process of adsorption or attachment is important in understanding storage and transport of faecal coliforms within catchments. Bacteria, including faecal coliforms, can become adsorbed or attached to any solid surface, i.e. sediment particles, rocks, plant and animal surfaces, organic matter, and to each other (Marshall, 1979). These organisms may also become coated by clay particles (Roper and Marshall, 1978). A description of the mechanisms underlying this attachment process assists in the understanding of bacterial dynamics in catchments.

The solid surfaces at the interfaces between objects generally have net negative charge and as a result attract cations which occupy the charge sites and effectively cancel out the negative surface charges. A layer of strongly adsorbed cations a few nanometres thick is formed at the interface surface, known as the Stern-layer. Beyond this layer the electrostatic forces decay almost exponentially, as does the difference in numbers of cations to anions. This zone is termed the diffuse layer (Marshall, 1979; White, 1979). The concentration of cations at particle surfaces causes repulsion between particles because of the like electrostatic charges. However, non-electrostatic forces also act between particles. These are Van Der Waals forces which have a weak attractive effect that exceeds the electrostatic repulsion in all but very low ionic strength solutions.

In low ionic strength waters electrostatic repulsion exceeds Van der Waals attraction at all but the very closest particle separations. For adsorption to occur the surfaces must actually make contact or collide with each other (Figure 3.1 (b)). In most freshwaters however, the salt concentration is high enough to reduce the extent of the diffuse layer, allowing the Van Der Waals attraction to exceed electrostatic repulsion. The particles or surfaces come to rest at a distance where the attraction and repulsion forces are in equilibrium. By this mechanism bacteria may become loosely adsorbed or attached at an interface and are desorbed by the application of a suitable shear. force (Marshall, 1979). Evidence for this kind of loose adsorption was noted by Grimes (1974) who observed that the disturbance and relocation of .. bed sediments by dredging operations caused a release of organisms.

Mineral sediment particles have a definable adsorptive capacity determined by surface area to volume ratio. The greater this ratio the larger the
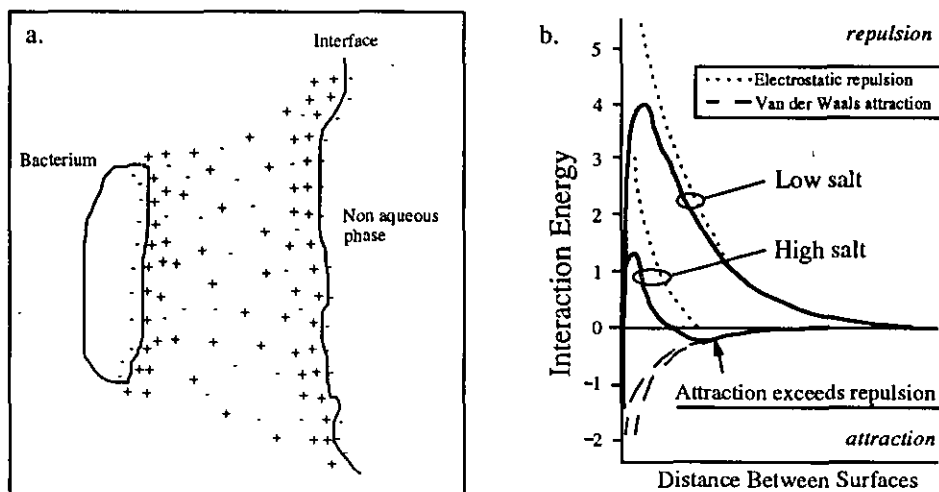
Figure 3.1 *(a) Schematic representation of the interaction between potentially overlapping cation clouds accompanying a negatively-charged bacterium as it approaches a negatively-charged interface. (b) Idealised curves showing the potential energy of interaction as a function of distance between a bacterium and an interface in solutions of different salt concentration (after Marshall 1979).*

density of charge sites available for adsorption. Clay minerals have excellent adsorptive properties, silts, fine sands and coarser sands have successively lower adsorptive capacity (Marshall, 1979). In sandy sediments the actual particle size distribution is important. Unlike clays, which have internal charge surfaces available for adsorption, only the outer surface of a sand grain is available for adsorption. Thus, the smaller the mean particle size the greater the availability of adsorption sites in a given volume of sediment.

The relationship between bacteria and particles does not appear to be well documented in the literature. It could be argued that the relative sizes of particles to organisms may be significant in determining both bacterial transport and survival dynamics. Very small particles, clays and humics for instance, may actually form a coating around larger organisms (Roper and Mitchell, 1978), affording protection and increasing the likelihood of settlement. If evenly coated with similar size particles, it is possible that the coating will not be easily detached from the organism. Where particles are of the same order of size as the organism the protection afforded by the particle may be minimal and the likelihood of detachment higher. For particle sizes increasingly larger than the organism, the particle might develop a coating of organisms and hence the organisms adopt the settlement characteristics of that particle. In this case the protection afforded by the particle may be less and the organisms easily detached in turbulent flows.

These types of attachment will not occur in isolation, they will take place at the same time as a range of other processes such as coagulation and flocculation. The dominant phase of adsorption

will depend on the flow regime, the supply, range and nature of particles and particle sizes. The degree of adsorption also varies between species of bacteria (White, 1979) and the degree of saturation of adsorption sites. Adsorption increases with an increase of pH above 8.0, the addition of divalent cations, i.e. increased ionic strength and, in soils, with decreasing soil moisture (Bitton, 1980).

3.2 Transport of faecal coliforms into surface waters

Bacteria enter stream and river channels from a great many sources. Point source inputs include effluent discharges such as domestic and farm effluents as well as storm-water drains, storm sewage overflows, etc. These inputs are measurable in terms of load and quality and thus their incorporation into water quality models is relatively straightforward. Bacterial inputs from non-point sources are not so easily dealt with due to the multitude of sources, pathways and variables which influence their passage into a channel. This section concentrates on the transport of faecal bacteria from the catchment surface into streams and rivers.

Sub-surface transport

Two of the main transport mechanisms for bacteria within soils are matrix through-flow and non-matrix through-flow. Experiments with disturbed and undisturbed soil columns have shown that macropore transport is not an important pathway for bacterial transport in disturbed soils. In undisturbed soils however, transport via macropores can result in significant contamination many tens of metres from the

source of bacteria (Smith et al., 1985). Knowledge of current land-use may therefore be quite significant in accurately determining bacterial inputs from agricultural land in upland catchments, because changes in land-use will alter the hydrological response of the catchment to rainfall and hence patterns of bacterial transport. During frequently occurring or short duration rainfall events however, the sub-soil transport of bacteria will be dominated by matrix throughflow resulting in no significant transport beyond a few metres (Germann et al., 1987). Macropore flow is more likely to occur when infiltration excess or saturation excess occurs, and bacterial transport is enhanced in the saturated zone (Hagedorn et al., 1981).

Hunter et al., (1992) examined the relative contributions of inputs from overland flow, matrix throughflow, and non-matrix through-flow to stream bacterial numbers in an upland experimental catchment in Yorkshire. Matrix throughflow was found to produce a small stream bacterial loading in comparison with non-matrix throughflow and overland flow. This was due to low concentrations and low flows. It was suggested that, as the bulk of matrix through flow input occurs at or below the stream surface, this flow mechanism could account for a large proportion of the water input to the channel, but could only input relatively small numbers of bacteria due to the capacity of the soil matrix to filter them out. Mechanisms which retain bacteria, such as filtration, are sometimes described as deposition mechanisms, those which dislodge trapped bacteria are referred to as entrainment mechanisms, the latter being of lesser importance when considering bacterial transport (Hornberger et al., 1992; Corapcioglu and Haridas, 1985). The main deposition mechanisms, i.e. restrictions to bacterial transport in the soil matrix, are straining

or filtration in the contact zones of adjacent pores, sedimentation in the pore spaces, and adsorption (Corapcioglu and Haridas, 1985). The presence of organic material can also limit the extent of bacterial transport in soils. Mats composed of bacteria or extracellular polymers form an integral part of septic tank drain fields, acting as fine filters to strain out organisms (Yates and Yates, 1988).

Hunter et al. (1992) observed faecal coliform loads of three and five orders of magnitude higher from non-matrix through-flow and overland flow than matrix through-flow, suggesting that matrix through-flow is not a significant contributor of base flow bacterial inputs to the stream channel. The low frequency of zero values of bacterial concentration at non-matrix through-flow sampling outlets suggested that those sites were end-points of extended macropore systems. Field observations showed that water and entrained organisms from the catchment surface close to the stream reached the non-matrix through-flow sampling sites via root systems and non-biological voids. It was suggested that most of these voids occurred near to the soil surface where plant root density is greatest.

Evans and Owens (1972) examined the response of pasture, underlain by an extensive land drainage system, to an application of piggery effluent (Figure 3.2). The variation in E.coli concentration in the land-drain water in the absence of fresh inputs of faecal material was also examined (Figure 3.3). This was affected by flow-rate, the number of bacteria in the soil or on the vegetation and the rate of application of slurry. The relationship between flow and E.coli concentration accounted for 77% of the observed E.coli variation, but was not valid while large volumes of applied slurry remained on the land surface. Figure 3.2 shows the rapid response of
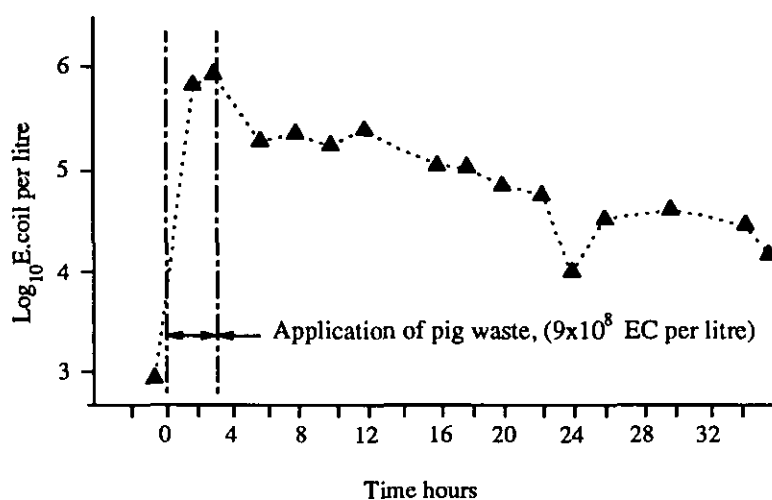

Figure 3.2 Response of _E.coli_ concentration in water draining an extensive land drainage system to an application of piggery waste (after Owens and Evans, 1972)
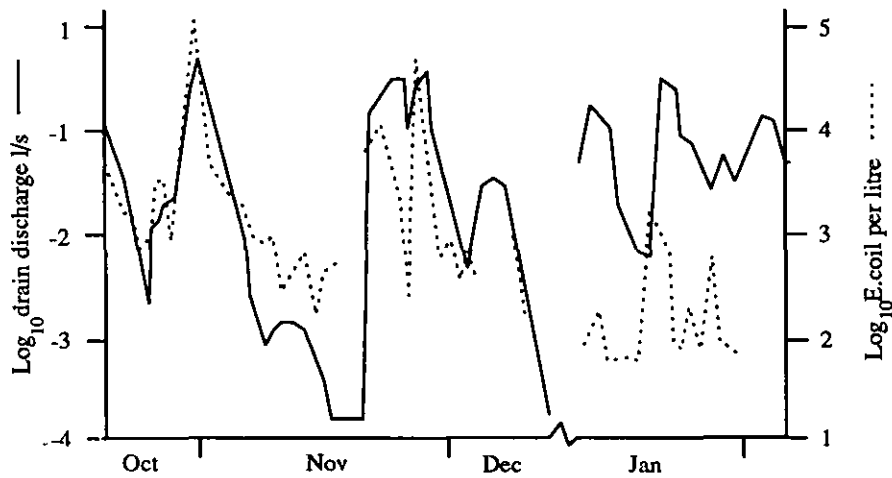
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Figure 3.3 Variation of *E.coli* concentration in water draining an extensive land drainage system in the absence of recent inputs of faecal bacteria (after Igneous and Evans, 1972)

the system to the application of waste. Discharge was monitored continuously throughout this test and found not to be affected by the application. After peaking, the bacterial concentration returned to near background levels within a matter of days.

The results of a later study (Evans and Owens, 1973) suggested that at very low discharges bacterial numbers rise (Figure 3.4) and suggested that this observation might merely be an artefact of the data resulting from too few samples at low flows. However, it may be that the lack of dilution by soil water resulted in a more concentrated leachate. If the latter was the case then the minimum bacterial concentration could have been due to dilution. At higher drain discharges the flow simply has an increased entrainment and washout capacity. This may be due to factors such as the proportion of the productive soil mass contributing bacteria into the flow, i.e. less at lower flows, or pore water velocity, greater
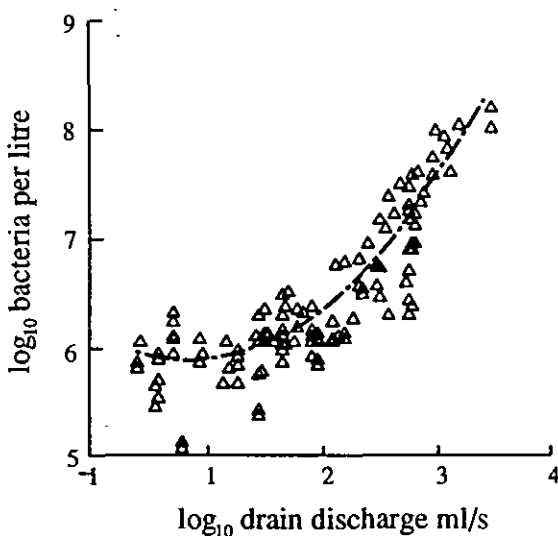
Figure 3.4 Curve showing relationship between *viable bacteria* and land drain discharge (after Owens and Evans, 1973)

detachment of bacteria from soil particles occurring at higher velocities. .

Rainfall run-off

Bacterial counts in receiving waters, resulting from non-point source contamination, will be highly dependant on local hydrological characteristics (Kunkle, 1970). The capacity of a particular rainfall event to transport large numbers of bacteria to a receiving water will depend upon the catchment characteristics, the nature of the rainfall event, the antecedent moisture status of the catchment and the supply of faecal bacteria to the catchment surface. Surface run-off and non-matrix throughflow provide efficient transport pathways during rainfall events and their development will depend upon antecedent moisture status, soil type, vegetation cover, slope and the presence of impermeable surfaces. The duration and intensity of the rainfall event are important to the development of infiltration excess, saturation excess and macropore flow. The local climatic conditions may influence evapotranspiration rates and hence effective rainfall and total transport capacity of a given storm. The spatial extent of the event will also be important in determining total and peak discharges of bacteria.

Hunter and McDonald (1991b) studying a small research plot adjacent to the River Skell in Yorkshire examined the relationship between faecal coliform concentration in overland and stream flow to parameters reflecting the rate and timing of rainfall events (see Table 3.1).

Variables were chosen to reflect the long and short term influences on bacterial loss from the catchment land store. The indicators of recent rainfall were significant in determining both overland and stream flow faecal coliform

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Table 3.1 *Variables used to test the significance of rainfall events on overland and stream flow bacterial dynamics (after Hunter and McDonald, 1991b)*

Variable Name	Description
RFT1	Rainfall in the 4HRS preceding sampling (mm)
RFT2	Rainfall in the 24HRS preceding sampling (mm)
RFT3	Rainfall in the week preceding sampling (mm)
TRF1	Time elapsed since daily rainfall > 1 mm (days)
TRF2	Time elapsed since daily rainfall > 3 mm (days)
TRF3	Time elapsed since daily rainfall > 10 mm (days)
NRF	No. days in preceding 10 when rainfall > 1mm
ST	Stage hgt. at time of sampling
TST	Time since stage hgt. > 0.23 m (base-flow)
RSD	Relative sheep stocking density
TMP	Mean air temperature for the day preceding sampling

concentrations, as was the time since the stream stage height was greater than baseflow. Relative stocking density and the temperature for the previous day also had some significance to stream and overland faecal coliform concentrations, respectively (Table 3.2). Variables that accounted for less than 2% of the observed variation in the data were ignored. The results agreed with examinations of seasonal trends which were found to be flow dependant. In the winter there was sufficient rainfall to maintain lower bacterial levels due to a greater rate of washout from the catchment land store (Hunter and McDonald, 1991b).

Kunkle (1970), studying upland catchments with permeable soils, found that bacterial contributions

Table 3.2 *Results of multiple regression analyses examining the significance of rainfall events to stream and overland flow bacterial dynamics (after Hunter and McDonald, 1991b.*

Variable	Multiple regression results for log10 FC concentration (FC/100 ml) in;	
	Overland flow (Mean FC concentration per 100 ml)	Streamwater
	Specific R^2	Specific R^2
RFT1	0.248	0.174
RFT2	0.155	0.154
TST	0.167	0.229
TMP	0.024	-
RSD	-	0.082
Multiple R^2	0.594	0.639

from areas away from stream margins were small compared to those derived near channel, and grazing had minimal impact when carried out away from the stream margins. Upland areas contributed little or no overland flow during storms, most storm run-off originating on saturated areas which built up along channel edges. It was therefore suggested that due to the run-off processes, bacterial contamination is probably more a function of activities in and around the stream channel than of basin-wide land-use. Similarly, Hunter and McDonald (1991a) found that moist areas, where overland flow was preferentially generated, contributed significantly higher numbers of faecal bacteria to surface waters than dry areas.

Hunter *et al.* (1992) working in the same research plot as Hunter and McDonald (1991b), examined specific relationships between stream faecal coliform concentrations and input loadings from various inflow components; overland flow, matrix through-flow and non-matrix through-flow, at 11 sites along the channel. The faecal coliform load contributed by overland flow was five orders of magnitude higher than matrix through flow and two orders of magnitude higher than non-matrix through-flow. Overland flow was a major contributor to stream bacterial load during both base and storm-flow conditions. Seeps, springs, return flows and protostreams all contributed to overland flow. Protostreams develop during rainfall where subsurface and surface flows combine to produce a defined and recognisable channel. Protostreams were suggested to be very important in the transport of bacteria from the catchment surface to stream-bed store. The bacterial input rate was suggested as being largely determined by rainfall conditions, positive correlation was found between faecal coliform input load and stream stage height from both overland and non-matrix throughflow sites (Figure 3.5). At a quarter of the overland flow sites however, this relationship was reversed as a result of localised depletion of the land store caused by rising flow and increasing bacterial removal. Faecal coliform load at these sites however, still increased with rising flow. It was suggested that areas prone to depletion may have been those subject to continual water movement and hence bacterial removal. Considerable variation was found in the median faecal coliform load values for overland flow at different sites reflecting the hydrological processes influencing the flow to a particular sampling point. Flows were derived, for example, from bacteriologically pure near-channel return flows, or from highly concentrated flow in protostreams. Strong positive relationships were also found between stage height and in-stream faecal coliform concentration.
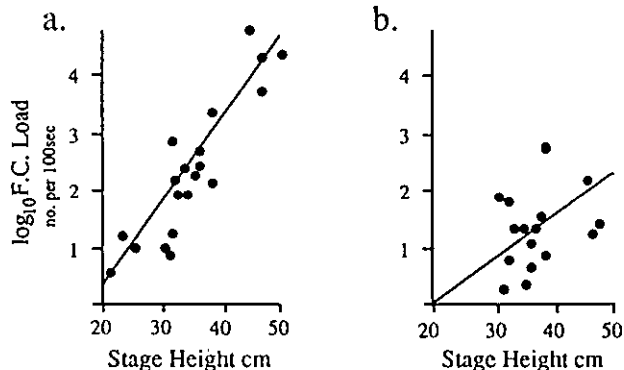
Figure 3.5 *Scatter plots of $\log_{10}$ faecal coliform load against stage height for inputs from (a) overland flow and (b) non-matrix through-flow (after Hunter et al., 1992)*

These studies indicate that faecal coliform inputs in upland areas of catchments are from the channel margins and that transport is concentrated at or near the ground surface. The impact of soil matrix through-flow on stream bacterial dynamics is insignificant and macropore and run-off in protostreams in response to rainfall dominates transport from these areas.

In lowland areas the significance of inputs to the stream channel is likely to change. Denser human population will result in impacts from effluent · treatment works and storage within the channel bed will become more significant. Indeed, Hunter et al. (1992) suggest that the relative importance of inputs from the catchment land store and the channel-bed bacterial store may depend to a great extent on the location at which the stream is sampled (Figure 3.6). At upland locations the flow conditions are such that erosion in the channel predominates, resulting in the likely dominance of land surface over bed store inputs.

3.3 Transport and storage of faecal coliforms in streams and rivers

Enteric bacteria in the aquatic environment exist both freely suspended in the water column and attached to particles and other solid substrates. The transport of bacteria in stream and river channels comprises two main components, the transient movement of bacteria stored in the stream channel bed and the movement of organisms suspended in the flow. The settlement deposition, storage and subsequent re-suspension of organisms is one of the major processes of bacterial transport in river channels. *E. coli* densities in the stream channel bed may be up to 1000 times higher than in the water column (Van Donsel and Geldreich, 1971; Matson et al., 1978; Stephenson and Rychert, 1982). The release of stored organisms from the stream bed may result from disturbances by crossing cattle, a passing boat or a person wading or paddling (Sherer et al., 1988). The main cause of resuspension is increasing flow resulting from storm run-off or artificially induced flow releases (Kunkle and Meiman, 1968; Kay and McDonald, 1980; Jenkins, 1984; Sherer et al., 1988). Large impoundments, such as lakes and reservoirs, act as bacterial sinks, low flow velocities may result in rapid settlement of particle associated organisms, free living faecal coliforms will undergo rapid die-off in the water column. Only major flow events are likely to result in significant transport beyond the water body (Johnson and Ford, 1987).

The settlement of organisms

The settlement process for faecal coliforms is driven by effects which act to increase the settlement potential of the organisms, the attachment to particulates and formation of flocs or clumps of bacteria result in the enhancement of
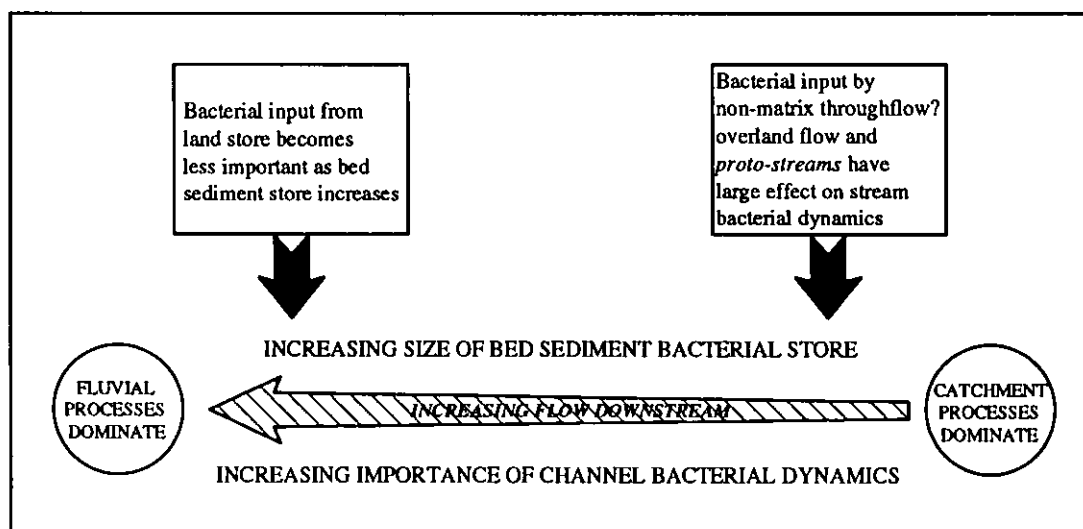
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Figure 3.6 *The relative importance of bacterial input rates from the catchment land store and the channel sediment store to stream bacterial dynamics, with distance downstream (after Hunter et al., 1992)*

settlement rates and hence removal from the water column.

