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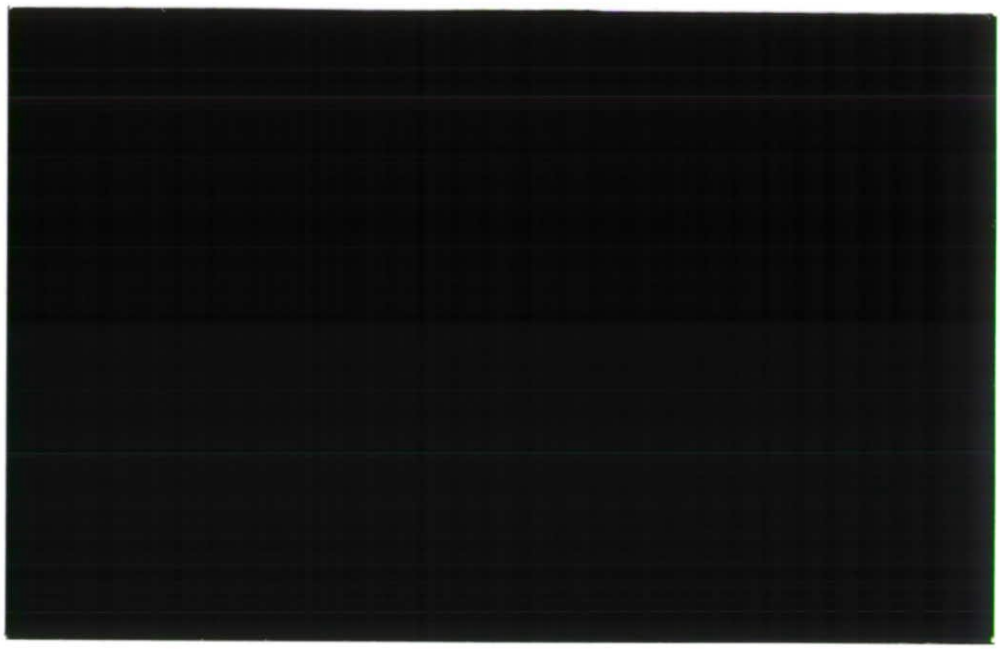

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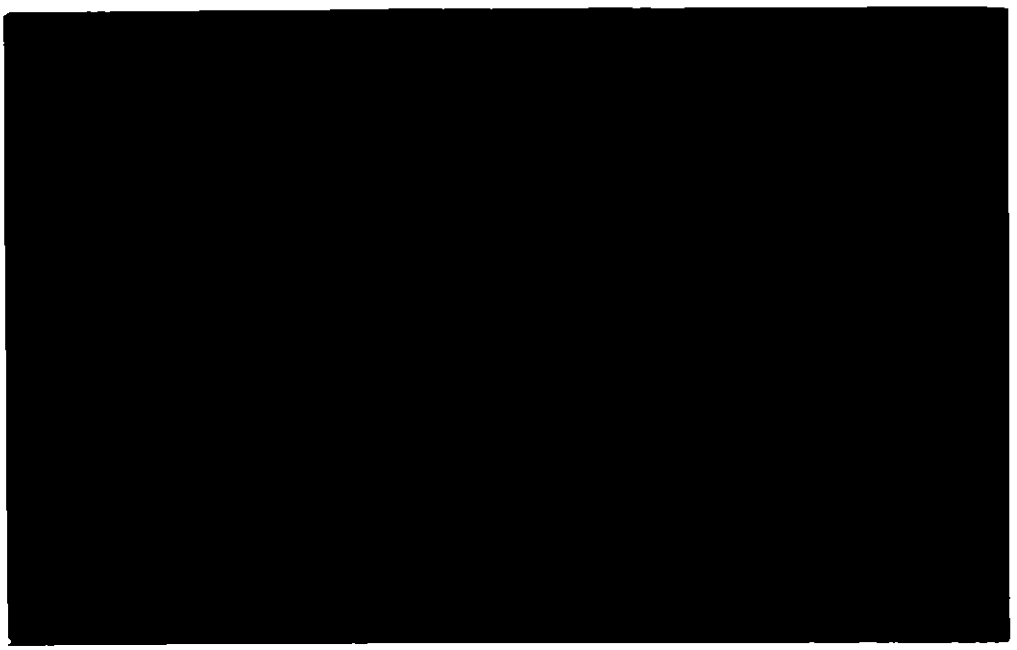