Experiments to assess whether the adsorption of bacteria onto estuarine silts and marine muds takes place in the water column or at the channel bed have shown that 20% of faecal coliforms are adsorbed immediately onto particles in suspension (Milne et al., 1986). Weiss (1951) suggested that effective adsorption was dependant on the availability of sufficient sediment with high adsorptive capacity. Matson et al. (1978) examined the relationship between water and channel-bed bacterial numbers up and downstream of a sewage effluent discharge. Upstream of the discharge point a statistically significant correlation of stream-bed to water column bacterial concentrations was found. This was not the case downstream and it was suggested that upstream the concentrations were in equilibrium, whereas downstream the sediment was saturated with respect to bacteria. More recent research (Milne et al., 1986) has shown however, that the low density particulates associated with sewage effluents remain in suspension at low velocities, hence limiting the ability of such bacteria to transfer from the water column to the stream-bed.

In the presence of muds and silts in estuary water bacterial deposition was found to be a function of time and the deposition rate was directly proportional to SS concentration. With sewage final effluent faecal coliform deposition from the top 30 mm was not a function of time and no significant alteration in concentration occurred after three hours. Similarly, when effluent was mixed with estuary water, deposition of faecal coliform was no longer a function of time and the deposition rate exhibited no correlation with SS. As the bacteria were found to be just as likely to become adsorbed to effluent particles as to estuarine silts and muds, the difference in deposition was attributed to the settling characteristics of the particles. The experiments were carried out with concentrations of between 5×10^4 to 7×10^4 E. coli per 100 ml and it was suggested that at other concentrations different behaviour may be observed. For example indigenous and microorganisms of faecal origin may compete for adsorption sites (Milne et al., 1986).

Jenkins (1984) examined bacterial settlement rates in still water using natural sediment from the River Washburn in Yorkshire. Up to 75% of E. coli settled out within the first few minutes of the experiments. The results suggested that 60% of the bacteria were associated with particles of less than 30 µm diameter, or whose settling velocity was equivalent to that of mineral sediment grains of less than 30 µm diameter. This result was found to be in agreement with the results of Mitchell and Chamberlain (1978). Gannon et al. (1983) and Auer and Niehuas (1993) found the majority of faecal coliforms (approx. 90%) to be associated with particle sizes of between 0.45-10µm. Settling rates in the order of 1.2 m.day⁻¹ for these particles have been demonstrated by sediment trap experiments in Onondaga Lake, New York (Auer and Niehuas, 1993). Such a settlement rate might be applicable to quiescent zones at river margins but turbulence effects would tend to reduce the net settlement rate.

Resuspension from the stream-bed bacterial store

Early studies in the United States (Morrison and Fair, 1966; Kunkle and Meiman, 1968; McSwain and Swank, 1977) demonstrated enhanced coliform concentrations during high or rising flows and possible links with suspended sediment concentration and the stream-bed/water contact area. It was suggested that the supply of organisms was finite, being exhausted by successive flow events (Elder, 1978) and a seasonal pattern of bacterial accumulation in the stream channel was observed, periods of low flow favouring sedimentation and vice versa (Streeter, 1934).

Kay and McDonald (1980), demonstrating inadequacies in the studies of Morrison and Fair (1966), McSwain and Swank (1977) and Kunkle and Meiman (1968) due to infrequent sampling and other factors, were prompted to make a more thorough examination of the sources and behaviour of faecal coliforms in streams during storm events.

An intensive programme of sampling for coliform organisms — both during periods of constant flow, to establish background variations in concentrations and in storm events to determine some genuine pattern in coliform response to flow events — was undertaken in the River Washburn catchment in Yorkshire (Kay and McDonald, 1980). Total coliform concentrations, observed over three 24 hour periods in summer and winter, were found to be highly variable (between 2 and 600 counts per 100 ml). During the rising limb of all storm events significant increases in concentration were observed. In order to determine the source of organisms causing these rises, controlled releases were made between two of the reservoirs in the Washburn system to generate artificial flow events. It was hypothesised that if the increased bacterial numbers during hydrograph events were a result of soil-matrix throughflow, non-matrix throughflow and overland flow, the artificially produced hydrograph would not cause an increase in the bacterial

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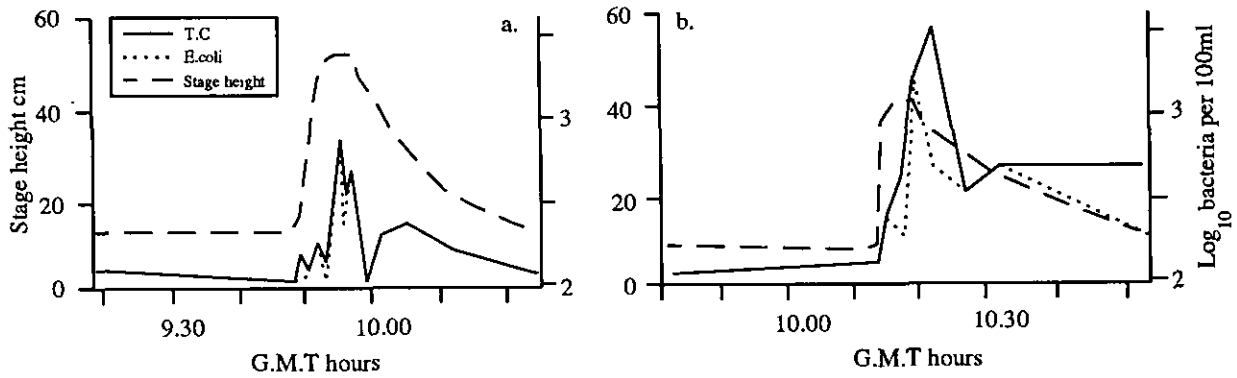
Figure 3.7 *Propagation of artificially generated hydrograph with response of total coliforms (TC) and E.coli concentration at locations (a) 400 m, and (b) 2500 m downstream of the hydrograph source (after McDonald et al., 1982)*

concentrations in the stream and may even reduce levels by dilution, given that the reservoir water was of low bacterial concentration (McDonald et al., 1982). The first experimental release showed this hypothesis to be false, marked bacterial peaks coincided with the peak stage of the hydrograph the magnitude of which was of similar to the increases observed during natural events. Causal mechanisms suggested for the observed responses were; entrainment of bacteria from the channel bed, entrainment of bacteria adhering to sediment on the channel bed, the release of organisms through bank wash and collapse and wash-out of channel pools. The second release was designed to cause bed disturbance with minimal stage rise. A six-fold increase in velocity was achieved following low flow velocities during a cold rain-free period. Again an increase in coliform concentration of similar magnitude to the natural hydrograph response occurred, suggesting a direct bacterial or indirect sediment/bacterial response (Kay and McDonald, 1980). Further releases made during the same

experimental programme indicated that the peak bacterial concentration increases with distance downstream, suggesting continued entrainment and accumulation of organisms from storage within the channel as the flood wave propagates (Figure 3.7). The finite nature of the channel supply of organisms was also observed during a prolonged release to provide water for slalom canoeing. High flow was maintained throughout the event and following the bacterial peak, which coincided with the hydrograph peak, the concentration began to fall and would eventually become exhausted (Figure 3.8).

To confirm the above results a release was made immediately after a major natural flow event that should have flushed the channel bed free of organisms. The release did not cause a bacterial peak and slight dilution was observed. It was noted that inputs from overland flow would have been removed during the passage of the peak and receding limb of the natural hydrograph (McDonald et al., 1982).
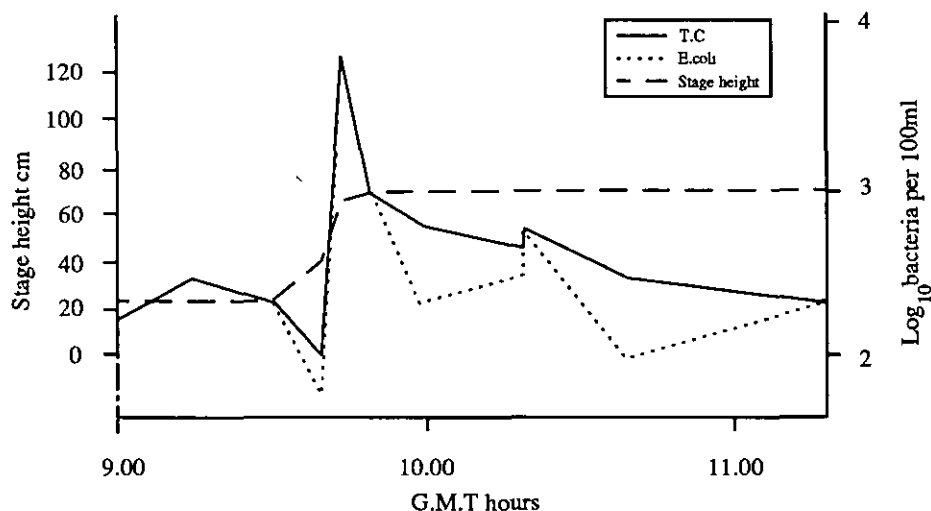
Figure 3.8 *The response of total coliforms (TC) and E.coli concentration to a step change in stage height (after McDonald et al., 1982)*

Hunter et al. (1992) suggest that the relative significance of bacterial inputs from channel bed and the catchment surface may depend on the location at which a stream or river is sampled, inputs from the bed, it is suggested, will generally increase in significance with distance downstream where lower flow velocities predominate and greater settlement occurs.

Settlement and die-off dominates the behaviour of faecal coliforms in lakes and reservoirs, although re-suspension may occur at the margins of the water body. Re-suspension of bacteria by wind-wave action was thought to be responsible for faecal coliform concentrations of up to two orders of magnitude higher at a depth of 1 foot above the bed than at the surface in Lake Houston in the United States (Davis and Valentino, 1985). In Lake Michigan (USA) a survey was carried-out of faecal coliform concentrations around the mouth of the Milwaukee River. Concentrations decreased with distance from the mouth, they also decreased with depth. The numbers of organisms were not even elevated in samples taken where bottom sediments were known to have been disturbed in the process of sampling. This behaviour was thought to be due to a thermocline at the interface of warm river water flowing over cold lake water. No significant difference in levels during dry or wet weather was found, the main inputs of faecal indicators being sewage effluents, urban run-off and storm overflows (Zanoni et al., 1978). The situation in DeGray Lake, Arkansas, was quite different. Storm flows from feeder streams were found to contribute considerable loads of nutrients, bacteria and suspended sediment, capable of travelling the full 32 km length of the reservoir in only eight days. The average residence time of the reservoir is 1.4 years (Johnson and Ford, 1987). These results suggest that although impoundments may generally act as a sink for organisms travelling through a river system, during high flows transport might be sufficiently rapid for large numbers of organisms to travel past the impoundment into the river downstream.

4 Existing models for faecal coliform transport and survival

4.1 Bacterial die-off models

Crane and Moore (1986) undertook a thorough examination of bacterial die-off modelling and the following section summarises some of this work.

Physically based models used to describe bacterial die-off are generally based on simple first order decay dynamics as given by Chick's Law (Equation 4.1 and Section 2.1).

$$\frac{N_t}{N_0} = 10^{-kt} \quad ..... \quad ie. \ N_t = N_0 . 10^{-kt} \tag{4.1}$$

The equation describes logarithmic die-off of a bacterial population (N) over a time period (0 to t), with constant die-off rate (k). Figure 4.1 shows form curves for bacterial decay from models based on first order dynamics. Curve 1 describes simple first order decay as given by Chick's Law (Equation 4.1).

Equations 4.2 to 4.10 extend Chick's Law in an attempt to account for a variety of observed die-off effects. Immediate and constant die-off of the entire population is described by Equation 4.1. Equation 4.2 allows for a period of extended survival and growth until $t > t_i$, the time delay, after which decline commences (Figure 4.1, curve 2).

$$\frac{N_t}{N_0} = 10^{-k(t-t_i)} \tag{4.2}$$

Equation 4.3 is proposed for die-off of a population composed of a number of sub-groups with different tolerances to environmental stresses.

$$\frac{N_t}{N_0} = a.10^{-k_a \cdot t} + b.10^{-k_b t} \tag{4.3}$$

A changing die-off rate may be observed over time as susceptible groups die-off more rapidly leaving longer lived sub-groups such that the overall die-off rate takes the shape of curve 3 in Figure 4.1. Equation 4.3 represents two bacterial sub-groups with a and b the proportions of the total bacterial population having die-off rates k_a and k_b (Streeter, 1934). This equation may be extended to give a complex series explaining a large number of sub-groups with different die-off rates (Equation 4.4).

$$\frac{N_t}{N_0} = a \, 10^{-k_a t} + b \, 10^{-k_b t} + ... \, n \, 10^{-k_n t} \tag{4.4}$$
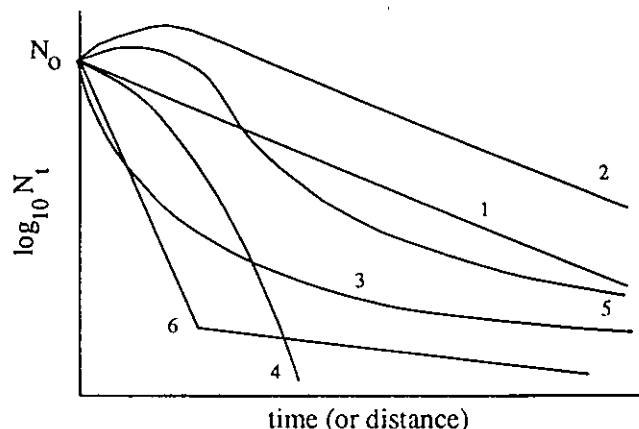
Figure 4.1 Bacterial die-off curves as predicted by models in the literature (after Crane and Moore, 1986)

A number of other models in the literature produce a function similar in shape to curve 3 (Equations 4.5 and 4.6), including the statistical model of Burton et al. (1987).

$$\frac{N_t}{N_0} = \frac{1 - 10^{-k' \cdot t}}{2.3 \, k' \cdot t} \tag{4.5}$$

Where k' is the die-off coefficient,

$$\frac{N_t}{N_0} = (1 + nk_0 t)^{-1/n} \tag{4.6}$$

and in Equation 4.6, k_0 is the initial die-off coefficient and n is the coefficient of retardation, $n>0$ for a type 3 curve, $n<0$ for type 4 (Phelps, 1944; Fair and Geyer, 1954).

$$\frac{N_t}{N_0} = 10^{-kt(-1/n)} \tag{4.7}$$

Similarly in Equation 4.7, a non-uniformity coefficient is used, again with $n>1$ for a type 3 curve, $n<1$ for type 4 (Fair et al., 1971). Curve 4 describes a situation where a continually rising death rate occurs, perhaps in the presence of toxic compounds or as a result of chlorination. Frost and Streeter (1924) used a statistical approach to produce a formula giving the function shown as curve 5 (Figure 4.1) and constants b, c, d and k in Equation 4.8 are empirically derived to fit the data.

$$\frac{N_t}{N_0} = \frac{b}{1 + (ct + d)10^{kt}} \tag{4.8}$$

Burton *et al.* (1987) in a study of bacterial survival in different freshwater sediments developed a statistically based model. In this study the only sediment characteristic for which there was an apparent relationship with die-off was particle size; this relationship, however, was not built into the model. The general model form was

$$y = A e^{-\beta \ln x} \qquad (4.9)$$

from which the following equation was derived;

$$\ln C = a + b[\ln (t + 1)] + \varepsilon \qquad (4.10)$$

lnC is the natural logarithm of the initial bacterial density, *a* is the intercept, *b* is the die-off slope, *t* is time and *ε* is the residual error.

For applications to river water quality these die-off models must be used in conjunction with hydrodynamic and dispersion models and the models need to include a component describing the transfer of organisms to and from storage within the stream channel.

4.2 Water quality models based on the advection-dispersion equation and bacterial die-off functions

A number of water quality models exist that simulate a wide range of pollutants and water quality characteristics, including bacterial concentration. Applications of such models are wide ranging and include lake systems, estuaries and rivers. Such models do not always offer a detailed description of the processes relating to bacterial dynamics and often only include simple time decay models such as those discussed in Section 4.1. These models do, however, incorporate hydrodynamic and dispersion equations in order to describe water flow in the channel and mixing and dispersion processes within the flow.

The advection-dispersion equation (ADE)
The ADE is commonly used in water quality modelling for describing contaminant transport. Two terms, one for advection (the longitudinal movement with the flow) and the other describing dispersion (the effect of mixing on the pollutant concentration). The model can be developed for one, two and three-dimensional transport depending on the nature of the system to be modelled. In rivers, based on the assumption that the flow is well mixed throughout its cross-section, a one-dimensional formulation is used (Equation 4.11).

$$\frac{\partial C}{\partial t} = -U \frac{\partial C}{\partial x} + D \frac{\partial^2 C}{\partial t^2} \qquad (4.11)$$

where,
x = distance downstream
C = concentration
U = mean flow velocity
D = dispersion coefficient.

In this simple formulation it is assumed that there are no inflows or sinks and that the contaminant is conservative in nature, i.e. it is not transformed by physical, chemical or biological reactions. Figure 4.2 is a graphical representation of how the ADE works, showing how the concentration of a pulse of contaminant entering a point x_0 in a river at time t_0 changes through time. Figure 4.2a, shows how the model works if $U=0$, i.e. for dispersion only, as might occur in a still pond. Figure 4.2b shows the variation in C if $D = 0$ i.e. for advection only, an unreal situation, used here to demonstrate the model. Combining the two effects gives Figure 4.2c showing transport downstream and attenuation of the contaminant concentration. The ADE is utilised in the following models and is applied to salinity variation, sediment and bacterial transport. For flood-wave propagation the velocity term in Equation 4.11 uses the kinematic wave velocity instead of mean velocity, the velocity of the flood-wave front is greater than the mean velocity.

Estuarine transport model using the advection-dispersion equation (ADE)
A physically based modelling approach has been applied to the dispersal and inactivation of bacteria in an estuary (Crowther, 1991; Wallis *et al.*, 1989). The model is based on Equations 4.12 to 4.13. The hydrodynamic model uses conservation of mass (or continuity) and conservation of momentum (Equations 4.12 and 4.13, respectively).

$$(W + W_s) \frac{\partial h}{\partial t} + \frac{\partial Q}{\partial x} = \frac{\rho_L}{\rho} L \qquad (4.12)$$

and

$$\frac{\partial Q}{\partial t} + \frac{\partial}{\partial x} \left[\frac{Q^2}{A} \right] + gAS_f + gA \frac{\partial h}{\partial x} + \frac{gAR}{2\rho} \frac{\partial \rho}{\partial x} = 0 \qquad (4.13)$$

where, *Q* is discharge in cumecs, *h* is stage height, *t* is time, *x* is distance downstream, *W* is the width of the estuary at the waters surface, W_s is the average width of storage areas, *L* is lateral inflows per unit length (m².sec⁻¹), *ñ* is density (kg.m⁻³), *A* is cross sectional area of flow (m²), *R* is hydraulic radius and S_f friction slope.

The model assumes that storage areas such as docks and basins do not affect the momentum balance in the estuary and that lateral inflows add no momentum to the flow. The effect on solute concentrations of the docks is also ignored, these
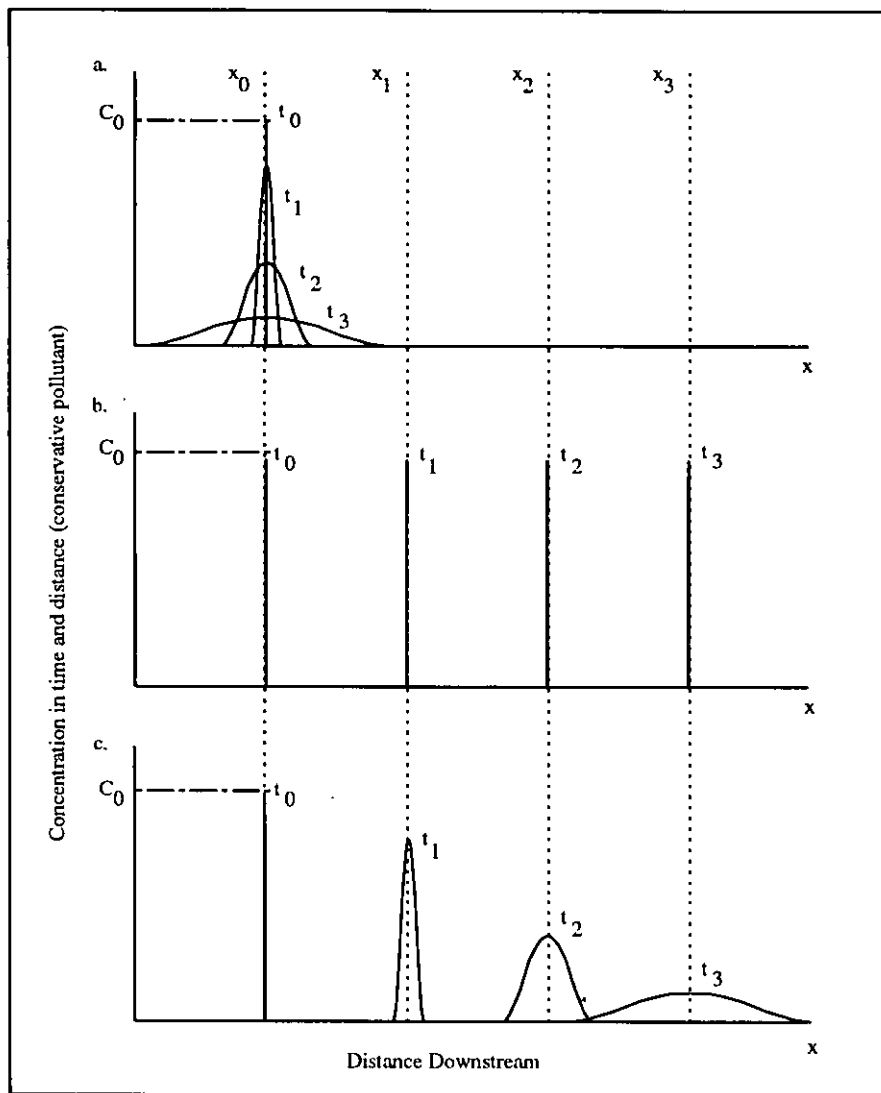
Figure 4.2 *A graphical representation of modelling the dispersal of a conservative contaminant under steady flow conditions using the ADE*

storage areas tending to be poorly mixed being long and narrow with small entrances. The advection-dispersion equation (ADE) is used to model transport of a conservative solute, in this case salinity, and is extended to account for sediment and bacteria (Equations 4.14 to 4.16. A is assumed constant to allow comparison with Equation 4.11).

$$\frac{\partial S}{\partial t} = -\frac{Q}{A}\frac{\partial S}{\partial x} + D\frac{\partial^2 S}{\partial x^2} + \frac{L}{A} \cdot S_L \qquad (4.14)$$

$$\frac{\partial C_s}{\partial t} = -\frac{Q}{A}\frac{\partial C_s}{\partial x} + D\frac{\partial^2 C_s}{\partial x^2} + \frac{L}{A} \cdot C_{sL} + \frac{E-D_p}{A} \cdot C_s \qquad (4.15)$$

$$\frac{\partial C_b}{\partial t} = -\frac{Q}{A}\frac{\partial C_b}{\partial x} + D\frac{\partial^2 C_b}{\partial x^2} + \frac{L}{A} \cdot C_{bL} + K.C_b \qquad (4.16)$$

where, S is salinity, C_s, C_b are the concentration of sediment and bacteria, D is the dispersion coefficient, E is the amount of sediment eroded from the channel bed, D_p is the deposition rate,

subscripts s and L refer to static storage areas and lateral inflows and K is the bacterial die-off rate (s^{-1}).

The influences on bacterial survival are lumped into one value, K, derived from experimental results as a function of local salinity and suspended solids concentration. The bacterial concentration is modelled as a function of flow with first order decay, the influence of sediment dynamics on the bacterial concentration is assumed to be unimportant (Equations 4.15 and 4.16). The reason for this apparent omission is not given, but it may, however, be due to the nature of the supply of faecal coliform organisms. Milne *et al.* (1986) found that there was no significant settlement of organisms from mixtures of sewage effluent and seawater.

Sediment is modelled using the Krone formula, relating entrainment, E and deposition, D_p to channel bed shear velocity (u_*) and critical

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velocities of deposition and entrainment, v_d and v_e. Equations 4.17 and 4.18 show the functions used (Crowther, 1991).

For deposition $u_* < v_d$,

$$D_p = (1 - \frac{u_*^2}{v_d^2})\ W_b.v_s \ \dots \ \text{for } u_* < v_d \qquad (4.17)$$

and for erosion $u_* > v_e$

$$E = (\frac{u_*^2}{v_e^2} - 1)\ W_b\chi \ \dots \ \text{for } u_* > v_e \qquad (4.18)$$

Where, χ is the erosion rate, W_b the bed width and v_s the settling velocity. Shear velocity, u_*, is given by

$$u_* = |\frac{Q}{A}| \cdot \frac{k}{\log_e\{30.2\frac{H}{k_s}\}} \qquad (4.19)$$

where k_s is the Nikuradse sand roughness coefficient, H is depth, and k, the von Karman constant for flow with sediment, is 0.174.

The hysteresis observed by Hjulstrom (1935), in a series of experiments examining the entrainment and settlement characteristics of individual sediment grain sizes, is accounted for by setting $v_e > v_d$ since the energy required to entrain particles is greater than that necessary to maintain their suspension. Therefore for $v_d < u_* < v_e$ the sediment will be maintained in transport, i.e. there is no net deposition or entrainment.

This method of assuming either entrainment or deposition is a different approach to that of Jenkins (1984), which assumes that deposition occurs continuously and the occurrence of either net entrainment or deposition depends upon the relative rates of erosion and deposition.

Application of Equations 4.17 and 4.18 to bacterial transport might be achieved by adjusting the threshold entrainment and deposition velocities, as appropriate.

River Transport Model (ADE)

The ADE has been frequently employed in water quality modelling, especially on large lowland rivers, for example the Tigris River in Iraq (Al-Layla and Al-Rizzo, 1989). This application of the model uses a one-dimensional version of the ADE to simulate a steady-state decline in total coliform count along a 75 km stretch downstream of the "Saddam" Dam. Microbial self-purification of the river is described by a simple first order decay coefficient similar to that of Crowther (1991) (Equation 4.16). Figure 4.3 shows the profile of measured and simulated coliform concentrations with distance downstream of the dam.
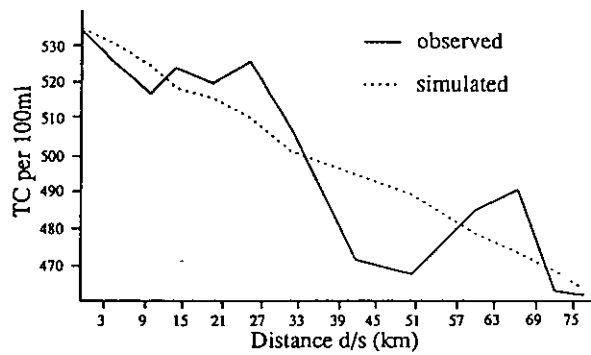
Figure 4.3 Profile of simulated and measured coliform bacteria concentrations for the Tigris river in Iraq

Whilst the general trend of coliform decay is effectively modelled, there is some inadequacy in either the selection of die-off rate or the identification of inputs of faecal contamination that may account for the under and over estimations observed in Figure 4.3.

4.3 IH QUASAR

The model QUASAR (Quality Simulation Along Rivers) was developed to assess the environmental impact of pollutants on river water quality (Whitehead et al., 1979). The model has been applied to range of UK rivers such as the River Tawe to assess heavy metal pollution, and the River Thames, to assess the movement and distribution of nitrates and algae (Whitehead and Williams, 1982; Whitehead and Hornberger, 1984). E.coli, nitrate, dissolved oxygen (DO), biochemical oxygen demand (BOD), ammonia, ammonium ion, temperature, ortho-phosphate, pH, and "conservative" water quality parameter concentrations can currently be modelled. The model performs a mass balance of flow and water quality sequentially down a river system. To model these parameters the river is divided up into reaches. The reach boundaries are determined by points in the river where there is a change in the water quality or flow due to the confluence with a tributary, the location of a sewage treatment final effluent discharge, an abstraction, or the location of weirs. Biological and chemical processes are incorporated for each determinand as required. The flow models are derived from the Muskingum-Cunge flow routing method and give a good approximation to the advection-dispersion equation (ADE) (Equation 4.20). For a full description of the application of this technique see Whitehead et al. (1979).

$$\frac{\partial Q}{\partial t} = -u\frac{\partial Q}{\partial x} + D\ \frac{\partial^2 Q}{\partial x^2} \qquad (4.20)$$

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Where, Q = discharge
        u = kinematic wave velocity
        D = dispersion coefficient

The flow equations provide discharge data for the contaminant mixing models. The treatment of E.coli is simplified with a first order temperature dependant decay rate (Equation 4.21). The organisms are assumed to undergo die-off and mixing within the channel only. The effects of sedimentation and resuspension are not modelled and the effect of sunlight on die-off ignored. The mass-balance equations are based on Continuously Stirred Tank Reactor (CSTR) theory taken from chemical engineering. This approach is similar to Aggregated Dead Zone (ADZ) or Active Mixing Zone (AMZ) dispersion modelling (Young and Wallis, 1986; Wallis et al., 1989; Wallis, 1993). Henderson-Sellers et al. (1988) discuss the relative merits of the ADE and ADZ approaches to dispersion modelling. The ADZ approach is felt to offer a better conceptuallisation of mixing processes in natural channels, gives a better fit to time series of tracer concentrations derived from field experiments and does not suffer from the numerical difficulties encountered when approximating the ADE in computer applications. Equation 4.22 shows the QUASAR formulation for E.coli at the time of writing.

$$\frac{dC_i}{dt} = \frac{Q_i}{V} \cdot U_{t-\tau} - \frac{Q_i}{V} \cdot C_i + I_i - kC_i \qquad (4.22)$$

Where, Q_t = discharge at time t
        V = reach volume
        C_t = bacterial concentration at time t
        $U_{t-\tau}$ = is the upstream input to the reach allowing for travel time τ.
        I_t = sources and sinks due to effluents, abstractions etc.
        k = temperature dependant bacterial die-off coefficient (days^{-1})

This form of the model is only capable of describing the distance decay observed downstream of a point-source input discharging large numbers of bacteria. In Section 5 this Equation 4.22 is modified by the addition of terms for the transfer of faecal coliforms to and from storage within the channel (Equation 5.8).

4.4   Process based modelling of bacterial dynamics in upland streams

In a model developed to predict bacterial numbers in upland streams (Jenkins et al., 1984), water and sediment bacterial concentrations are represented by mass balance equations such that:

$$C = I + W - S - D \qquad (4.23)$$
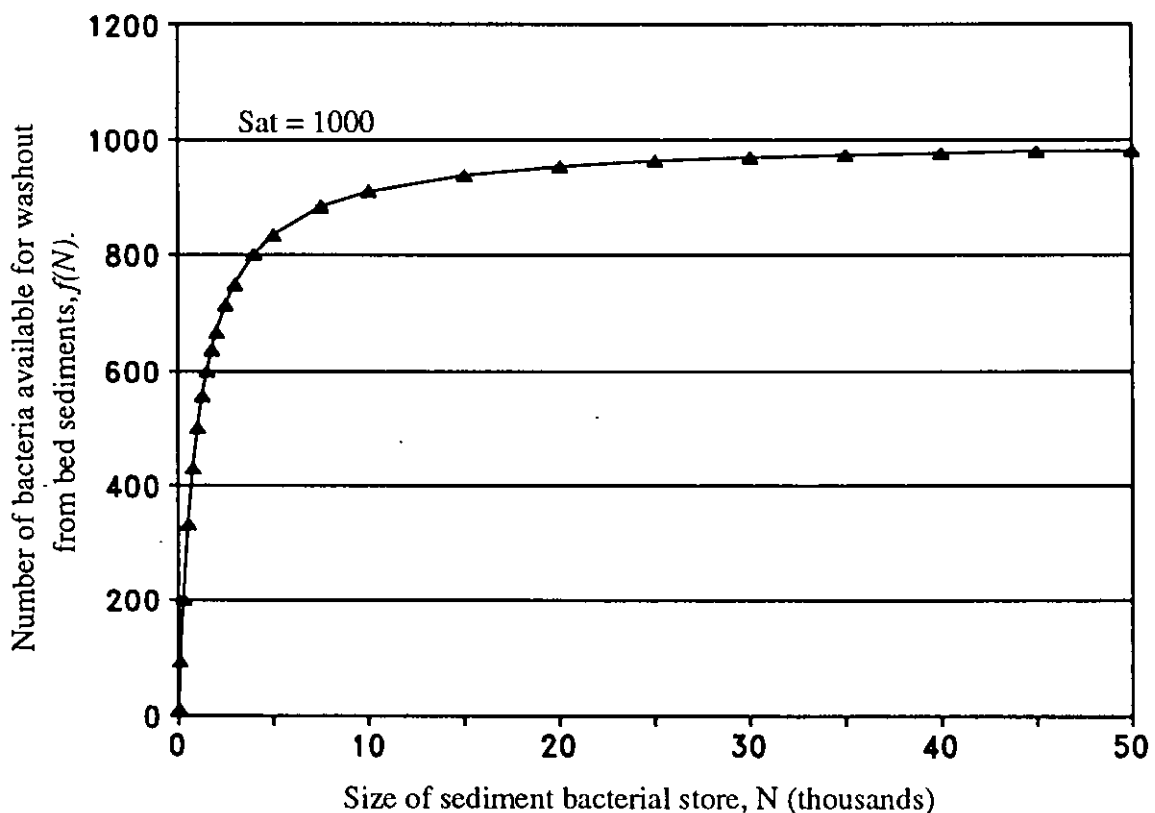
$$N = S - W - D \qquad (4.24)$$

Figure 4.4 Profile of simulated and measured coliform bacteria concentrations for the Tigris river in Iraq (after, Al-Layla and Al-Rizzo, 1989)

where C is the concentration of $E.coli$ in the flow, N is the number of $E.coli$ in channel bed sediments, I is input from upstream and the land-surface, W is washout from bed sediments, S is loss to sediments (settlement). D is net die-off as a result of environmental stresses.

Sediment behaviour is determined by the flow characteristics of the stream i.e., turbulence and velocity as related to discharge. The onset of washout is assumed to occur at some threshold discharge below which the washout term is zero. It was recognised that this may be a simplification of a more complicated fluvial process associated with sediment release following cobble movement and may also be sensitive to the adherence characteristics of the bacteria to the bed and/or sediment. Non-point source inputs are perceived to be associated with "quick" and "base" flow run-off components. The base-flow component comprises soil through-flow and groundwater flow, responsible for maintaining inputs during dry periods. The quickflow component relating to rainfall events over the catchment, resulting in surface run-off and non-matrix through-flow. It is assumed that rainfall-induced processes increase in intensity through the storm and that inputs to the channel increase linearly with discharge. Background and discharge related input are assumed to be diluted by the volume of flow. The input of bacteria to the stream from the surrounding catchment (I) is, therefore, given by:

$$ I = \frac{I_B}{Q} + \frac{I_Q Q}{Q} = \frac{I_B}{Q} + I_Q \qquad (4.25) $$

where I_B is background input, I_Q is discharge related input and Q is discharge.

Bacterial inputs from the catchment surface are assumed to enter the water store only, in the first instance. Transfer between the water and bed sediment bacterial stores is modelled as an internal function of the reach. Increased inputs raise the sediment store bacterial numbers indirectly by providing larger numbers of bacteria for sedimentation. It is suggested that this sedimentation effect is unlikely to remain linear as discharge increases due to the effect of turbulence, which is assumed to reduce the rate of settlement. The model uses a threshold discharge value at which bacterial settling is halved (Equation 4.26)

$$ S = P_{set} \frac{Q}{1 + Q/P_Q} \, C = p.C \qquad (4.26) $$

where P_{set} is the rate of settlement (proportion of total load settling per unit time), P_Q is the discharge at which P_{set} halves.

The model accounts for changes in stream hydraulic characteristics with discharge, which will affect the rate of change of numbers of bacteria per unit stream bed area i.e. as discharge rises the increased hydraulic radius allows bacteria to settle over a larger area (Equation 4.27)

$$ \frac{dN}{dt} = \frac{P_{set} \dfrac{Q}{1 + Q/P_Q} \, C}{v.w} = \frac{S}{v.w} \qquad (4.27) $$

where N is the number of bacteria in the bed sediment store per unit bed area, v is velocity and w is the channel width.

It is assumed that $v.w = Q/h$, where h is the mean flow depth and that $h \propto Q^F$, where $F \approx 0.5$, therefore:

$$ v.w = \frac{Q}{Q^{0.5}} \qquad (4.28) $$

Substituting Equation 4.27 into 4.28 and rearranging gives

$$ \frac{dN}{dt} = P_{set} \frac{C}{1 + Q/P_Q} \, Q^{0.5} \qquad (4.29) $$

This modification causes a slight decrease in the number of bacteria settling into the sediment store once $Q > P_Q$. The entrainment of organisms at some threshold discharge Q_T is given by

$$ W = a(Q^2 - Q_T^2)N \qquad (4.30) $$

The constant, a, represents the total bacterial wash-out for $Q > Q_T$ and is expressed as

$$ a = \frac{T_{H2}}{2.Q_T + 1} \qquad (4.31) $$

where, T_{H2} represents the proportion of the bacterial store washed out at a discharge of $Q_T + 1 \, m^3.s^{-1}$.

Equation 4.30 is further modified to account for the fact that washout was found to be a function of the size of the bed sediment store, i.e. it is assumed that the entrainment capacity of the flow limits the amount of bacteria washed-out of the bed store. In Equation 4.30, N is replaced by:

$$ f(N) = \frac{N. Sat}{N + Sat} \qquad (4.32) $$

where Sat is the maximum value of $f(N)$, which only approaches Sat for values of N much greater than Sat (Figure 4.4). This term effectively limits the number of organisms available for washout at any time-step.

The term for bacterial die-off D in Equations
4.23 and 4.24, is based on the assumption that
the net die-off follows simple first order decay
dynamics

$$C_t = C_0 \, e^{-kt} \qquad (4.33)$$

where C_0 and C_t are concentrations at time $t=0$
and t and k is the die-off rate coefficient.

The differential equations for the model are as
follows:

$$\frac{dC}{dt} = -K_c.C - p.C + \frac{a(Q^2 - Q_t^2)N}{Q} + \frac{I_R}{Q} + I_Q \qquad (4.34)$$

$$\frac{dN}{dt} = -K_n.N + p.Q.C - a(Q^2 - Q_t^2)N \qquad (4.35)$$

where K_c and K_n are bacterial die-off coefficients
in water and sediment, p is the settlement
coefficient.

In validation runs the model was found to predict
both timing and magnitude of bacterial peaks
adequately, under a range of conditions. It was
suggested, however, that further calibration and
parameterisation were needed to produce the
observed coincidence of bacterial peaks with
respect to the hydrograph peaks (Jenkins et al.,
1984).

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5 Towards a new model of faecal coliform stream dynamics

Faecal coliform concentrations are commonly modelled using water quality models based on the Advection Dispersion Equation (ADE) (see Henderson-Sellers *et al.*, 1988) with additional terms for die-off or disappearance using simple first order decay coefficients to describe the net reduction in concentration with time and/or distance travelled downstream (Wallis *et al.*, 1989; Al-Layla and Al-Rizzo, 1989; White and Dracup, 1977). Extensions to these simple first order models include the development of a range of characteristic die-off curves and the incorporation of terms to relate the effects of temperature and insolation on coliform die-off (Auer and Niehaus, 1993; Crane and Moore, 1986; Canale *et al.*, 1973). *E.coli* has been modelled in this manner in QUASAR, a river network water quality and flow model developed by the Institute of Hydrology. Alternative, statistically-based approaches use multivariate analysis to develop models of coliform concentrations related to a number of physical and/or chemical influences (Mahloch, 1974) or, for example, to variables describing the timing, frequency and duration of rainfall or flow events (Kay and McDonald, 1983). In-channel storage of faecal coliforms, however, has been largely ignored in modelling applications, despite the fact that its significance to water quality has been recognised for some time (Van Donsel and Geldreich, 1971; Matson *et al.*, 1978). The first model to incorporate terms for the transfer of organisms to and from storage within the channel was developed by Jenkins (1984). This section describes controlled field experiments designed to extend the work of Jenkins (1984), leading to the development of a new conceptualisation of faecal coliform dynamics in stream channels.

5.1 Field studies for the examination of in-channel storage processes

Experiments were carried out at three sites where artificial changes in flow could be made in order to determine the channel faecal coliform response without inputs from the adjacent catchment. The data derived were used to formulate the structure and parameterise the new model.

The study sites
These were in mid-Wales on the Afon Rheidol, which flows through Aberystwyth into the Irish Sea, the Afon Clywedog in the headwaters of the

River Severn and the River Washburn, a tributary of the River Wharfe north of Leeds.

In the Rheidol catchment controlled flow releases were provided by a hydroelectric scheme. The river was sampled 10 km downstream of the Cwm Rheidol reservoir in the catchment flood plain. The reach is characterised by a partially confined irregularly meandering pool riffle sequence with a bed slope of approximately 1:660. The soil is clay loam. Faecal inputs are derived from the grazing of sheep and cattle and there is also a small sewage treatment works.

The Afon Clywedog in the Upper Severn was sampled at either end of a 4 km reach immediately downstream of the Llyn Clywedog reservoir. The reach is topographically confined and initially of step-pool configuration, rapidly changing to a pool riffle sequence of bed-slope 1:100. The immediate banks and valley slopes are grazed by sheep, domestic inputs are thought to be minimal. The soil is clay loam.

On the River Washburn releases are made between a series of four impoundments. Sampling for faecal coliforms was carried out during a release, made for white-water canoeists, between Thruscross and Fewston Reservoirs. The channel in this reach is a pool riffle sequence with a slope of around 1:100 confined within a narrow valley. The channel banks are stabilised by trees, boulders and, in places, by the underlying bedrock. The valley floor is covered by clay loam soil and is grazed by sheep and cattle. Either end of a 1.5 km reach was sampled.

Sampling and analysis techniques
Sampling commenced prior to the releases to establish initial concentrations. Release waters were also sampled. Stage, temperature and conductivity were recorded at each sampling interval, samples being taken from as near to the centre of flow as possible at approximately 0.6 of the flow depth. Samples of 400 ml were collected in pre-sterilized containers and stored in the dark prior to transportation to the laboratory for analysis. Duplicate samples were taken at intervals of 10 samples for quality control purposes. From each sample six enumerations were made using the membrane filtration technique (H.M.S.O, 1983). Three lots of 100 ml and 1 ml were filtered and enumerated within six

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Site	Sample point	Full data	Rise	Recession
Rheidol (17.2.93)		80.41	127.1	63.7
Washburn (26.5.93)	upstream	126.5	-	-
	downstream	211.7	-	-
Clywedog (28.5.93)	upstream	34.8	68.6	13.5
	downstream	407.9	892.23	140.3

hours of collection, with the results expressed as colony-forming units (cfu) per 100 ml. By making replicate enumerations the confidence interval about an estimate is reduced by a factor related to the square root of the number of replicates (Fleisher and McFadden, 1980; Fleisher et al., 1993). In this case triplicate enumeration results in approximately a 1.73 times improvement in accuracy. The use of two filtering volumes ensures ease of counting both high and low numbers of organisms.

Experimental releases

The first release carried out on the Afon Rheidol on 17.2.93, sampling at one site only, was an artificial hydrograph based on data from the adjacent Ystwyth catchment. The hydrograph shape was approximated by a series of steps which in later experiments were exaggerated as they were found to cause a marked faecal coliform response. A step change in flow was made on the

Washburn and a further artificial hydrograph on the Clywedog.

Initial observations

All release waters were of low bacterial concentration and dry antecedent conditions meant that catchment derived inputs were minimal. The observed responses are therefore assumed to result from inputs to the flow from channel storage. Figures 5.1 and 5.2 and Table 5.1 demonstrate the accumulation of entrained organisms in the flow, with distance downstream during sampling on the Clywedog and Washburn. The Washburn results demonstrate the finite nature of the channel-bed supply of organisms (Figure 5.2).