MODELLING

FAECAL COLIFORM CONCENTRATIONS

IN STREAMS

By Jeremy Wilkinson, Alan Jenkins,
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PECD Reference No. 7/7/385

Final Report

Project Duration: 1 July 1991 to 30 June 1994

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Report Date: January 1995

i. Forward

This project was funded by the Department of Environment, Water Directorate, under project number PECD Ref. No. 7/7/385. It was started in 1 July 1991 and completed at the end of June 1994. Other contributions to the project were made by the National Rivers Authority who funded an expansion of the data collection elements of the programme and the Natural Environment Research Council. The project was undertaken by the Institute of Hydrology in conjunction with the Centre for Research into Environment and Health, University of Leeds.

Acknowledgements:- The NRA Regions or PLC's who provided data; South West, Welsh, Thames, Anglian, Northumbrian, Severn-Trent NRA's, North West Water Plc. and South West Water Plc. Simon Durbin of Powergen's Cwm Rheidol Power Station, Tim Harrison of Severn-Trent NRA and Peter Joyce of Yorkshire Water Plc. for their kind assistance in making the experimental flow releases possible. Dr Lorna Fewtrell and the staff at the Centre for Research on Environment and Health, University of Leeds, and staff from Acer Environmental played a key role in the field and microbiological aspects of the work. Also Dr John stoner of the NRA for his collaboration with David Kay in the study of 13 upland Welsh catchments.

ii. Executive Summary

This is the final report to DoE on contract PECD 7/7/385 "Modelling *E.coli* in streams" awarded to the Institute of Hydrology 1991-1994. The stated objectives 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; and to develop an integrated predictive model of faecal coliform concentrations from point and non-point sources". The work can be integrated with research in areas seeking to describe bacterial water quality and to assess the risks associated with exposure to waters containing faecal contamination.

The above objectives have been met by a combined approach including a literature survey, field experimentation, model development and validation using both field and existing databases, and the examination of national GIS databases in assessing land-use impacts.

1. Examination of the key influences on the survival of faecal coliforms in streams and rivers demonstrated that these 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. 90% reduction of an initial population in a few hours might be expected in bright sunlight, in darkness the organisms may persist for many days. The effect of solar radiation is reduced in turbid waters where 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. In sewage contaminated or oxygen stressed waters faecal coliform survival is extended.

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 throughflow 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.

2. 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 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 Section 6 of this report 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: 1. The channel-store is distributed across the entire channel. 2. The regions of storage respond sequentially to rises in flow. 3. Any given rise in flow will produce entrainment of organisms from the channel. 4. At any quasi-steady flow the active supply area of organisms will become depleted. 5. 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 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. 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 it also has scope for application to other water quality determinants.

3. Analysis of faecal coliform concentrations in 13 upland Welsh catchments and data from ADAS land use maps and 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 domonstrated 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 landuses. The results indicate the importance of near channel areas as delivery fronts from faecal coliform supply areas within the catchment.

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 and study of travel-time effects, this would represent valuable enhancement of the 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 scope of the model includes assisting in 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 of local bathing waters; and to assist drinking water abstractors predict the timing, duration and magnitude of events of peak bacterial concentration in order to prevent the intake of large loadings of faecal contaminants.

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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; and to develop an integrated predictive model of faecal coliform concentrations from point and non-point sources". The work can be integrated with research in areas 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 contained 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; to assist in the assessment of changes in effluent discharges or land-use on, for example, the health risk posed to recreators by a given river reach; assess the loadings of faecal contaminants to the marine systems and hence the impact on compliance of local bathing waters; and to assist drinking water abstractors predict the timing, duration and magnitude of events of peak bacterial concentration in order to prevent the intake of large loadings of faecal contaminants.

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 comprises: *Escherischia*, *Klebsiella*, *Enterobacter* and *Citrobacter* types, however, not all are associated with faecal contamination; *Citrobacter* is not a faecal coliform. The faecal coliform index, which is used for 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 and surface waters inputs may be categorised as either point or non-point sources (Table 1.1).

Non-point or diffuse source inputs of faecal contamination are essentially derived from the catchment surface representing 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 landuse; 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, such as piped discharges. Piped discharges include leachate from land drainage systems, sewage works outfalls, combined sewer storm overflows or industrial discharges.