Responses to stepped hydrographs

The faecal coliform responses to releases on Afon Clywedog and Rheidol (Figures 5.1 and 5.3) exhibit three main phases; low concentrations
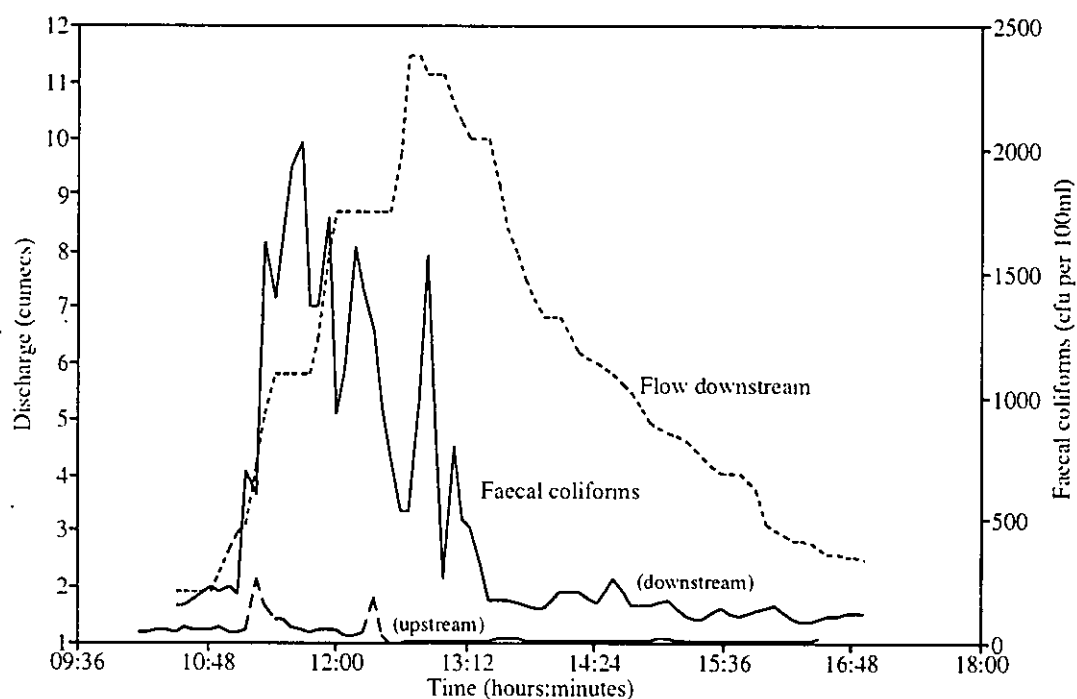
Figure 5.1 Raw faecal coliform and flow data for the experimental flow release in a 4km reach of the Afon Clywedog downstream of the Clywedog reservoir (28.5.93)
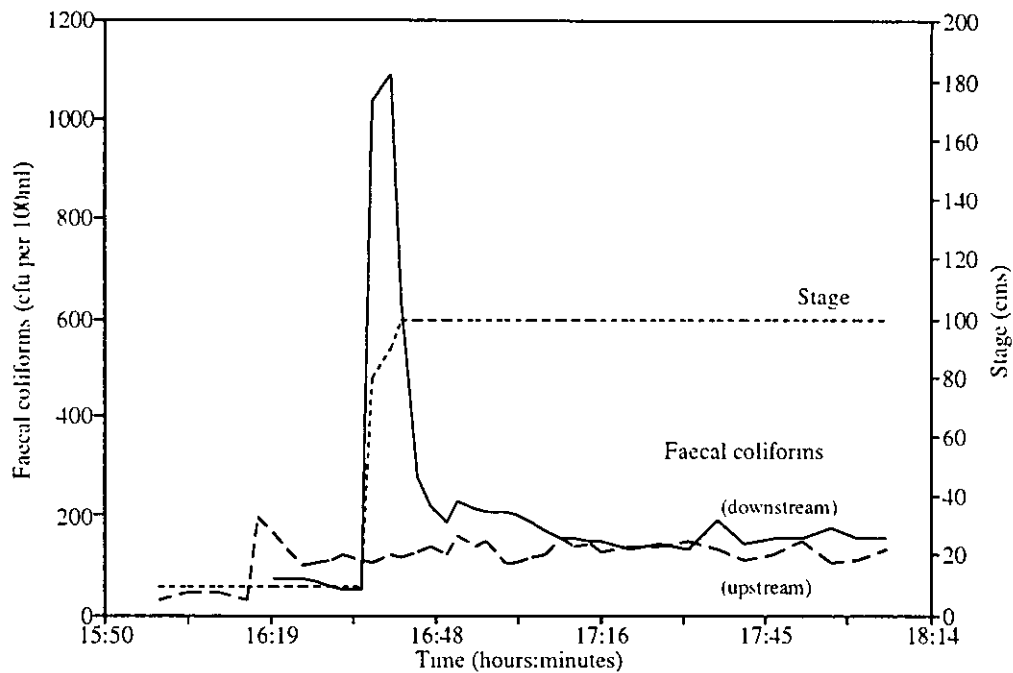
Figure 5.2 *Raw faecal coliform and flow data for the experimental flow release in a 1.5 km reach of the River Washburn downstream of the Thruscross reservoir (26.5.93)*
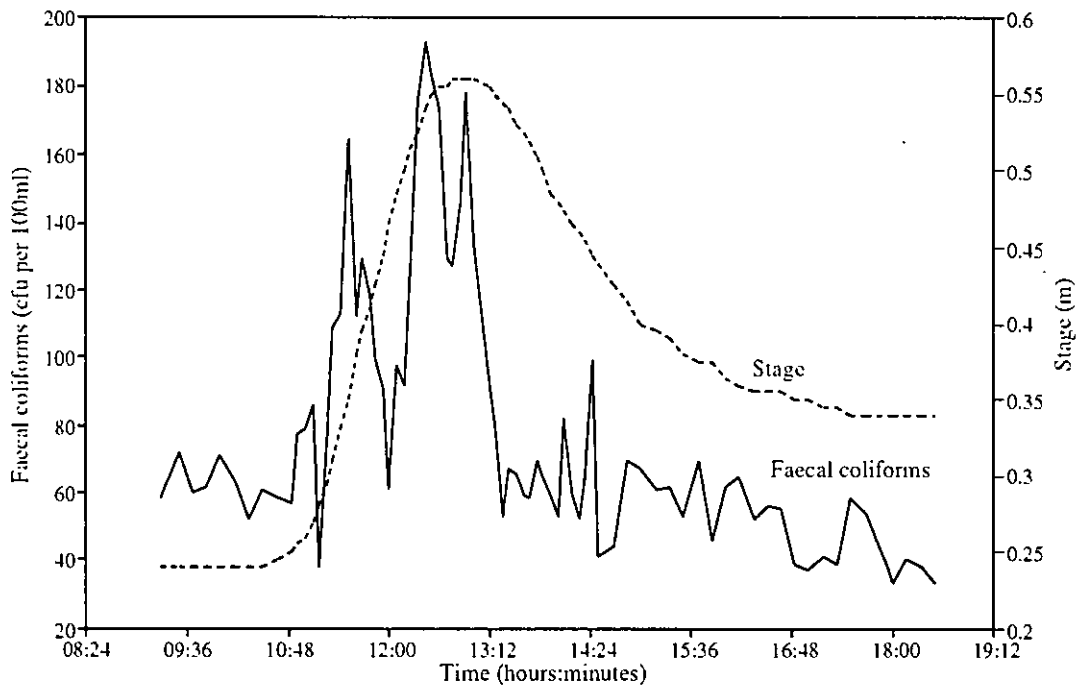
Figure 5.3 *Raw faecal coliform and flow data for the experimental flow release on the Afon Rheidol at a site sampled 10 km downstream of the Cwm Rheidol Dam (17.2.93)*

preceding the release, enhanced concentration coinciding with the rising limb of the hydrograph and a return to background concentrations following the peak flow (Table 5.1). Examination of the Clywedog data suggests that each step change in flow causes a faecal coliform response. In the Washburn (Figure 5.2) the steep rise in faecal coliform concentration corresponds to the step change in flow. The peak concentration is followed by an exponential decline demonstrating the rapid entrainment of organisms at the highly

turbulent wavefront washing the available organisms from bed storage with its passage downstream. The supply of organisms is now diminished and the faecal coliform concentration returns to that of the reservoir water entering the channel.

The on-set of the hydrograph recession, at both the Rheidol and Clywedog sites, resulted in the immediate reduction in bacterial concentration to background concentrations. The organisms

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available from storage are assumed to have been entrained into the flow and transported beyond the study reach, further pockets of organisms stored in the channel only becoming available for entrainment at higher flows.

The results suggest that organisms are entrained from storage within the stream channel over a range of rising flows, each quasi-steady flow representing a threshold for entrainment at the next rise in flow. The observed bacterial response is likely to be affected by the distribution of organisms within the channel and the interaction of the flow with such supply areas.

Investigation of the behaviour of $E.coli$ within the bed of the River Washburn has shown that the organisms are heterogeneously distributed. Weed covered sites in slow flowing water were found to accumulate organisms during low flows and become depleted at higher flows. In fast flowing water this behaviour was reversed, areas of weed cover captured organisms at higher discharges. This behaviour was repeated at bare bed sites but to a lesser extent. In areas of sediment accumulation the numbers of organisms increased during low flows and $vice\ versa$ (Jenkins, 1984).

Large scale dead-zone features may represent significant storage areas for faecal coliforms. The low flow velocities and large residence times of such features are ideally suited to the accumulation of organisms which may settle and remain trapped until some flow disturbance releases them. Reynolds $et\ al.$ (1991) studied a dead-zone in the River Severn which was found to have a residence time in the order of 25 days.

If large scale dead-zones are assumed to be a significant source area for faecal coliforms their non-uniform occurrence along a reach and successive washout might result in irregular bacterial peaks such as those observed during the artificial flow event on the Afon Rheidol (Figure 5.3). Certainly, the wash-out of a large pocket of storage in the vicinity of a sampling site is likely to result in the observation of a bacterial peak.

5.2 Model development; the formulation of in-channel storage equations

In the model described below the heterogeneities in channel storage are ignored, assuming that the effects of dispersion within the flow smooths out the spikes produced by entrainment from irregularly distributed storage zones. Entrainment is modelled purely as a function of flow. This assumption appears to be valid for the Clywedog and Washburn data, but less so in the Rheidol

which is more meandering and may have more large dead-zone stones.

The channel-bed store equations

The formulation of these equations is based on the assumption that the entrainment of organisms is governed by changes in flow with respect to the channel bed area. Areas of bed storage are assumed to occur randomly over the channel bed and to undergo entrainment over a wide range of flows. Jenkins (1984) demonstrated that $E.coli$ in the bed of the Washburn were concentrated at the sediment/water interface. It is, therefore, assumed that as discharge increases, bed areas where the flow was previously insufficient to cause entrainment undergo rapid scour from a thin surface layer of particles with uniform entrainment characteristics. In the new model, entrainment from the many areas of storage is lumped to give a single value at a reach outlet. The total number of organisms are assumed initially to be evenly distributed throughout the channel bed and entrained sequentially as discrete numbers of organisms on rising flows. A proportion of the storage available is assumed to be depleted of organisms at any flow. As flow rises, from any preceding value, more organisms become 'available' and are rapidly entrained into the flow leaving these areas depleted until sufficient recharge at lower flows has occurred. This is achieved in the model by partitioning the total number of organisms, NT_t, in the bed-store into j sub-units resulting in N_{tj} organisms in each store

$$N_{tj} = \frac{NT_t}{j} \quad and \quad \Delta Q = \frac{Q_m - Q_0}{j} \quad (5.1)$$

The range of observed discharges, Q_0 to Q_m, is divided by the number of sub-stores, giving the "bed access flow interval", ΔQ. The number of sub-stores either undergoing washout (n_e) or deposition (n_d) is then determined, thus:

$$n_e = \frac{Q_t - Q_0}{\Delta Q} \quad and \quad n_d = \frac{Q_m - Q_t}{\Delta Q} \quad (5.2)$$

where Q_t is the discharge at time t. At constant flow n_e bed areas are assumed to be depleted of organisms, hence further entrainment can only occur if the flow rises.

Organisms can be re-worked within the channel, being entrained from one sub-store and deposited into another. The net change in total bed-storage in numbers of organisms is given by

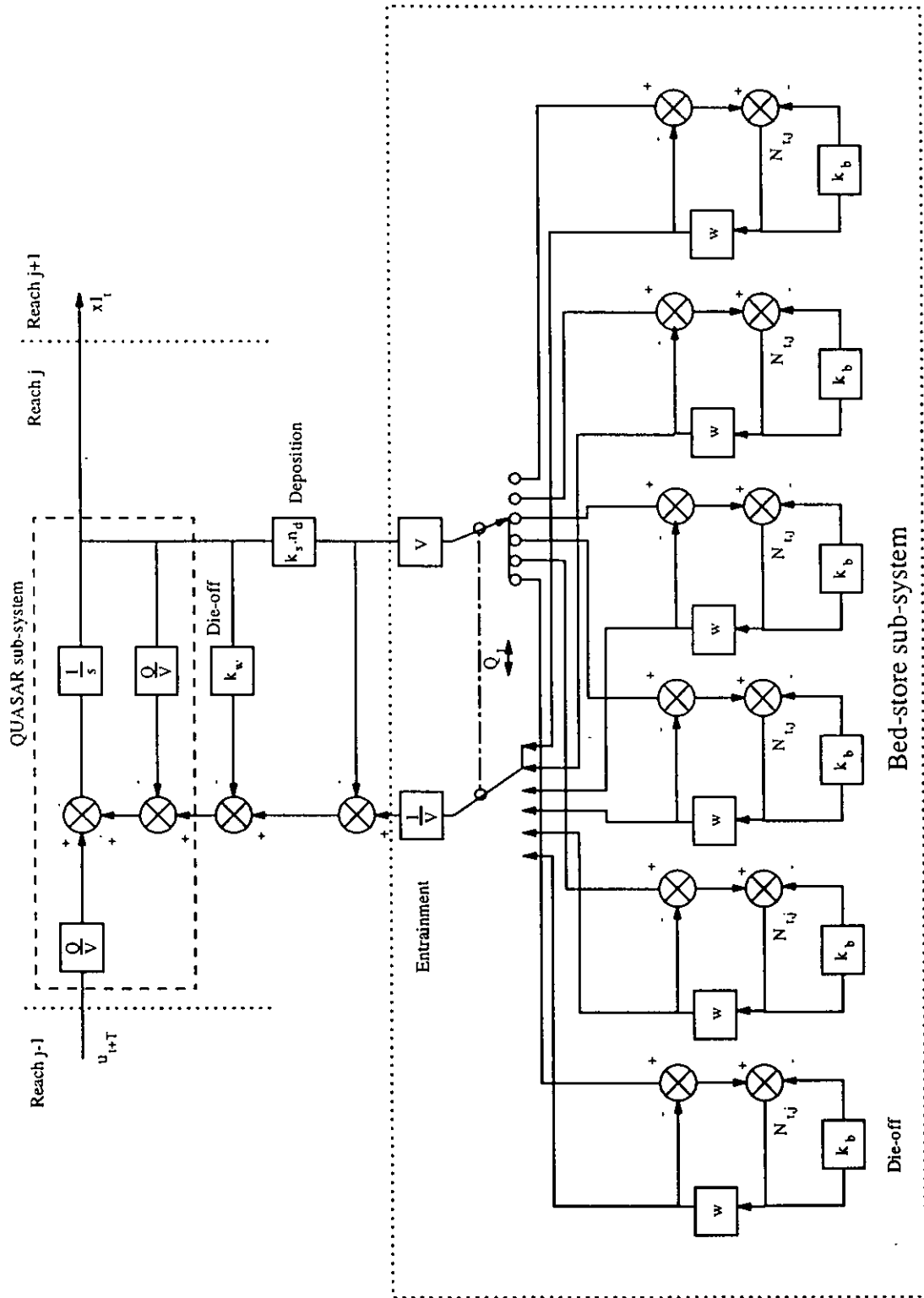$$\frac{dNT_t}{dt} = V.k_s.n_d.x_t - w \sum_{j=0}^{n_e} N_{tj} \quad (5.3)$$

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Figure 5.4 *Continuous time systems block diagram showing the internal structure of the faecal coliform model (for convenience, only six channel sub-stores are shown)*

where x_i is the water faecal coliform concentration, V is the volume of water in the reach and k_s is the settlement rate to each sub-store. Entrainment occurs from n_e channel-bed partitions, the change in storage in each partition is

$$\frac{dN_{ij}}{dt} = -(w + k_b).N_{ij} \qquad (5.4)$$

where, w is the entrainment or washout rate and k_b is the net bacterial die-off rate.

Addition by settlement occurs into n_d individual bed partitions at a constant rate. The change in storage in each sub-store is

$$\frac{dN_{ij}}{dt} = V.k_s.x_i - k_b.N_{ij} \qquad (5.5)$$

Bacterial die-off within the bed is assumed to occur at a constant rate. This is an over-simplification of a complex process controlled largely by nutrient supply and the antagonistic behaviour of the indigenous biota (Verstraete and Voets, 1972). A simple self-regulating population dependent die-off function might be more appropriate, whereby an increase in the number of settled organisms causes the die-off rate to increase. The significance of die-off will be more important when considering model stability for longer periods of data.

Bacterial dynamics in the water column

In the water column at a point in space and time the faecal coliform concentration is assumed to be a mass balance of deposition, die-off, dispersion, transport out of the reach and upstream inflows, lateral inflows and entrainment from the channel bed.

In QUASAR dead-zone mixing is assumed to dominate dispersion within a reach. The term dead-zone is used broadly to include areas of storage marginal to the main channel and all of the small effects such as reverse flows on bends or in pools, turbulent eddies and wakes associated with roughness elements within the bulk flow. Such areas may, in fact, relate to bacterial source areas within the channel. The dispersive effect is achieved by the assumption that each reach is comprised of a number of Continuously Stirred Tank Reactors, CSTR, in series.

The concentration of a conservative solute in V is governed by changes in the inflow concentration u_i. If u_i is greater than the concentration in the dead-zone, x_i, that concentration rises. If u_i is less than x_i, x_i falls. Advection is accounted for by a pure time-delay τ. For a conservative solute or

contaminant in a single reach the model may be written as a mass-balance of the form

$$\frac{dV.x_i}{dt} = Q.u_{i-t} - Q.x_i \qquad (5.6)$$

Where, Q is discharge. Dividing both sides of Equation 5.6 by V gives;

$$\frac{d.x_i}{dt} = \frac{Q}{V}.u_{i-t} - \frac{Q}{V}.x_i \qquad (5.7)$$

(This is the basic form of the QUASAR model to which extra source and sink terms are added as appropriate for the system and determinand to be modelled.

In this case the determinant is faecal coliforms which are assumed to be completely mixed throughout the channel cross-section and the effect of cumulative entrainment longitudinally is lumped together at the reach outlet. The change in bacterial concentration is represented by:

$$\frac{d.x_i}{dt} = \frac{Q}{V}.u_{i-t} - (\frac{Q}{V} + k_b.n_d + k_w).x_i + \frac{w}{V}.Ne_{i-t_e} \qquad (5.8)$$

$$Ne_i = \sum_{j=0}^{n_e} N_{ij} \qquad (5.9)$$

Where k_w is the net faecal coliform die-off rate in water and τ_e is a time delay applied to the entrained organisms. Figure 5.4 shows the systems block diagram for the faecal coliform model: only six channel sub-stores are shown for simplicity.

5.3 Faecal coliform survival; relating die-off to physical and chemical environmental variables

The death or die-off rate coefficients, k_b and k_w, are composed of a number of terms, as derived from the literature, to account for effects of temperature, solar radiation and pH (Section 2) such that the die-off rate in water;

$$k_w = k_w + \Delta k_i + \Delta k_T + \Delta k_{pH} \qquad (5.10)$$

where

$$\Delta k_i = \alpha.I_{z,avg} \qquad (5.11)$$

is the change in die-off rate due to solar radiation, a is a rate coefficient, $I_{z,avg}$ is the average light intensity in Watts/m^2 received over the entire water depth (z). The derivation of this equation is summarised in Section 2.3 and includes example coefficients derived from earlier studies

$$\Delta k_T = k_w \{1 - 10^{\beta(T_i - T)}\} \qquad (5.12)$$

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In Equation 5.12, q is the slope of the die-off/temperature response curve. T_j and T_g are temperature in °C at time step t-1 and t_i respectively. The derivation of this equation and values of q are given in Section 2.2. In summarising the literature studies of the effect of temperature on coliform die-off, it was noted that the effect of temperature was less marked in sewage contaminated waters. Values of q were around 0.045 in non-sewage impacted waters and 0.013 in sewage contaminated waters.

$$\Delta k_{pH} = k_w \{1 - \cosh (a [pH_{k_{wmin}} - pH])\} \qquad (5.13)$$

The effect of pH on faecal coliform die-off can best be described using a hyperbolic cosine function (Equation 5.13). The die-off rate rises steeply either side of $k_{w,min}$, the pH value at which the die-off rate is a minimum, a is a rate parameter for fitting the curve. For the derivation of Equation 5.13 and values of a, refer to Section 2.7.

The equation for die-off in the channel bed store, k_b, can include the above terms if required, but the rate coefficients would be expected to be different, in fact, lower than in the water column. Alternatively, a constant value might suffice. In the model runs described in this section and in Section 6 a constant value of k_b was used. In the absence of adequate data to parameterise the

extra terms in Equation 5.10, this approach simplifies model calibration. Similarly parameterisation and data availability for these terms in k_w might lead to a further simplification, in which it is assumed that die-off varies sinusoidally with the seasons and that random daily variations in die-off, resulting from, say, cloud cover, can be ignored such that

$$k_w = k_{wmin} + k_{kw} (1 + \cos [\pi + \omega t]) \qquad (5.14)$$

where, $k_{w,min}$ is the minimum die-off rate and k_{kw} scales the amplitude of the seasonal variation in die-off. Note that the cosine function is shifted by π radians to start at a minimum, ω is the angular velocity of the Earth's orbit of the Sun in radians/day and t is the time step in days starting the shortest day of the year.

5.4 Modelling results

In the flow events sampled in the field the observed behaviour was dominated by entrainment and transport through the reach. The effects of die-off and settlement are assumed to be insignificant as the time scales over which they act are large relative to the speed a parcel of water travels through the study reaches. It was necessary to provide the initial total number of
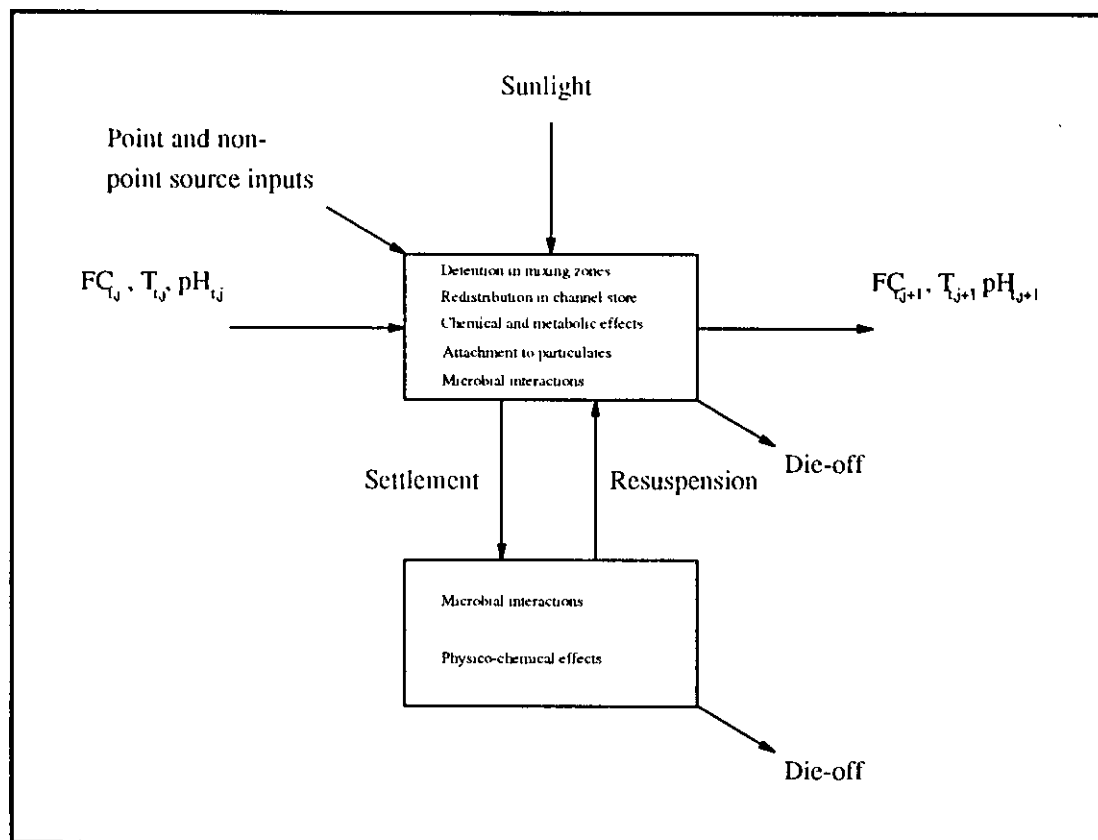
Figure 5.5 Two-box conceptualisation of the faecal coliform model as applied to a single river reach

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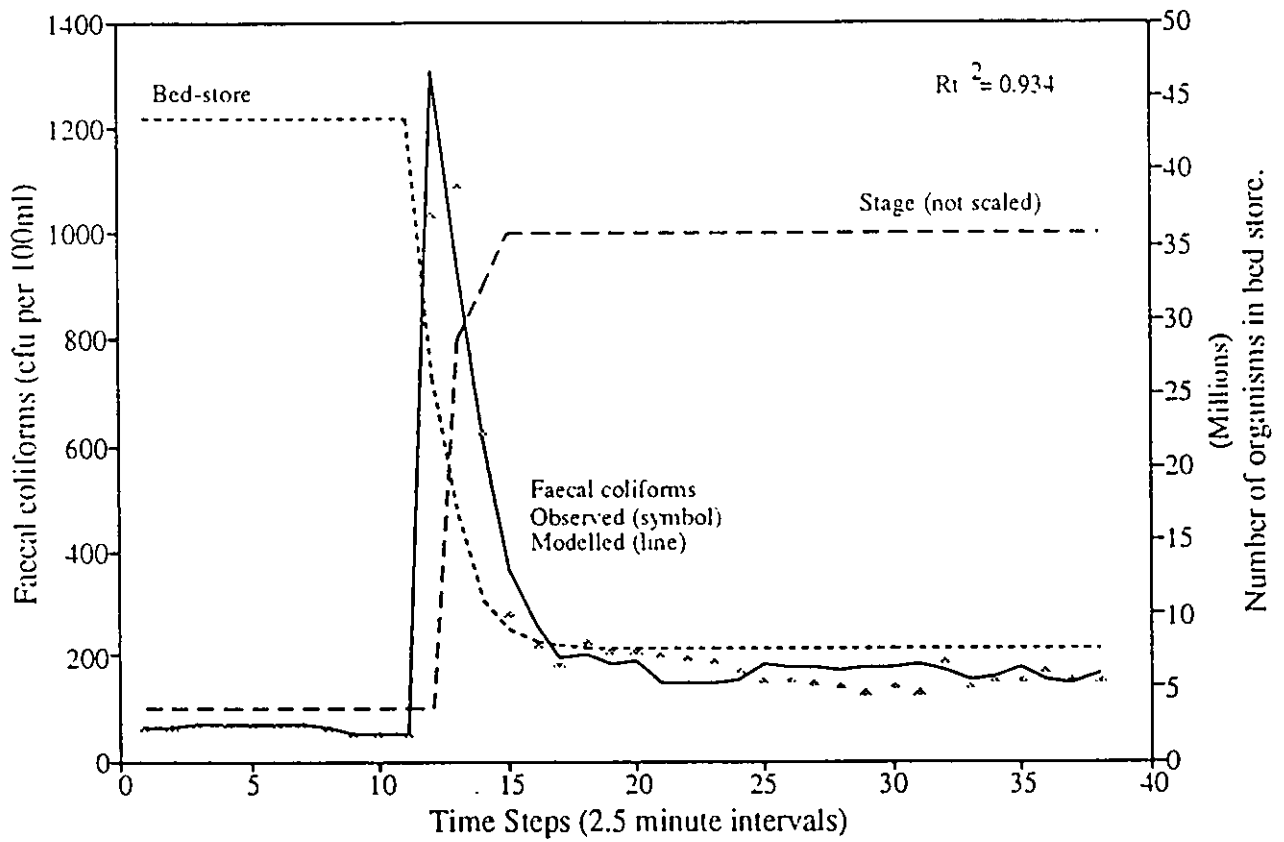
Figure 5.6a *Observed and modelled faecal coliform concentrations in response to a step change in flow on the River Washburn. Rt² is a goodness of fit criterion and should be approaching 1*
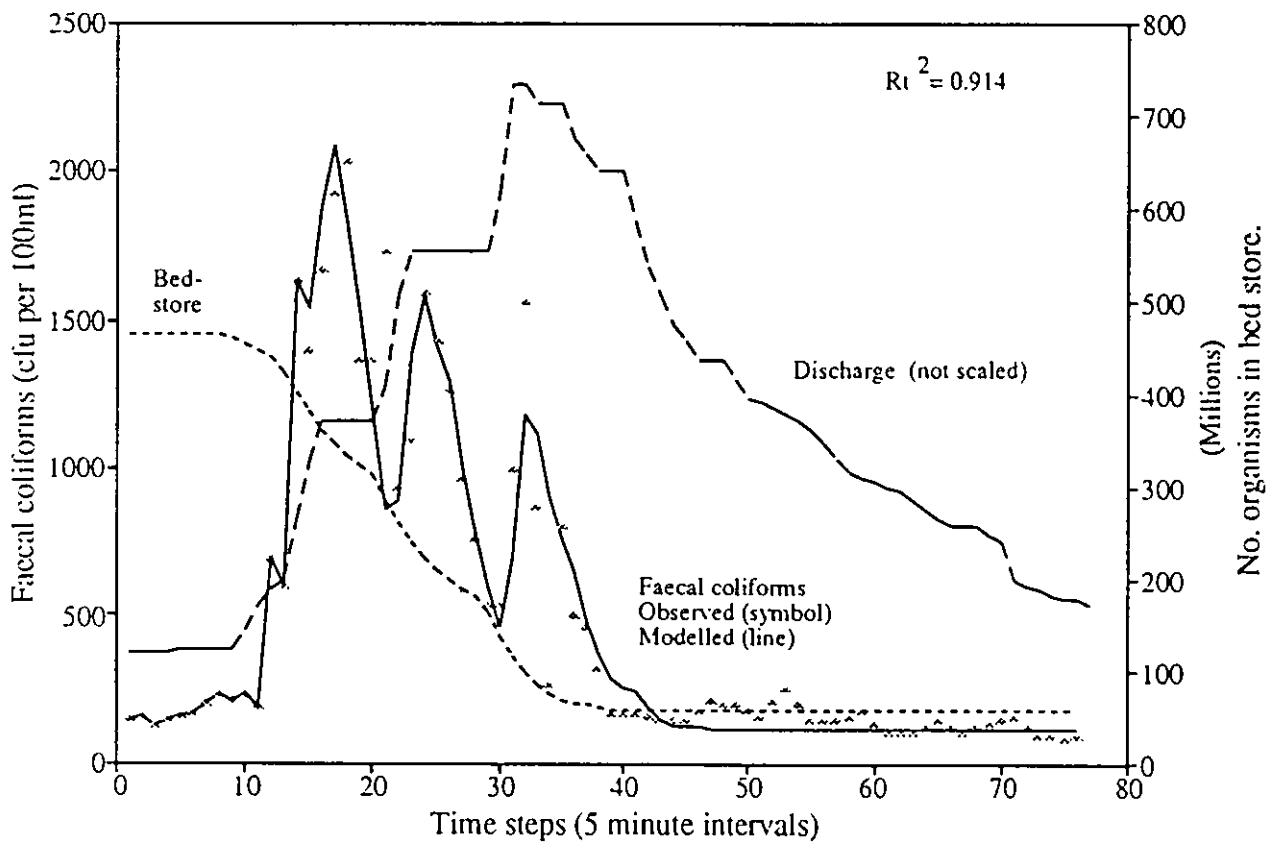
Figure 5.6b *Observed and modelled faecal coliform concentrations in response to a stepped artificial flow event on the Afon Clywedog*

Faecal coliforms (cfu per 100ml)

250

200

Bed-store

150

100

50

0

$Rt^2 = 0.548$

Stage (not scaled)

0   20   40   60   80   100   120

Time steps (5 minute intervals)

No. organisms in bed store.

(Millions)

250

200

150

100

50

0
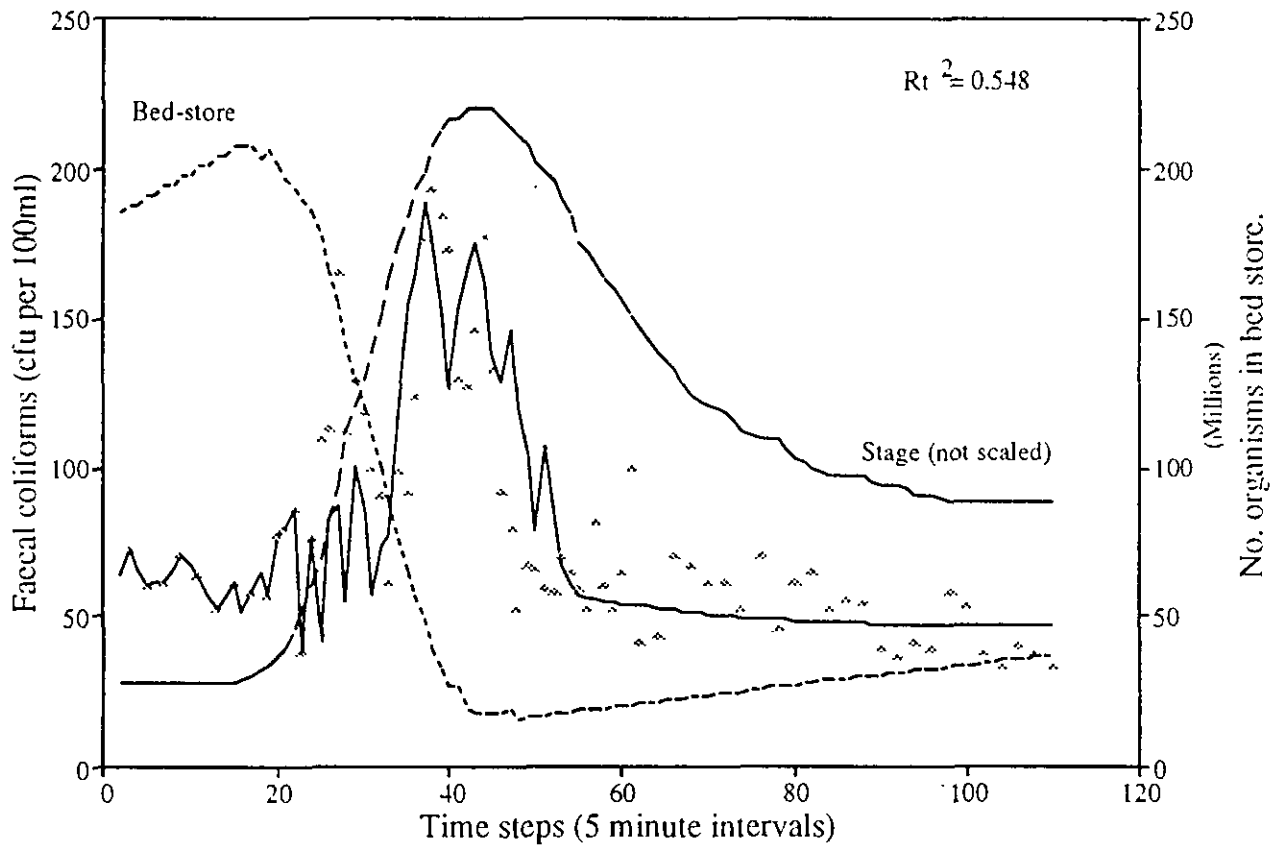
Figure 5.6c Observed and modelled faecal coliform concentrations in response to a stepped artificial flow event on the Afon Rheidol

organisms in bed-storage NT_0 and a number of organisms input to maintain the background concentration where necessary. The number of bed-store partitions was arbitrarily set at j=100. The time delay τ_e was found to give a considerable improvement in model fit for the Rheidol and Clywedog data and is in the order of 0.2-0.3 times the travel times of the respective reaches.

The observed bacterial concentrations on the Washburn and Clywedog can be modelled well (Figures 5.6,a and b). On the Rheidol, however, the model fit to the observed data was not as good (Figure 5.6,c). This may be attributable to a number of factors. The reach length is such that the stage increments upstream are attenuated out at the sampling site, hence there is insufficient information to perturb the model. The reach

studied, meanders and contains a number of steep bends. Dead-zones downstream of these bends acting as source areas of organisms may dominate entrainment resulting in a response that is not regular. The reach is regularly flushed by releases for hydroelectric power resulting in low faecal coliform concentrations at which considerable random variation in detection levels might be expected.

The model presented here gives a good conceptualisation of in-stream faecal coliform dynamics, being able to reproduce field observations where existing models would fail. Further developments might include modelling other microbiological determinands (e.g. faecal streptococci) and particle associated contaminants such as heavy metals or hydrocarbons.

6 Model application and validation

The new model for faecal coliform dynamics in stream channels has been shown to give a good description of entrainment episodes in response to artificial flow releases. An application of the model to data from the routine monitoring programmes of the National Rivers Authority and water supply companies is presented.

The difficulties presented by model application to data not intended for modelling are discussed and techniques for making model application and calibration possible demonstrated. Following calibration, model response to hypothetical land-use and climate change scenarios is examined.

6.1 Data availability for model application and validation

The completion of a detailed questionnaire by the laboratories responsible for analyses of both the NRA and water companies' data established the use of common analytical standards and practices (H.M.S.O., 1983) thus ensuring that it would be acceptable to use faecal coliform data from these different sources. Where possible a broad suite of other water quality determinands were included and some of the locations coincided with flow gauging sites, the latter being available from the IH National Water Archive. The microbiological data are stored on the IH database and have been used in an analysis of relationships with land-use (see Section 7).

The faecal coliform model described in Section 5 requires input/output (i/o) data and is applied on a reach by reach basis. The minimum data requirement for model calibration/validation is i/o time-series of faecal coliform concentrations for the top and bottom of the modelled reach and flow data for one site. Data for solar radiation and turbidity would also be beneficial for calibration of the die-off component of the model, however, given certain assumptions die-off can be made into an implicit function of the model as will be shown later.

Much of the data available to this project were for single sampling sites or multiple sampling sites that are too far apart. These data are unsuitable for model calibration/validation for the following reasons.

1. In the case of a single sampling site with flow data, the modeller has two choices; treat the data as either the input or output time-series i.e. attempt to model a reach downstream or upstream of the site for which the data are available. If the data are to be treated as an input series, the modeller can use best estimates of parameter values from knowledge of existing applications or from the literature but can have no idea of the accuracy of the model prediction. It simply cannot be compared with actual values. Similarly, if the data were to be treated as an output time-series, it may be possible to reproduce the observed faecal coliform concentrations but it would be very difficult to establish the source of organisms resulting in the observed bacterial concentration and care would be required in choosing realistic parameter values.

2. Where multiple sites on one river system are concerned, the main problem is the distance between the sites. The greater the distance between the sites, the larger the number of tributaries contributing to the flow and bacterial load, and the greater the likely number of effluent discharges. Although it might be possible to calibrate the model between the sites, it would be necessary to make gross assumptions about the inputs and impossible to assign accurately a weighting to the impact of each on the whole system.

Of the many sampling sites in England and Wales for which faecal coliform data are held, only six river systems had potential for model application. Of these, the Tyne and the Thames had too few data points while the Wye, Teifi and Dee had multiple sites which were too far apart. All other sites were single monitoring points.

This left the River Exe in Devon with two sampled sites along a 6 km reach (see Figure 6.1). The upstream site is sampled by the National Rivers Authority approximately every two to three weeks and is located downstream of the Thorverton sewage effluent discharge. The site coincides with a flow gauging station for which data could be retrieved from the IH National Water Archive.

The downstream site is at the Pynes raw water intake operated by South West Water plc. This site is sampled on a weekly basis. Data for the years 1990 and 1991 were chosen for model application.
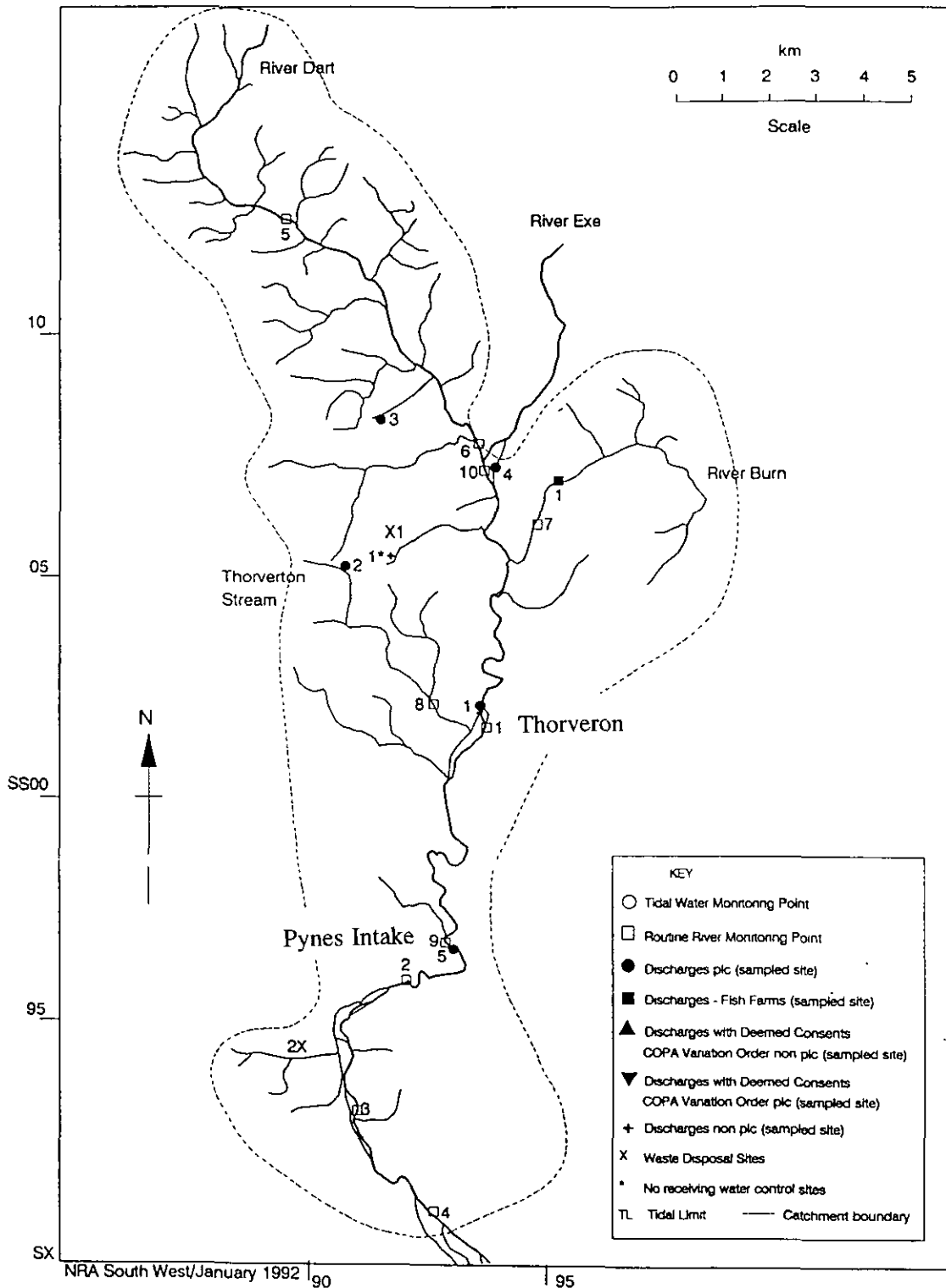
Figure 6.1 The River Exe showing sampling sites at Thorverton (1□) and Pynes raw water intake (9□), the faecal coliform model has been applied to the 6 km reach between these sites

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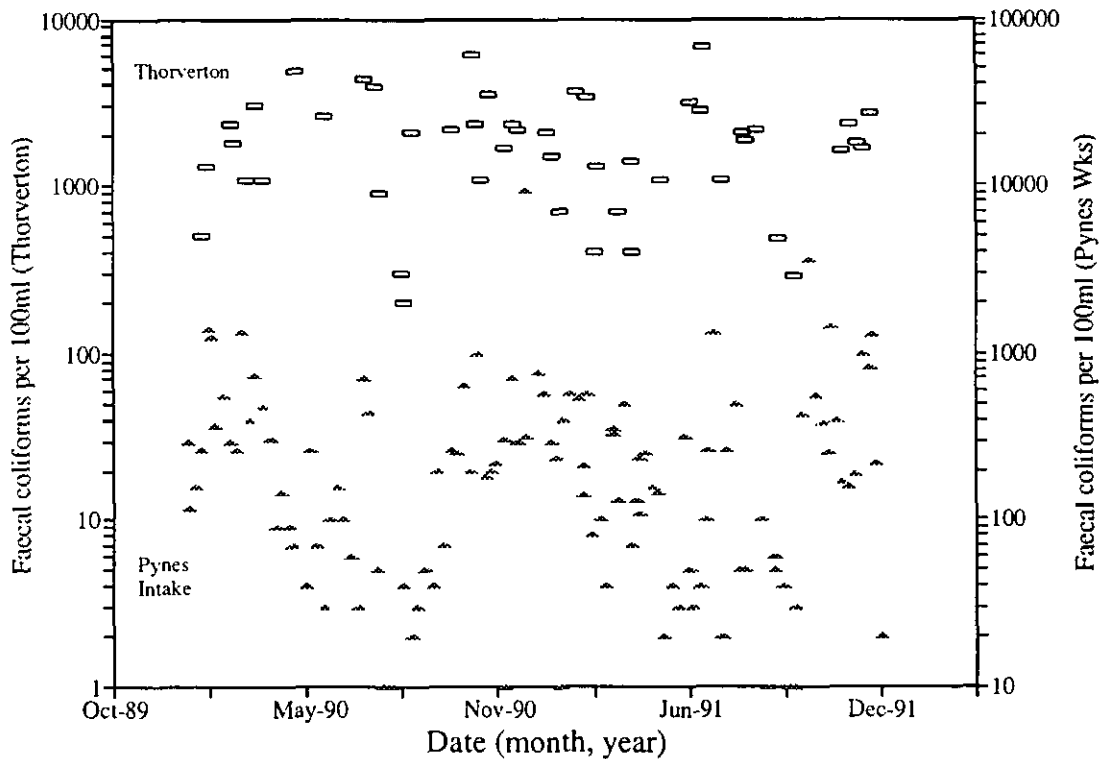
Figure 6.2 *Time series of faecal coliform concentrations at Thorverton STW and the Pynes water intake on the River Exe in Devon; data are presented in their raw 'as sampled' state*

6.2 Data preparation

Because of the dynamic nature of the relationship between flow and faecal coliform concentrations it was decided that a daily (or ideally, more frequent) time-step was suitable for running the new model. To run the model on a daily basis, it was necessary to provide daily time-series for flow and the upstream faecal coliform input concentration. However, although daily flows were available from the National Water Archive for the upstream site at Thorverton, it was necessary to generate a new daily time series for the faecal coliform concentration since sampling for this parameter only took place at two to three week intervals.

The time series for the Pynes intake consists of regular weekly samples taken as part of the routine raw water monitoring undertaken at that works (Figure 6.2). The values obtained for this site demonstrate a seasonal trend but it was difficult to discern a similar trend in the data for Thorverton, making it difficult to decide the best way of generating a new time-series for that site.

It was assumed that the faecal coliform concentration at Thorverton was dominated by the impact of the effluent discharge and that the concentration would vary seasonally according to the dilution afforded by the flow in the river. To simulate the high variance observed in most faecal coliform time-series, a random noise function was superimposed on the seasonal trend from which the new data were derived (Figure 6.3).

Comparison of Figures 6.2 and 6.3 shows that the time-series of faecal coliform concentrations for Thorverton and Pynes Intake exhibit opposite seasonality. A possible explanation for this difference is as follows: the concentrations at ‘ Thorverton are assumed to result from dilution by the flow of the dominant input of organisms from the sewage discharge upstream. Sewage effluent derived faecal coliforms are likely not to be associated with settleable solids and will lack the protection afforded by such particulates from harmful sunlight and to lesser extent microbial predation (Sections 2 and 3). In the reach downstream of Thorverton rapid die-off in the water column would be expected. At higher flows travel times through the reach will be reduced, hence the proportion of released organisms reaching the downstream site will be greater. Further, the supply of solids from the channel and catchment upstream of the reach will improve the survival conditions resulting in slower die-off.

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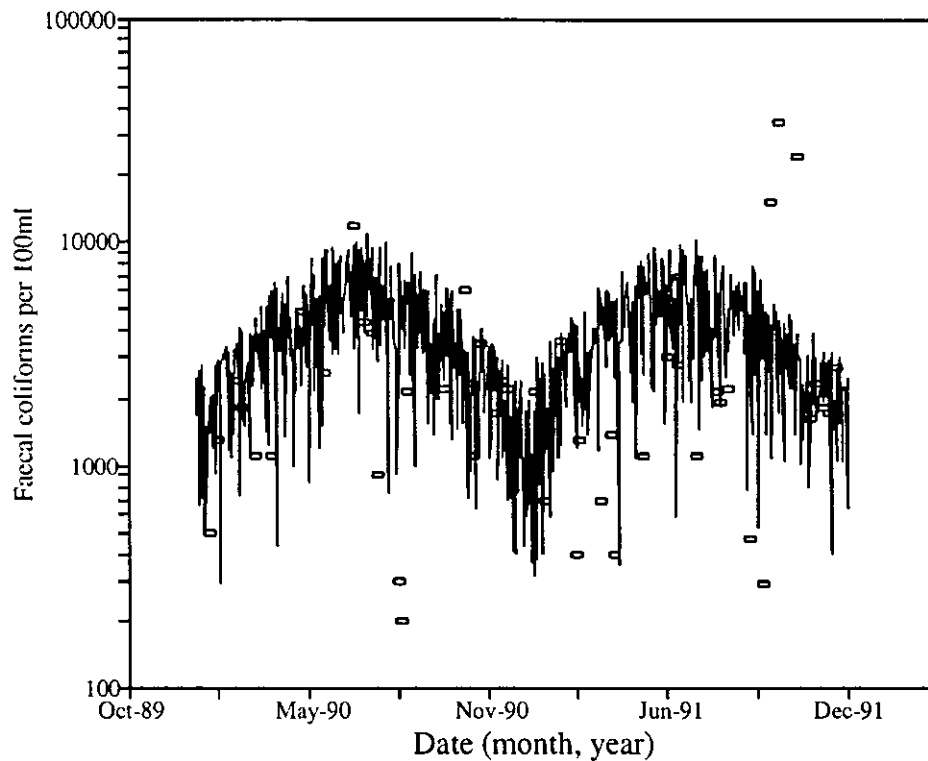
Figure 6.3 Raw data (□) and artificially generated time series (solid line) of faecal coliform concentrations for the River Exe at Thorverton

6.3 Model calibration

The values of parameters chosen for the calibration were initially derived from the literature (Auer and Niehaus, 1993; Evison, 1989; Flint, 1987; Shereretal., 1992). Manual adjustment was used to achieve the model fit (Table 6.1). The value for settlement velocity chosen (0.04 m.d^{-1}) is far lower than that found by Auer and Niehaus (1993). Their value of 1.17 m.d^{-1} was for particulates in the range 0.45-10 µm, with which over 90% of faecal coliform bacteria were found to be associated. This settlement velocity was estimated from settlement trap accumulation in a lake environment. The lower value used in this study may be justified in two ways; the first is the fact that the stronger mixing in river flow will result in a

lower net accumulation compared to that in a lake and the second is that although the actual gravitational settlement rate is unchanged, forward and upward motion within the flow will result in less settlement.

Die-off in the channel-bed store was assumed to occur at a constant rate (Table 6.1). Die-off in the water column was modelled with the equation (see Section 5.3 and Table 6.1);

$$k_w = k_{w\,mm} + k_{bw}\,(1 - \cos\,[\pi + \omega f]) \tag{6.1}$$

This function can be used in the absence of data for solar radiation and temperature and was found to simulate the seasonal effect of these variables very well (Figure 6.4b).

Table 6.1 Parameter assignments for the faecal coliform component of QUASAR

Parameter	Description	Assignment	Value
kw	die-off in water	cosine seasonal trend	0.3 to 0.9 per day
kb	die-off in bed stores	constant	0.007 per day
ks	settlement velocity	constant	0.004 m.d^{-1}
w	washout rate	constant	0.04 per day
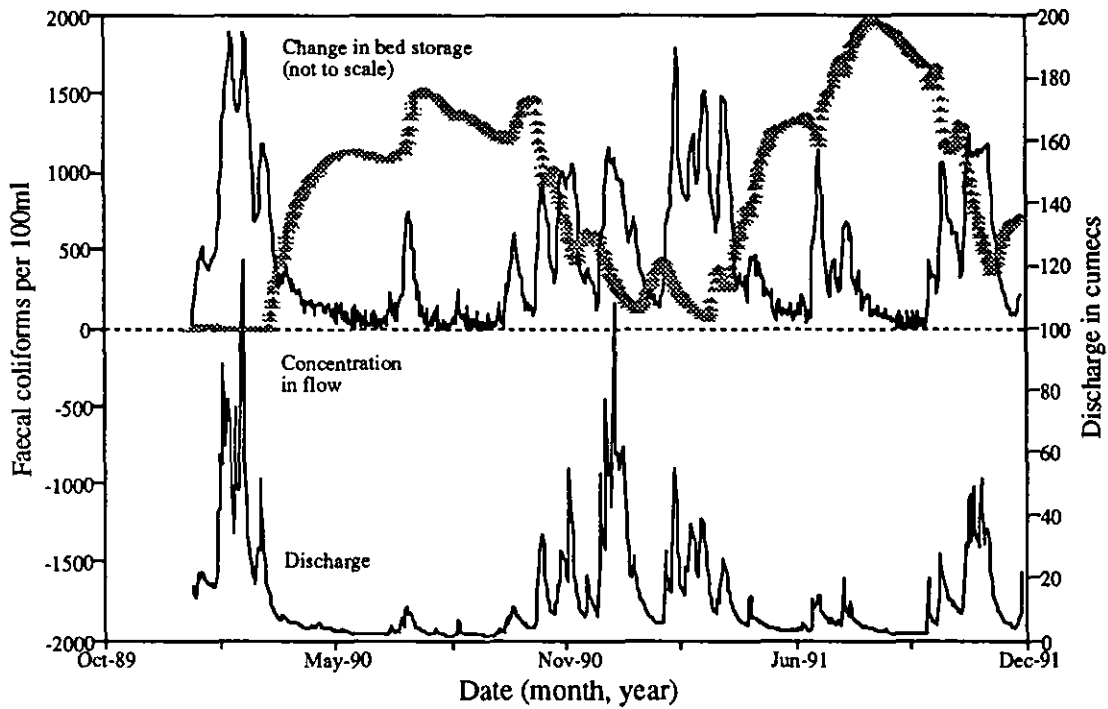
Figure 6.4a *Time series of flow and modelled faecal coliform concentration and channel storage for the River Exe at Pynes water intake*
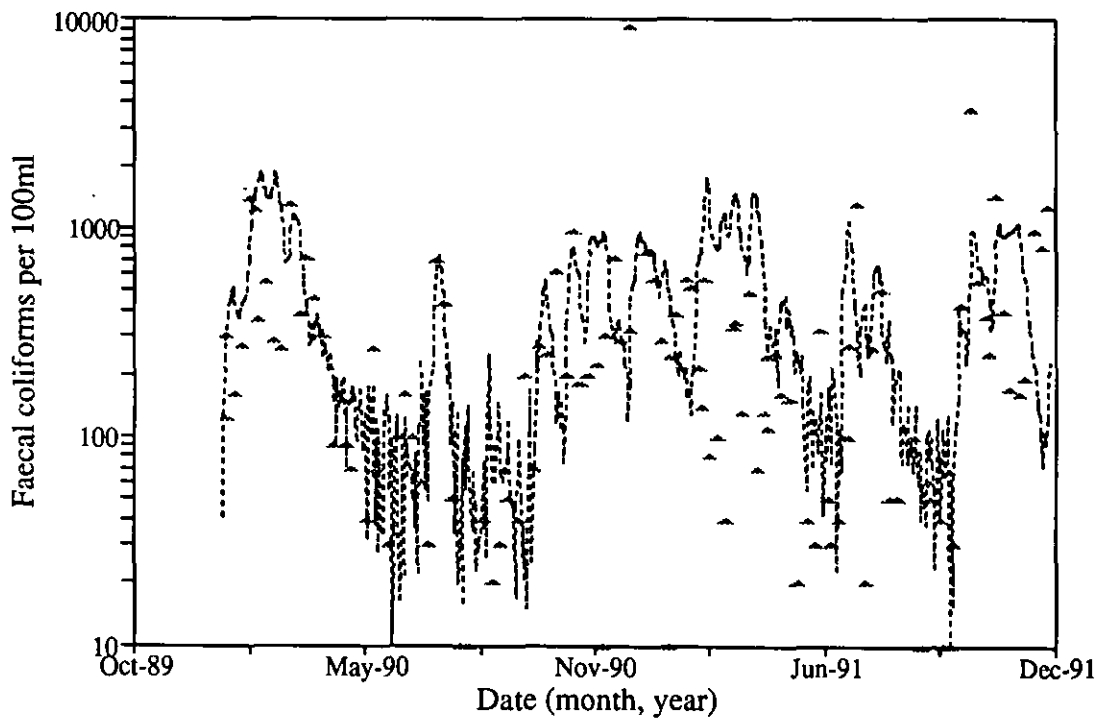
Figure 6.4b *Modelled and observed time series of faecal coliform concentrations for the River Exe at Pynes water intake*

Figure 6.5 is a time series chart with axes labeled "Faecal coliforms per 100ml (reach 2)" (left, 6000 to -6000) and "Faecal coliforms per 100ml (reach 6)" (right, Thousands, 12 to 0). Curves labeled "Reach 2" and "Reach 6". X-axis "Date (month, year)" from Oct-89 to Dec-91.
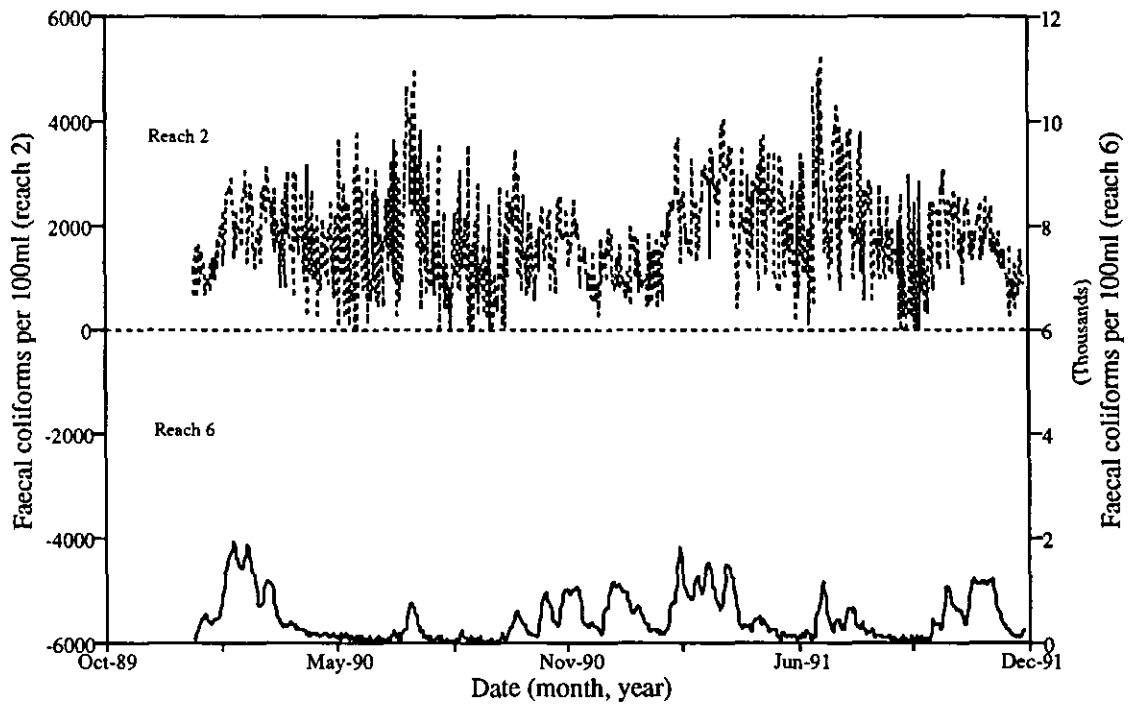
Figure 6.5 Time series of modelled faecal coliform concentration at the output of model reaches 2 and 6; each reach is 1km in length

Figure 6.6 is a time series chart with y-axis "Number of faecal coliforms (Times 10E12)" (0 to 350) and x-axis "Date (month, year)" from Jan-90 to Dec-91. Legend box "Reach Number" with curves labeled 1, 2, 3, 4, 5, 6.
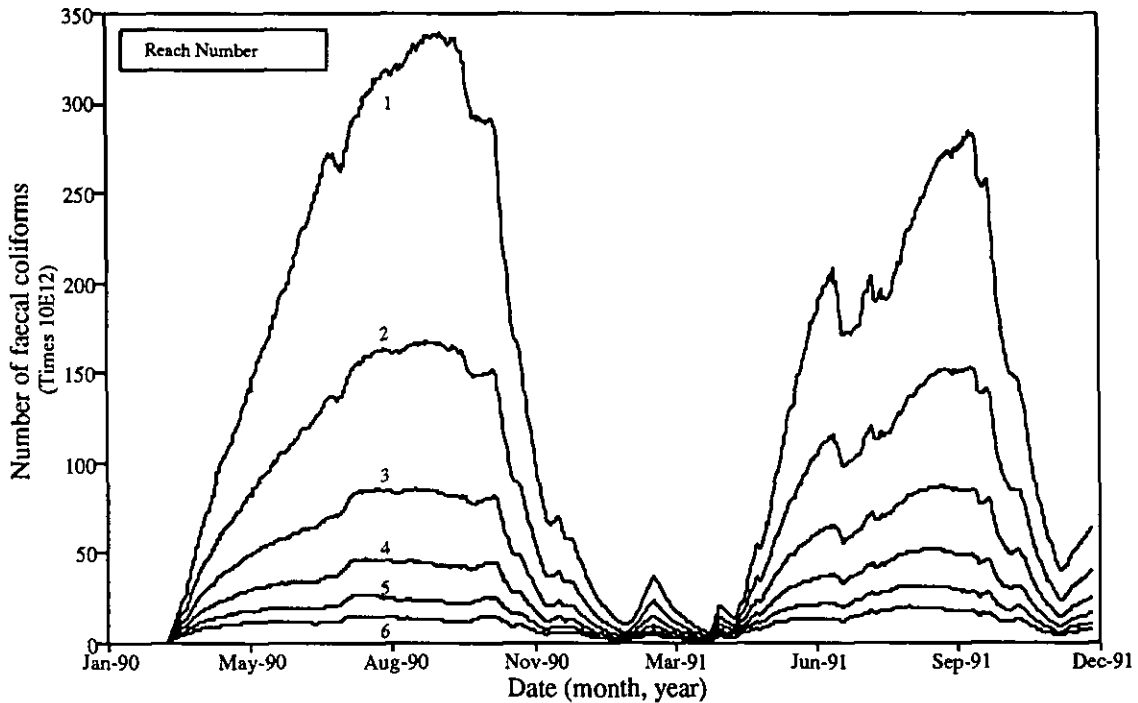
Figure 6.6 Time series of modelled faecal coliform numbers in channel storage for reaches along the River Exe between Thorverton STW and Pynes water intake

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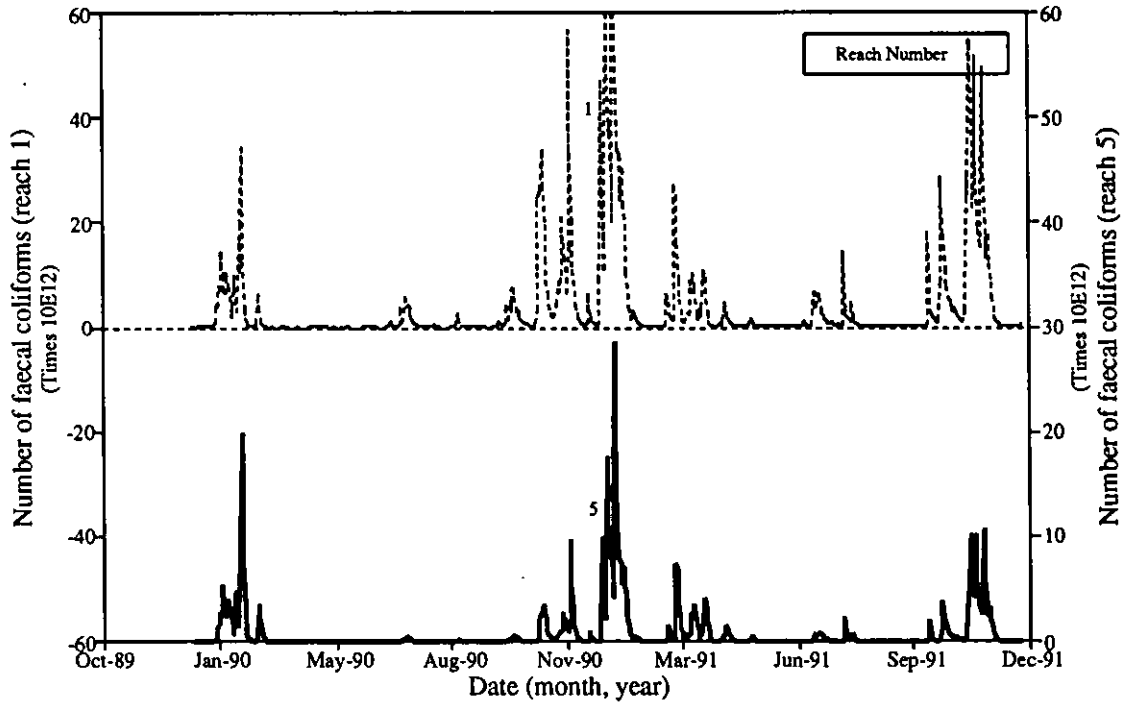
Figure 6.7a *Time series of modelled faecal coliform entrainment episodes from channel storage during flow events; reaches 1 and 5 are shown*
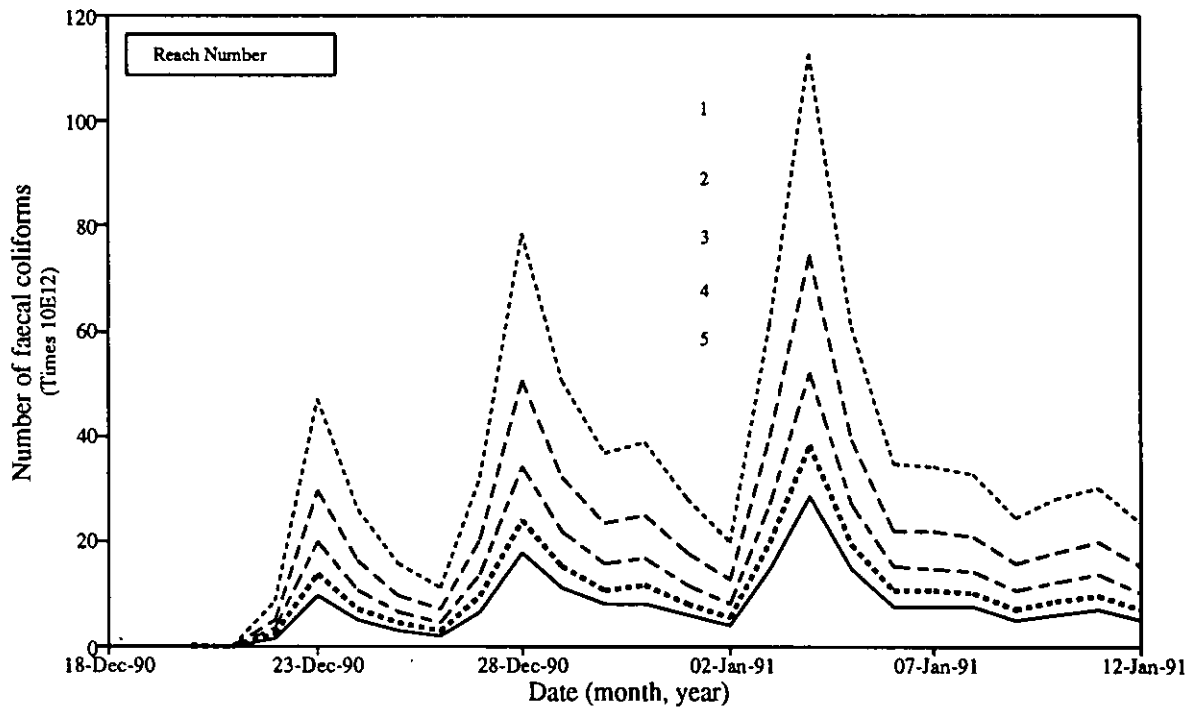
Figure 6.7b *Time series of three modelled faecal coliform entrainment episodes over the New Year period 1990-91 in model reaches 1 and 5 of the River Exe between Thorverton STW and Pynes water intake*

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6.4 Model operation

Figures 6.4 a and b show the model fit to the observed data and time series of flow and change in channel storage. Figure 6.4a shows the change in channel-bed storage with flow; bacterial concentration is also shown. A rise in channel-bed faecal coliform numbers coincides with each hydrograph recession. The general trend indicates the net channel flushing during winter high flows and accumulation during the summer months. Once settlement has reduced the numbers of organisms available for settlement from the water column, net die-off in the channel store commences. This is seen after the summer flow events of July 1990 and June 1991. Figure 6.5 shows how the bacterial concentration changes with distance downstream. The concentrations at the top of the modelled reach are higher than at the bottom. This results from die-off and settlement.

Figure 6.6 shows the rapid reduction in channel storage with distance from the effluent discharge at Thorverton reflecting the reduced concentrations in the overlying water. Figures 6.7a and b show episodes of entrainment from channel storage which coincide with the flow events in the river.

6.5 Scenario tests; input loadings

The aim of the coliform input loading scenarios was to demonstrate the effect of simple increases and decreases in faecal coliform concentrations that might result from changes in population density, land-use/stocking practices or sewage effluent treatment processes. Figure 6.8 shows the changes in model output to increases of +15% and reductions of -10% in faecal coliform loadings. The changes in loading were made by simple multiplication of the $\log_{10}$ transformed artificially generated input time series for the Thorverton sampling site.

Table 6.2 shows the percentage changes in arithmetic and geometric mean faecal coliform concentrations for the two input scenarios. The increased input scenario results in an approximate 20% increase in the $\log_{10}$ mean faecal coliform concentrations, the reduced input scenario causes means which are approximately 13 and 15% lower at Pynes Intake.

6.6 Scenario tests; impacts of climate change on flow regimes

The impact of climate change on flow regimes and consequently on faecal coliform concentrations has been examined, in particular, the impact of the change in flow on the modelled
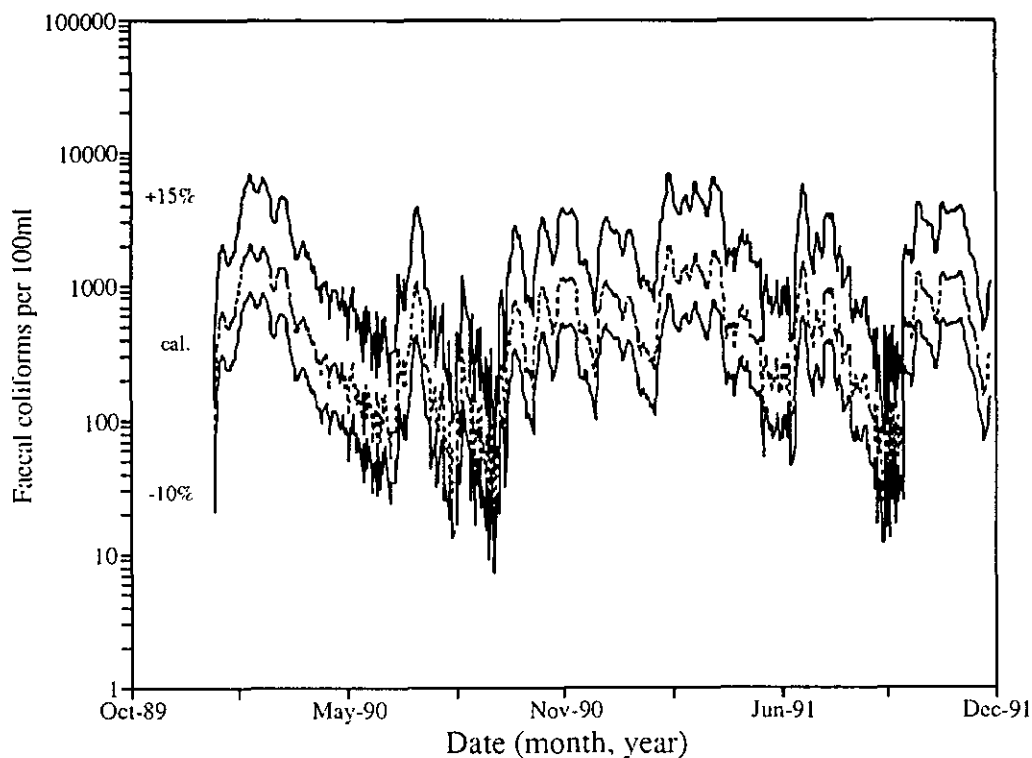
Figure 6.8 *Time series of modelled faecal coliform concentration at Pynes water intake showing the impact of increased and reduced faecal coliform loadings*

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*All faecal coliform concentrations no. per 100ml	Mean faecal coliform concentration	Geometric mean faecal coliform concentration	Maximum flow in cumecs
Modelled	453	273	122.03
Dry scenario	322 (-5.6%)	192 (-6.5%)	78.79
Wet scenario	491 (+1.1%)	302 (+1.6%)	137.89
15% increase in inputs	1529 (+19.5%)	942 (+21.7%)	122.03
10% reduction in inputs	202 (-13.15%)	117 (-15.2%)	122.03

reach for the wettest and driest scenarios projected for the year 2050.

Arnell (1992) suggests that increasing concentrations of greenhouse gases will have both direct and indirect effects on hydrological processes. The most obvious impact will be on the magnitude, intensity, duration, frequency and timing of rainfall events, with the obvious impact on the flow regime of the river draining the catchment. Examples of possible changes might be a flashier response resulting from drier antecedent conditions and perhaps more extreme convective rainfall events, generating rapid flow in desiccation cracks with low infiltration into the soil matrix. Conversely a slower response might result from more extended periods of frontal rainfall, the catchment might be wetter and infiltration into the soil matrix might result in the catchment draining more slowly. Changes in effective rainfall resulting from changes in evaporative losses from the catchment will also be very significant.

The UK Climate Change Impact Review Group (CCIRG) has proposed wettest and driest scenarios with reductions or increases of up to 30% in annual average runoff in the year 2050 (Arnell, 1992).

Scenarios of +/-20% are examined in this study by the modification of the 1990-92 flow series for the River Exe at Thorverton (Figure 6.9). The flow series is adjusted using a simple transfer function of the form:

$$\frac{Qo_k}{Qi_k} = \frac{c.b_0.z^{-1}}{1 - c.a_1.z^{-1}} \qquad (6.2)$$

Where, Qi and Qo are the existing and modified flow series, $a_1 = 1-b_0$, $b_0 > a_1$ resulting in a steeper flood peak and shorter hydrograph recession and *vice versa*. The multiplier c is adjusted to raise or lower the annual average runoff, $c=1$ for no change.

For the purpose of these tests, the faecal coliform concentration input to the model was kept the same in order to examine the effect of the change in flow on the modelled faecal coliform concentration at Pynes intake. Figure 6.10 shows the model output for the driest and wettest scenarios; in general the differences in concentration between the two scenarios are minimal (Table 6.2), although for the driest scenario the low flow values of faecal coliform concentration are much lower than for the wettest scenario. This difference can be attributed to the increased residence time of the water in the river reach resulting in enhanced settlement and die-off.

The likely impacts of climate change on faecal coliform concentrations are as uncertain as the impact on flow regimes. The evidence presented here suggests that higher temperatures and more sunshine hours will result in more rapid die-off because drier conditions will lead to more rapid die-off on the catchment surface and in soils. Conversely, wetter conditions will result in lower river residence times resulting in less die-off; more moist soils will improve survival and may lead to more rapid transport through the catchment. Increased cloud cover will result in extended survival.

In general, the model is seen to operate well for these extended periods of data. The numbers of organisms in the channel store are stable and in effect self regulating. No initialisation value is needed for the channel store. The entrainment and settlement functions perform well; detailed discussion of this behaviour is given in Section 5. The seasonal trend observed in the data is modelled with a simple cosine function for die-off changes resulting from solar radiation and temperature and overcomes the problems resulting from lack of data for these environmental variables. A further benefit of this function is a reduction in the number of parameters needed to calibrate the model. Only four parameters are

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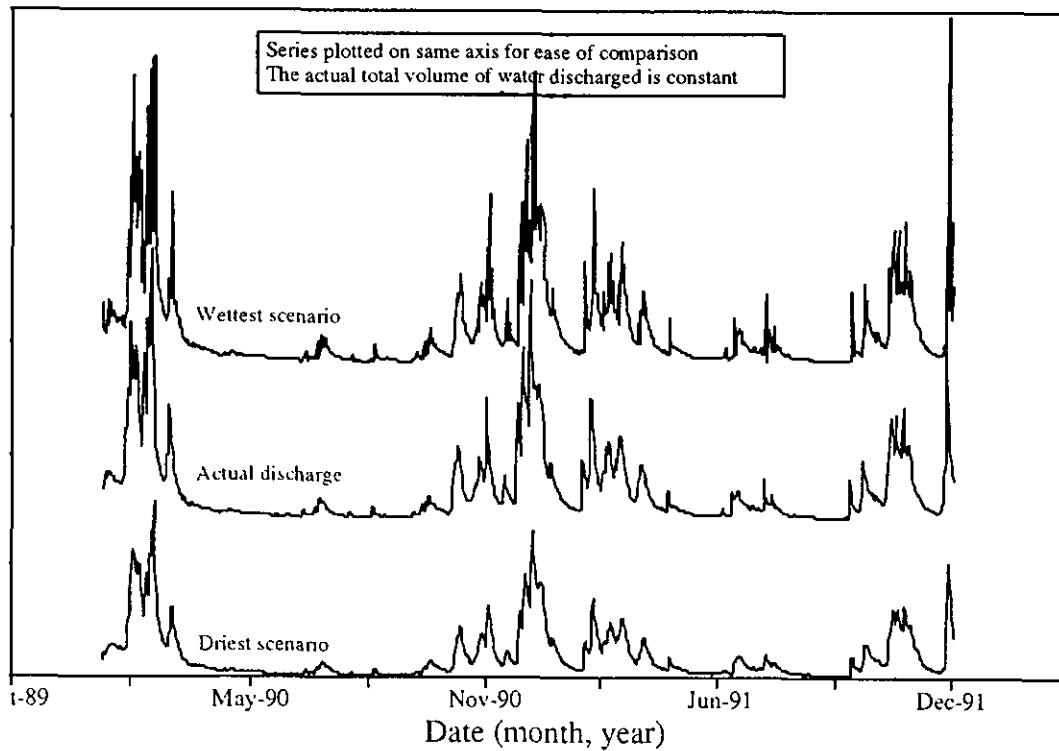
Figure 6.9 *Time series of actual and modified flows in the River Exe at Thorverton. The flows have been modified on the basis of CCIRG projections for the year 2050 (Arnell, 1992)*
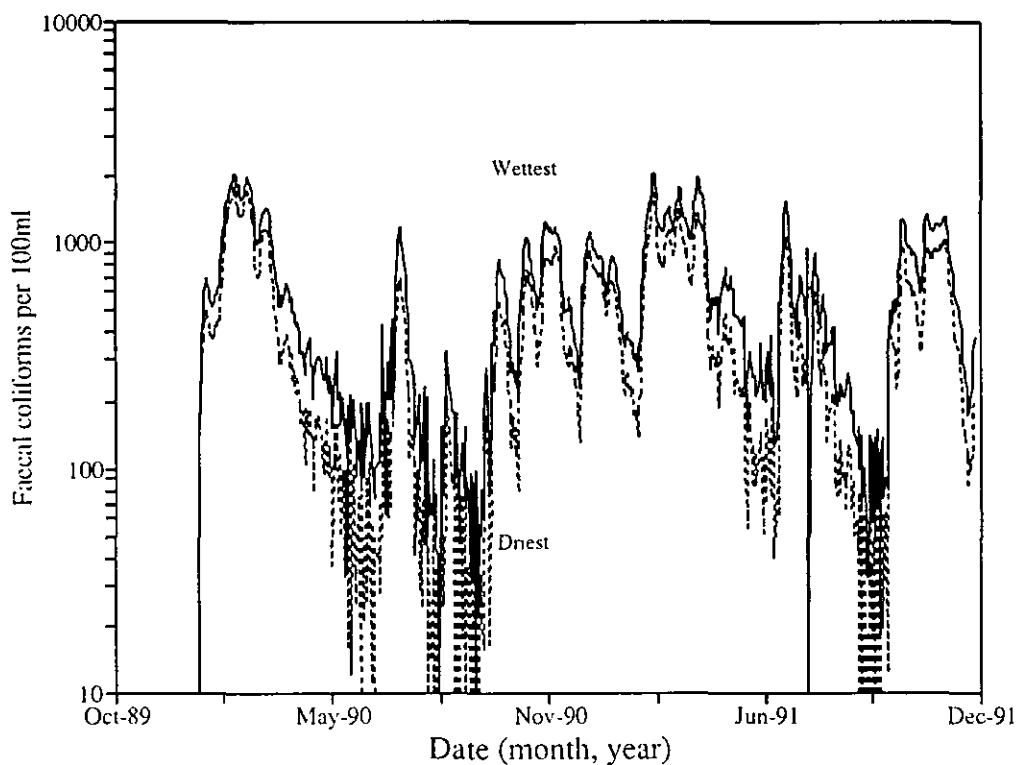
Figure 6.10 *Time series of modelled faecal coliform concentrations in the river Exe in response to the driest and wetted flow predictions shown in Figure 6.9*

required, a scalar for the water die-off coefficient, bed-store die-off coefficient, settlement velocity and entrainment rate. No previous model has given a satisfactory description of faecal coliform river dynamics; the model applied here not only gives a good fit to the observed data but also has scope for application to other water quality determinants. These include particulates and particulate-associated contaminants such as heavy metals, organic compounds and radio-nuclides.

7 Relationships between faecal coliform concentrations, land-use and catchment characteristics

This section summarises two studies examining the relationship between stream faecal coliform concentrations and descriptors of catchment land-use and farming practices. The aim is to develop the ability to predict non-point (land-use) derived faecal coliform delivery in a catchment. Such a relationship could be incorporated into the new model of faecal coliform stream dynamics, enhancing the ability to differentiate between point and non-point sources of contamination.

The first study provides an analysis of faecal coliform data for a set of upland catchments in Wales, investigating the relationships between faecal coliform concentration and the Agricultural Development and Advisory Service (ADAS) land classification data as well as information derived from a farm questionnaire survey of fertiliser use and animal populations in the target catchments. The second study examines the relationship between faecal coliform concentration and the Institute of Terrestrial Ecology's land-cover classification system.

7.1 ADAS land classification data and farm questionnaire study

This study defined the relationship between land-use and faecal coliform delivery for a set of upland catchments in west Wales. The use of catchment descriptors relating to the supply of organisms to the stream channel might assist in the prediction of non-point source faecal coliform concentrations both for planning purposes and in the calibration of a river network model such as QUASAR.

Materials and methods
The study catchments were within the acid-sensitive region of west Wales (Hornung et al., 1990). This area is in the "Grade A Less Favoured Area" defined by The Welsh Office Agriculture Department (W.O.A.D.) (W.O.A.D., 1984). Land in the catchments is suitable for either (i) upland hill farming (mainly sheep with some dairy and beef cattle production) or (ii) coniferous forestry (W.O.A.D., 1986a, 1986b). Land-use data for each catchment were acquired by (i) digitising A.D.A.S. land classification maps (A.D.A.S., 1969) and (ii) a farm questionnaire survey following initial contact via National Farmers Union and Farm Workers Union offices. The latter survey included

variables such as fertilizer use (Limestone and NPK) and populations of sheep and cattle. Stock on agricultural upland catchments represent an important source of faecal coliform organisms. Stocking densities of ewes, cattle and the combined total were calculated per unit area of agricultural land (Grade IV + Grade V) (animals.km^{-2}).

Streams were sampled at the catchment outlet at regular intervals (generally weekly) between January and October 1984. Because of this sampling framework, these 'spot' samples tend to reflect base-flow stream conditions.

After aseptic collection in pre-sterilized 250 ml Pyrex glass-stoppered bottles, samples were returned to the laboratory in a cold box and analysed within six hours. In the laboratory, samples were first diluted in 99 ml and 90 ml of sterile Ringers solution to provide serial dilutions. Generally 10 ml and 100 ml of original sample were filtered through Gelman 0.45 μm microbiological filters. The filters were placed on a Membrane Lauryl Sulphate Broth (Oxoid) and incubated in copper canisters placed in a calibrated water bath for four hours at 30°C. The canisters were then transferred to a second water bath for a further 14 hours incubation at 44°C (± 0.25°C). Thermotolerant coliform enumeration followed recommended practices (H.M.S.O. 1983: Section 7.9.4.2, Page 46). The count at 18 hours is technically a faecal coliform organism or thermotolerant coliform count (see H.M.S.O., 1983, section 7.9.2, Page 45). No confirmatory procedure was adopted to define the numbers of *Escherichia coli* within the overall thermotolerant coliform group enumerated. This is normal practice for raw water (H.M.S.O., 1983: Page 46). All counts are expressed per 100 ml.

Results
Agricultural land classes fall into three categories in the study catchments; (i) forest, (ii) Grade IV and (iii) Grade V. Grade IV class includes land in the valley bottoms used for silage production, lamb fattening and some dairy/beef production. Grade V class covers areas of open moorland used for lamb production. The areas of catchments in these three land-use categories are shown in Table 7.1. This Table also details catchment areas and annual rainfall. Table 7.2

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Table 7.1 *Study catchment areas, annual rainfall and land-use class areas*

Catchment	Area(km²)	Rainfall(mm)	Forest(km²)	Grade IV(km²)	Grade V(km²)
Berwyn	10.00	1715	6.57	1.33	2.10
Groes	12.40	1715	1.51	2.72	8.17
Camddwr	15.97	1225	1.53	7.16	7.28
Ystwyth Trib.	2.36	1470	2.08	0.00	0.28
Nant Milwyn	3.89	1470	0.03	0.03	3.83
Nant Ceiswyn	8.30	1715	6.74	0.00	4.57
Afon Dulas (N)	6.71	2100	5.56	0.00	1.15
Afon Dulas	26.67	2200	16.90	0.47	10.28
Afon Cerist	7.05	2300	0.03	0.18	6.84
Nant Iago	6.18	2000	3.60	0.05	2.53
Afon Harnog	9.65	2200	2.60	0.26	6.79
Nant Helgog	3.54	2200	0.00	0.21	3.33
Nant Mwyro	4.76	2000	0.01	0.00	4.75

Table 7.2 *Fertilizer use and stocking rates*

Catchment	NPK(tonnes)	Limestone(tonnes)	Breeding ewes	Cattle
Berwyn	17	106	2435	47
Groes	33	122	5825	1071
Camddwr	220	205	5476	302
Ystwyth Trib.	0	0	0	0
Nant Milwyn	0	0	1400	0
Nant Ceiswyn	15	50	1680	30
Afon Dulas (N)	1	0	200	3
Afon Dulas	23	257	2380	47
Afon Cerist	30	20	2693	81
Nant Iago	14	0	1060	70
Afon Harnog	4	0	460	55
Nant Helgog	0	0	450	0
Nant Mwyro	0	0	1790	15

Table 7.3 *Pearson correlation coefficients (r) (and significance (p)) between land classes, rainfall and land-use survey variables*

Variable	% Forest	% Grade IV	% Grade V
Rainfall	-0.0301 (0.461)	-0.5621* (0.023)	0.2679 (0.188)
NPK	-0.2384 (0.216)	0.9059* (0.000)	-0.1184 (0.350)
Limestone	0.0692 (0.411)	0.6011* (0.015)	0.2939 (0.165)
Breeding ewes	-0.4010 (0.087)	0.8129* (0.000)	0.0917 (0.383)
Cattle	-0.2412 (0.214)	0.8915* (0.000)	-0.1113 (0.359)
Total animals	-0.3959 (0.090)	0.8706* (0.000)	0.0624 (0.420)
Ewe density	-0.1969 (0.260)	0.3662 (0.109)	0.0528 (0.432)
Cattle density	-0.1745 (0.284)	0.8561* (0.000)	-0.1679 (0.292)
Animal density	-0.2104 (0.245)	0.4549 (0.059)	0.0308 (0.460)

n = 23 in all cases   *significant at α < 0.05

summarises fertilizer and stock data from the farm survey.

Relationships between the proportion of land in the three use categories and land-use data from the farm surveys were investigated using Pearson product moment correlation. This analysis revealed strong positive correlation coefficients (r) (p < 0.02) between fertilizer use, stock numbers, cattle density and the proportion of Grade IV land in the catchments (Table 7.3). The proportion of Grade IV land also showed a significant inverse relationship with annual precipitation.

Table 7.4 *Summary statistics for faecal coliform concentrations (count 100 ml⁻¹) in the study catchments*

Catchment	Geometric Mean	Log₁₀Std Dev	Minimum	Maximum	N
All Samples	26.797	1.039	1.00	8912.51	328
Afon Berwyn	142.880	0.881	2.00	3019.95	29
Afon Groes	304.492	0.859	6.03	6025.60	27
Camddwr	499.035	0.878	13.18	8912.51	23
Ystwyth Trib.	4.420	0.715	1.00	72.44	27
Nant Milwyn	40.400	0.717	0.00	2.84	31
Nant Ceiswyn	1.999	0.572	1.00	39.81	14
Dulas (Dovey Forest)	0.191	0.225	1.00	6.92	15
Dulas (at Corris)	9.257	0.940	1.00	398.11	15
Afon Cerist	47.641	0.849	1.00	1202.26	28
Nant Iago	4.321	0.790	1.00	288.40	29
Afon Harnog	13.723	0.811	1.00	173.78	29
Nant Helgog	26.733	0.666	1.00	316.23	29
Mwyro	24.177	0.876	1.00	1047.1329	32

Table 7.5 *Pearson correlation coefficients (r) (and significance (p)) between faecal coliform parameters and land-use variables*

Variable	Geometric Mean	Log₁₀ Std Dev	Minimum	Maximum
Area	0.3750 (0.116)	0.4151 (0.079)	0.3713 (0.106)	0.3790 (0.101)
Rainfall	-0.5971* (0.016)	-0.0855 (0.391)	-0.6070* (0.014)	-0.5446* (0.027)
Forest	-0.1706 (0.289)	0.0944 (0.380)	-0.1498 (0.313)	-0.1434 (0.320)
Grade IV	0.9731* (0.000)	0.3198 (0.143)	0.9928* (0.000)	0.9559* (0.000)
Grade V	0.3720 (0.105)	0.5766* (0.020)	0.3609 (0.113)	0.3916 (0.093)
% Forest	-0.3202 (0.143)	-0.3276 (0.105)	-0.2756 (0.181)	-0.3015 (0.158)
% Grade IV	0.9868 (0.000)	0.3359 (0.131)	0.9756* (0.000)	0.9742* (0.000)
% Grade V	-0.0700 (0.410)	0.2623 (0.193)	-0.1109 (0.359)	-0.0839 (0.393)
NPK	0.8758* (0.000)	0.2846 (0.173)	0.9457* (0.000)	0.8507* (0.000)
Limestone	0.5840* (0.018)	0.4696 (0.053)	0.5789* (0.019)	0.5999* (0.015)
Breeding ewes	0.8718* (0.000)	0.5068* (0.039)	0.7964* (0.001)	0.8978* (0.000)
Cattle	0.8551* (0.000)	0.2485 (0.207)	0.9382* (0.000)	0.8233* (0.000)
Total animals	0.9166* (0.000)	0.4908* (0.044)	0.8638* (0.000)	0.9342* (0.000)
Ewe density	0.4270 (0.073)	0.3596 (0.144)	0.2735 (0.183)	0.4735 (0.051)
Cattle density	0.8164* (0.000)	0.2808 (0.176)	0.8930* (0.000)	0.7956* (0.001)
Animals density	0.5087* (0.038)	0.3799 (0.100)	0.3700 (0.107)	0.5510* (0.025)

n = 23 in all cases   * significant at α < 0.05

The results of faecal coliform analysis are summarised in Table 7.4. High geometric mean concentrations (>100 FC.100 ml⁻¹) are associated with those catchments with relatively high proportions of Grade IV land (13% - 45%). Low geometric mean concentrations (<10 FC.100 ml⁻¹) occur in catchments with relatively high proportions of forested land (>45%).

Moderate geometric mean concentrations (10 - 100 FC.100 ml⁻¹) appear to be associated with those catchments showing high proportions of Grade V land (>70%). This pattern also reflects the stocking density in the catchments. The low geometric means are associated with forested catchments, where the sole input of faecal coliform organisms is from the resident wildlife populations. In the agricultural catchments,

particularly those supporting relatively high numbers of cattle (i.e. those with a high proportion of Grade IV land), the agricultural livestock represents the most significant source of faecal coliform organisms.

Relationships between the geometric mean, log₁₀ standard deviation, minimum and maximum counts and land-use parameters were analysed using Pearson product moment correlation. The resultant correlation coefficients (r) and their corresponding significance levels (p) are shown in Table 7.5. Strong positive correlations are evident between the geometric mean faecal coliform concentrations and the amount of Grade IV land, stock numbers, cattle density and NPK fertilizer use (p 0.001). The relationship with limestone application was also positive and

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Table 7.6 *Percentage land cover types for 12 UK catchments derived from the ITE 1978 32 land class system.*

Catchment	Built up	Coniferous forest	Broadleaf forest	Misc. natural woodland	Moorland	Bog
Monachyle	0.15	5.9	0.75	3.94	31.64	20.72
River Exe	9.25	8.45	6.63	0.7	4.64	0.01
River Axe	15.34	1.5	13.95	0.68	0.02	0
Irfon	1.57	2.07	6.45	1.04	8.73	0.07
Lugg	3.84	2.03	5.91	1.22	10.86	0.62
E.Cleddau	21.94	9.57	5.03	1.19	2.64	0.01
W.Cleddau	10.72	14.19	5.94	0.55	2.74	0
Teifi	8.69	2.66	5.77	0.89	4.56	0.02
Glaslyn	3.42	2.95	5.27	3.57	11.51	1.02
Dwyfor	4.25	2.09	5.83	1.69	7.61	0.16
Aled	1.85	2.25	6.29	1.48	8.68	0.14
Duddon	5.47	3.92	2.05	7.11	9.29	5.58

Catchment	Heathland	Upland grasses	Permanent grassland	Leys	Under cultivation
Monachyle	27.87	8.04	0.81	0	0
River Exe	0.92	11.22	29.77	19.27	8.29
River Axe	0	2.31	14.15	27.06	22.38
Irfon	2.06	16.47	40.91	16.69	3.73
Lugg	1.63	14.99	33.12	15	10.28
E.Cleddau	0.55	8.29	26.89	16.52	6.66
W.Cleddau	0.52	9.43	24.11	20.79	9.88
Teifi	0.8	16.58	40.81	13.48	4.3
Glaslyn	4.38	16.51	32.39	15.41	3.34
Dwyfor	2.24	16.32	39.07	15.99	3.56
Aled	2.49	16.38	39.92	16.71	3.6
Duddon	21.02	11.27	16.98	16.17	1.11

significant ($p<0.02$). Similar patterns are evident for minimum and maximum faecal coliform concentrations. The $\log_{10}$ standard deviation value was used as an index of the variance in faecal coliform counts. This variable showed weaker, significant ($p<0.05$), positive correlations with the amount of grade V land and number of breeding ewes.

Conclusions

The analysis of faecal coliform concentrations in 13 upland Welsh catchments shows a consistent pattern with land-use in the catchments. Catchments with higher proportions of improved Grade IV agricultural land, with higher fertilizer use and livestock densities, produce higher geometric mean faecal coliform counts than forested upland catchments. This will tend to reflect the higher loading of faecal coliform organisms from livestock in agricultural catchments.

7.2 ITE land-cover classification study

The second of these studies examined relationships between faecal coliform concentrations in 12 UK catchments and ITE land-cover classification.

Materials and methods

The ITE 1978 land classification uses a 1 km grid square and is organised into 32 classes, each comprising proportions of 11 land-cover types. The percentages of the 11 land-cover types were calculated from the land class data for each catchment (Table 7.6). The land-cover types might then be categorised according to whether they are likely to represent significant source areas for faecal coliforms i.e. those expected to be subject to high stocking densities or faecal waste application practices.

The 12 catchments used in this study were selected because they combined several attributes such as good water quality data and flow data collected at the same point in the river network. They also represent a broad range in area, hydrology and bacteriological water quality (Tables 7.7 and 7.8).

The data used in these analyses were derived from routine sampling programmes undertaken by the National Rivers Authority, the Water

Table 7.7 Catchment area, mean annual rainfall, mean flow and area under cultivation for 12 UK catchments

Number	Catchment	Area (km²)	Rainfall (mm)	Mean flow (m³.sec⁻¹)	Area under cultivation (km²)
1	Monachyle	7.7	2734	0.49	0
2	River Exe	600.9	1270	15.79	49.8146
3	River Axe	288.5	999	4.93	64.5663
4	Irfon	72.8	1815	3.2	2.7154
5	Lugg	203.3	1022	3.9	20.8992
6	E.Cleddau	183.1	1441	5.98	12.1945
7	W.Cleddau	197.6	1293	5.38	19.5229
8	Teifi	893.6	1349	28.3	38.4248
9	Afon Glaslyn	68.6	3097	5.77	2.2912
10	Dwyfor	52.4	2092	2.52	1.8654
11	Aled	11.6	1363	0.16	0.4176
12	River Duddon	47.9	2174	3.15	0.5317

Table 7.8 Summary statistics for raw and log transformed faecal coliform data for 12 UK catchments for the period 1988 to 1991

Catchment	N	N log₁₀ FC	Maximum	Arithmetic Mean	Geometric Mean	Std Dev	Log₁₀Std Dev	Skewness	Log₁₀ Skewness
Monachyle	42	34	920	57.76	6.81	187.03	7.50428	4.16214	1.02043
River Exe	85	85	39000	3626.47	2038.89	6146.17	2.70692	4.45821	0.32805
River Axe	106	105	52000	4517.26	1794.62	8852.08	3.46242	3.52007	0.69146
Irfon	118	117	2300	186.36	102.94	272.18	3.4976	5.45184	-0.97039
Lugg	66	62	2000	577.42	452.1	509.68	2.21371	1.38223	0.16077
E.Cleddau	174	169	5000	845.93	552.67	870.39	2.64697	1.79982	0.10775
W.Cleddau	167	164	9700	887.37	497.67	1136.26	3.47164	3.89617	-1.22236
Teifi	840	828	9500	1191.73	716.07	1348.18	3.07006	2.51802	-0.88101
Glaslyn	25	23	410	25.12	6.27	81.21	4.23008	4.80714	1.12496
Dwyfor	48	46	2900	302.26	125.08	463.9	5.41819	4.01661	-1.05529
Aled	15	15	1380	234.47	58.02	394.3	7.04663	2.27455	-0.11434
R. Duddon	76	52	1650	81.3	23.97	242.17	6.85149	5.02336	0.0056

Companies and IH-Scotland. To permit comparison of data from different sources a detailed questionnaire was sent to each of the laboratories responsible for microbial analyses. This established the use of common analytical standards and practices complying with Report 71 (H.M.S.O., 1983).

Mean faecal coliform concentrations were calculated for each site for a four-year period of record between 1988 and 1991 (Table 7.8), multiple scatter plots and multiple regression tables were produced for faecal coliform values and each individual land class, the actual area of each land class in the catchment, rainfall, flow and catchment area.

The initial tests highlighted correlations between certain variables. Visual examination of scatter plots of the correlated variables showed that the land cover and faecal coliform data were skewed towards the origin. This clumping towards the origin was overcome by log₁₀ and square root transformations of the data. This did not necessarily improve correlations but led to a more

random scatter of the residuals to the model fits (Figure 7.1).

Multiple scatter plot representation of the data demonstrated the multicolinearity of the land-use variables and prompted the lumping of the data into broad categories perceived to relate loosely to upland and lowland areas. The upland group (B) was negatively correlated to faecal coliform parameters, the lowland group (A) was positively correlated. The results of the analyses are presented below.

Results

Multiple scatter plots and regression analysis highlighted both positive and negative relationships between percentage land types and faecal coliform values (Table 7.9). The scatter plots proved to be useful in identifying relationships not apparent in the regression tables as a result of outliers, non-normality and skewness in the data.

The most significant relationship found was that between maximum and mean faecal coliform

Table 7.9 Pearson correlation coefficients (r) (and significance (p)) between faecal coliform parameters, percentage land-cover types and land-cover areas

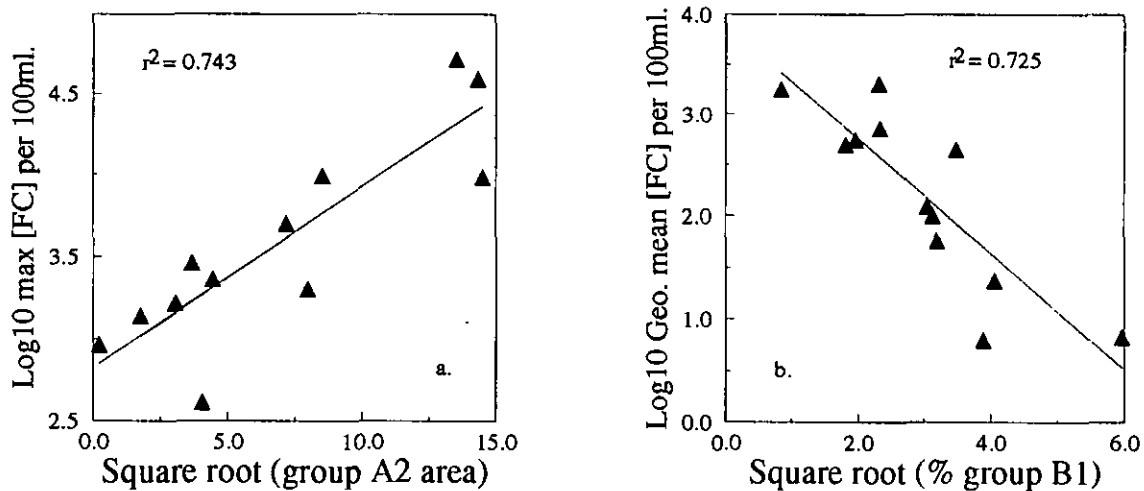| Variable	Geometric mean	Log$_{10}$ Std Dev	Maximum
Area	0.6305 (0.0279)	-0.5633 (0.0565)	0.4586 (0.1338)
Rainfall	-0.6023 (0.0382)	0.5540 (0.0616)	-0.5002 (0.0977)
Flow	0.5469 (0.1353)	-0.4913 (0.1048)	0.2824 (0.3739)
% Built	0.5497 (0.0641)	-0.5488 (0.0647)	0.4879 (0.1076)
[x Area / 100]	0.7364 (0.0063)	-0.5893 (0.0438)	0.5979 (0.0400)
% Conifers	0.1734 (0.5900)	-0.2038 (0.5251)	0.0559 (0.8631)
[x Area / 100]	0.6805 (0.0149)	-0.5323 (0.0748)	0.4701 (0.1230)
% Broadleaf	0.6610 (0.0193)	-0.4951 (0.1017)	0.7607 (0.0041)
[x Area / 100]	0.8155 (0.0012)	-0.5702 (0.0529)	0.7315 (0.0069)
% Misc. natural	-0.4978 (0.0996)	0.6684 (0.0175)	-0.3972 (0.2011)
[x Area / 100]	0.3740 (0.2310)	-0.4185 (0.1757)	0.2133 (0.5056)
% Moors	-0.5069 (0.0926)	0.6154 (0.0331)	-0.4647 (0.1279)
[x Area / 100]	0.3643 (0.2444)	-0.5258 (0.0791)	0.1256 (0.6973)
% Bog	-0.3155 (0.3178)	0.6347 (0.0266)	-0.2416 (0.4494)
[x Area / 100]	0.3960 (0.2026)	0.4909 (0.1051)	-0.3571 (0.2545)
% Heath	-0.4197 (0.1743)	0.7470 (0.0055)	-0.3261 (0.3009)
[x Area / 100]	0.0134 (0.9671)	0.0734 (0.8207)	-0.0881 (0.7853)
% Upland grasses	-0.5203 (0.0829)	0.0416 (0.8979)	-0.6299 (0.0281)
[x Area / 100]	0.3613 (0.2485)	-0.4521 (0.1401)	0.1694 (0.5986)
% Permanent grasses	-0.1143 (0.7235)	-0.3632 (0.2459)	-0.2409 (0.4507)
[x Area / 100]	0.4316 (0.1612)	-0.4801 (0.1142)	0.2460 (0.4409)
% Leys	0.5585 (0.0591)	-0.4980 (0.0994)	0.6129 (0.0341)
[x Area / 100]	0.8290 (0.0009)	-0.6072 (0.0363)	0.6967 (0.0118)
% Cultivated	0.7356 (0.0064)	-0.5500 (0.0639)	0.8050 (0.0016)
[x Area / 100]	0.9425 (0.0001)	-0.5896 (0.0436)	0.9130 (0.0001)

Figure 7.1 Plots of faecal coliform parameters against positively (a) and negatively (b) correlated groupings of land-cover descriptors in 12 UK catchments

Table 7.10 *Correlation coefficients (r²) (and significance (p)) between transformed faecal coliform parameters and percentage land-cover and land-cover areas*

Land-cover variable (Skewness)	Faecal coliform value (Skewness)	N	r² (p, where p > 0.0001)	Outlier
Percentage land cover results				
%heath (1.2358)	l_m_fc (-0.6110)	10	0.8646	1,12
%heath (1.2358)	l_g_fc (-0.9978)	10	0.8948	1,12
s_%heath (-0.2407)	l_g_fc (-0.9978)	10	0.8301	1,12
%misc_nat (1.6934)	l_g_fc (-0.7281)	11	0.8341	12
s_%miscnat (1.3658)	l_g_fc (-0.7281)	11	0.8527	12
% moors (2.4098)	l_max_fc (0.5331)	11	0.7533 (0.0005)	1
s_%group B1 (0.7817)	l_g_fc (-0.4616)	12	0.7251 (0.0004)	-ᵇ
s_%group B2 (1.6754)	l_g_fc (-0.6029)	11	0.8022 (0.0002)	9
s_%group B1 (0.9905)	l_max_fc (0.3981)	11	0.7483 (0.0006)	1
Land-cover area results				
l_broad (-1.0140)	l_g_fc (-0.6029)	11	0.9229	9
s_broad (0.8420)	s_g_fc (0.9381)	11	0.9393	8
s_broad (0.6792)	l_max_fc (0.7824)	9	0.9830	5,8,9
cultiv (0.7179)	g_fc (1.4583)	11	0.8882	9
cultiv (0.7179)	s_m_fc (1.1550)	11	0.9232	9
l_cultiv (-0.6861)	l_g_fc (-0.6029)	10	0.9319	9
s_cultiv (0.1365)	l_m_fc (0.0641)	11	0.9091	9
s_cultiv (0.1365)	s_g_fc (0.6819)	11	0.9305	9
s_cultiv (0.4714)	l_max_fc (0.7824)	9	0.9886	5,8,9
s_leys (0.6604)	s_g_fc (0.9381)	11	0.9069	8
s_leys (0.5052)	l_max_fc (0.7824)	9	0.9419	5,8,9
group A1 (1.2316)	g_fc (1.6163)	11	0.9857	8
s_group A2 (0.3876)	s_g_fc (0.7567)	11	0.8518	-
s_group A2 (0.3876)	l_max_fc (0.5331)	12	0.7436 (0.0003)	-ᵃ
s_group A2 (0.5003)	l_max_fc (0.7824)	9	0.9807	5,8,9

Key:- The prefixes used in this table refer to the type of mean and the method of transformation used to improve the skewness of the raw data; m, arithmetic mean, g, geometric mean, l, $\log_{10}$ transformation, s, square root transformation. ᵃᵇ(see Figure 7.1).

Land-cover groupings:-
Group A1 = cultivation + leys + broadleaf woodland + built-up area (km²)
Group A2 = cultivation + leys + broadleaf woodland (km²)
Group B1 = moorland + miscellaneous natural woodland (% of catchment area)
Group B2 = moorland + miscellaneous natural woodland + heathland + bogs (% of catchment area)

concentrations and percentage of cultivated land. These results were considerably improved by conversion to the actual cultivated area in each catchment (*r>0.9, p=0.0001*) implying that some aspect of cultivated land management generates a significant number of faecal coliforms.

This result prompted a similar conversion of the other land-cover types but only produced improvements in fit with certain variables (Table 7.9). Correlations with variables which, as percentage values, were negatively related became more scattered, e.g. miscellaneous natural woodland, moors, bog, heath etc. Examination of scatter plots for the improved

correlations demonstrated the colinearity of a number of the variables, as previously mentioned. These were grouped in different orders and tested against the faecal coliform values. The optimum grouping (group A2) was found to be the sum of broadleaf woodland, leys and cultivated area (Table 7.10, Figure 7.1).

Percentage land-cover variables negatively correlated with faecal coliforms were also grouped. The sum of moorland and miscellaneous natural woodland (group B1) gave the best fit to the $\log_{10}$ converted geometric mean faecal coliform values, following square root conversion (*r² = 0.7251, p = 0.0004*) with no outliers.

Log_{10} transformation of the faecal coliform means and maximum values led to an improvement in fits to both percentage and area land-cover values. Square root transformation of the land-cover areas improved the normality of that data and also the fits to faecal coliform values (Table 7.10).

Figure 7.1 shows the two most significant relationships between faecal coliform values and group A2 and group B1 variables. These results are highly significant and demonstrate that the appropriate grouping of potential causative variables and non-linear transformations can eliminate the problems caused by outliers, non-normality and skewness in the data.

Discussion

The positive relationships with the group A2 area variables suggest that the actual area contributing organisms within a catchment is more significant than the percentage coverage of each land type. This result is, in part, attributable to catchment area although catchment area itself is not a sufficient predictor of faecal coliform concentration (Table 7.9). It is likely that larger catchments will have greater proportions of agricultural land and hence livestock concentrations as well as sewage and farm waste application practices, thus deriving greater faecal coliform concentrations.

The relationship with group B1 variables indicates that non-agricultural land-uses derive low faecal coliform concentrations, the faecal coliform numbers derived from such areas being caused by wildlife, as suggested in Section 7.1. Table 7.3 shows %Forest and %Grade V land to be negatively correlated and unrelated, respectively, to animal stocking variables in the 13 Welsh upland catchments.

There is a similarity between this and the Group A2 results in that they relate to catchment size or scale. Clearly this might not be the case for a broader sample of catchments where headwater areas may be largely agricultural.

Having established these land-use relationships it is important to consider the physical interpretation of the results. Catchments with larger proportions of poor/upland land areas have lower faecal coliform concentrations. In larger catchments where valleys and floodplains are likely to have intensive agricultural uses (Grade IV land; Table 7.3, cultivated area; Table 7.9, 7.10). In terms of modelling non-point source contamination these results might be considered in terms of travel times within the catchment and the implications for die-off and retention mechanisms.

Kunkle (1970), studying upland catchments with permeable soils, found that bacterial contributions from areas away from stream margins were small compared to those derived near-channel and grazing had minimal impact when carried out away from the stream margins. Upland areas contributed little or no overland flow during storms, most storm run-off originating on saturated areas which built-up along channel edges. It was suggested, therefore, that due to the run-off processes, bacterial contamination is probably more a function of activities in and around the stream channel than of basin-wide land-use. Similarly, Hunter and McDonald (1991) found that moist areas, where overland flow was preferentially generated, contributed significantly higher numbers of faecal bacteria to surface waters than dry areas.

This might explain why the square root of land-cover area gives a straight line fit to log_{10} maximum and log_{10} geometric mean faecal coliform concentrations. Faecal coliform concentrations are known to be log-normally distributed; log_{10} conversion normalises their distribution. Square rooting the land-cover area produces a length which may represent the length of stream channel passing through the land-use area of interest. Each bank represents the termination of flow pathways draining the catchment, carrying organisms from contaminated areas. The speed of the flow path and the distance to the channel will determine the proportion of organisms initially undergoing transport that actually survive to enter the channel.

Future studies should examine travel times within catchments in conjunction with an assessment of the current spatial distribution of contaminative land-cover. The development of isochrone maps combined with overlays of contaminative land-use cover would allow the catchment planner to assess current land-use impacts and determine appropriate land-use strategies using risk maps based on bandings of % loss due to die-off and retention processes.

The expansion of the sample of sites analysed and the availability of enhanced land-use information in conjunction with pursuit of the suggestions made above will extend this work and improve confidence in the relationships derived. The Institute of Terrestrial Ecology has recently developed a 17 class land-cover map for the UK with 25 metre resolution. This comprises a total of 25 sub-classes and differentiates between land types, i.e. rough pasture, pasture, meadow, tilled land etc., and has good potential for building on the results already presented.

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8 Conclusions and suggestions for further work

1. The key influences on the survival of faecal coliforms in streams and rivers are light and turbidity, temperature and pH. Faecal coliform survival is influenced by a wide range of other interactions and mechanisms; of these, competition from — and predation by — the natural microbial community, particle interactions and nutrient effects are the most important. The self purification ability of a water is also a function of water quality. In poor quality or sewage-contaminated waters the purifying effect of sunlight and the influence of temperature changes are diminished.

These die-off effects are reflected in the seasonality observed in long time series of faecal coliform concentrations. For example, in winter fewer daylight hours, lower temperatures, a more moist catchment surface, shorter residence times in river reaches, protection from light and predation by particulates, results in enhanced survival. Supply and transport factors also tend to result in higher winter concentration, more rapid transport from the catchment surface, frequent operation of storm sewage overflows and scouring of settled organisms. In the summer months these effects are reversed: die-off throughout the catchment is enhanced and low flows result in minimal transport within the catchment. This is not, however, always the case. Where the rate of input of organisms to a stream is very high, die-off effects will be reduced and dilution will cause the greatest observed change in bacterial concentration.

2. Examination of the key processes of faecal coliform transport within catchments demonstrated how the significance of different processes and sources of faecal contamination change with location. In headwater areas the supply of organisms is dominated by non-point sources; organisms are transported from the catchment surface by a combination of surface run-off and non-matrix through-flow in the subsurface zone during rainfall events. Further downstream the emphasis changes, point sources and channel storage interactions becoming more significant to the supply of contaminative organisms.

3. Previous models for faecal coliform dynamics used a range of approaches. Multivariate statistical approaches relate the bacterial concentration to a number of driving variables using simple statistical relationships. Simple deterministic first order decay functions have been used to describe exponential die-off of a bacterial population and in application to rivers have been combined with equations to describe fluid mixing processes and flow hydraulics. These models all lacked the necessary structure to describe the process of bacterial transport in rivers adequately. The model of Jenkins (1984) sought to describe the transfer of organisms to and from storage within the channel.

Field experiments carried out during the project provided information to improve upon the results of Jenkins. Although the eventual model structure is similar, fewer parameters are required.

4. The model structure and operation incorporates the following assumptions:

- the channel-store is distributed across the entire channel;

- the regions of storage respond sequentially to rises in flow;

- any given rise in flow will produce entrainment of organisms from the channel;

- at any quasi-steady flow, the active supply area of organisms will become depleted.

Entrainment is assumed to cease once the flow recession commences. Further higher flows will still release organisms from storage.

The model incorporates terms for the effects of sunlight and turbidity, temperature and pH on faecal coliform survival in the water column. These were derived from data in the literature.

5. The model was successfully applied to a reach of the River Exe in Devon for the years 1990 and 1991. The model was seen to operate well for extended periods of data, the numbers of organisms in the channel store remained stable and were, in effect, self regulating. Seasonal effects were modelled with a simple cosine function accounting for die-off changes resulting from solar radiation and temperature, overcoming the need for data for these variables and reducing the number of parameters needed to calibrate the model. Previous models have not given a satisfactory description of faecal coliform river

dynamics. The model applied here gave a good fit to the observed data and has scope for application to other water quality determinants.

6. Further development of the model might consider extending its scope to modelling particulates or other particle associated contaminants such as metals, hydrocarbons or radio-nuclides, and other microbiological determinands such as faecal streptococci. Future field-programmes might seek a more intensive application of the model for further validation with detailed monitoring of all relevant variables over a period of several months.

7. Consistent patterns between land-use and faecal coliform concentrations in 25 UK catchments were found. Faecal coliform concentrations in 13 upland Welsh catchments and data from ADAS land-use maps and derived from surveys of stocking practices and fertilizer use were analysed. Catchments with higher proportions of improved agricultural land, with higher fertilizer use and livestock densities, were found to produce higher geometric mean faecal

coliform concentrations than forested catchments. This finding reflects the higher loadings of organisms from livestock in agricultural catchments.

A further 12 catchments in England, Scotland and Wales representing a broad range of size, land-use and faecal contamination were examined. The results were consistent with the Welsh study and confirmed that agricultural land classes and groupings of classes perhaps relating to the lowland nature of the catchment produce more faecal coliforms than more upland catchments with non-agricultural land-uses.

Further studies should examine the relationships between faecal coliform concentrations and the more recent ITE land-use classification system which differentiates between grasslands used for pasture or rough grazing, etc. Combined with the analysis of a greater number of sites, study of travel-time effects and the spatial distribution of land-uses, this would represent a valuable enhancement of the results already presented here.

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