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 (examples, 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 dilution by receiving water can be achieved. (All values per 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].	10-10^3 per 100ml

Other obscure or less likely 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 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).

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2. RELATIONSHIPS BETWEEN ENVIRONMENTAL CONDITIONS, WATER QUALITY AND FAECAL COLIFORM MORTALITY

The key influences on the survival of faecal coliforms in streams and rivers are light and turbidity, temperature and pH. Faecal coliform survival depends on a wide range of interactions and mechanisms; competition from and predation by the natural microbial community, particle interactions and nutrient effects. This section attempts to draw together the wealth of experience gained in this field over many years to quantify some of the major influences on faecal coliform survival in the form of general equations which can be used in a mathematical model, identify apparent weaknesses in past research and suggest areas of study that would benefit our knowledge of the subject.

Sunlight is probably the most important single driving variable with regard to faecal coliform die-off in streams and rivers. T_{90}s in the order of a few hours might be expected in bright sunlight, in darkness the organisms may persist for many days. The effect of solar radiation is reduced in turbid waters where light penetration is reduced and the organisms are shielded by an envelope of small particles (see Section 2.8).

Faecal coliform die-off increases with temperature, as a result of enhanced metabolism and increasingly aggressive activity from the natural stream biota (see Section 2.5). Self purification is a function of water quality. In poor quality or sewage contaminated waters the ability to eliminate contaminative organisms is reduced. Similar results have been observed in low oxygen environments as might be expected under the effects of contamination. The effect of temperature change on die-off is also reduced in poor quality waters.

The relationship between pH and faecal coliform die-off can be represented by a simple hyperbolic cosine function, where, pH_{min} is the pH at which die-off is at a minimum (k_{min}) and die-off being more rapid at higher or lower values.

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 through-out the catchment is enhanced and low flows result in minimal transport within the catchment. Where the rate of input of organisms to a stream is very high, die-off effects will be swamped-out and dilution will cause the greatest observed change in bacterial concentration. This will effectively reverse the seasonal pattern suggested above and might be observed downstream of an effluent discharge.

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, 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. Understanding this highly complex range of interactions requires knowledge spanning many disciplines and is beyond the scope of this study.

Some of the studies reviewed in this section of the report use actual *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 it's initial size). 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, ie:

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

$$-k.t = \log_{10} N_0 - \log_{10} N_t$$

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

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 (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}, $\log_{10} N_0 - \log_{10} NT_{90} = 1$.

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

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 $T_{½}$ 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 $T_{½}$ values can be converted to die-off rate k;

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

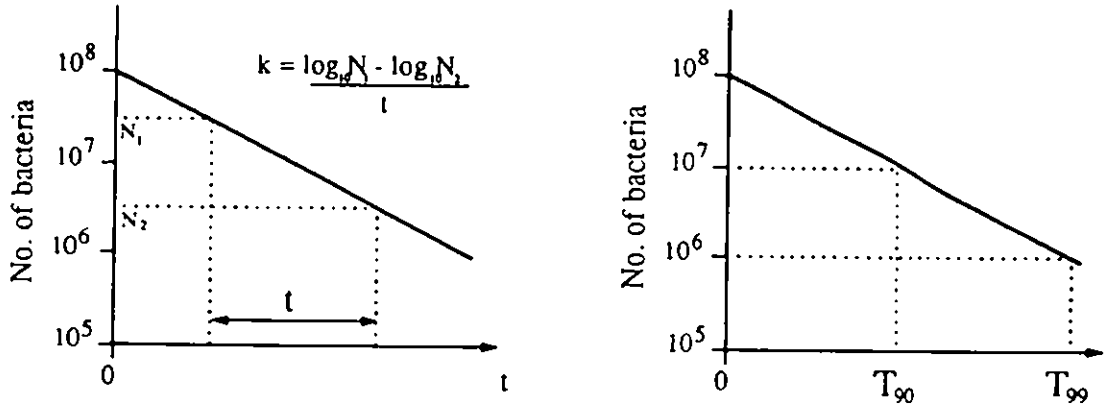

Figure 2.1 Graphical derivation of die-off rate k and 90% reduction time T_{90} assuming simple first order decay dynamics.

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), however, 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.

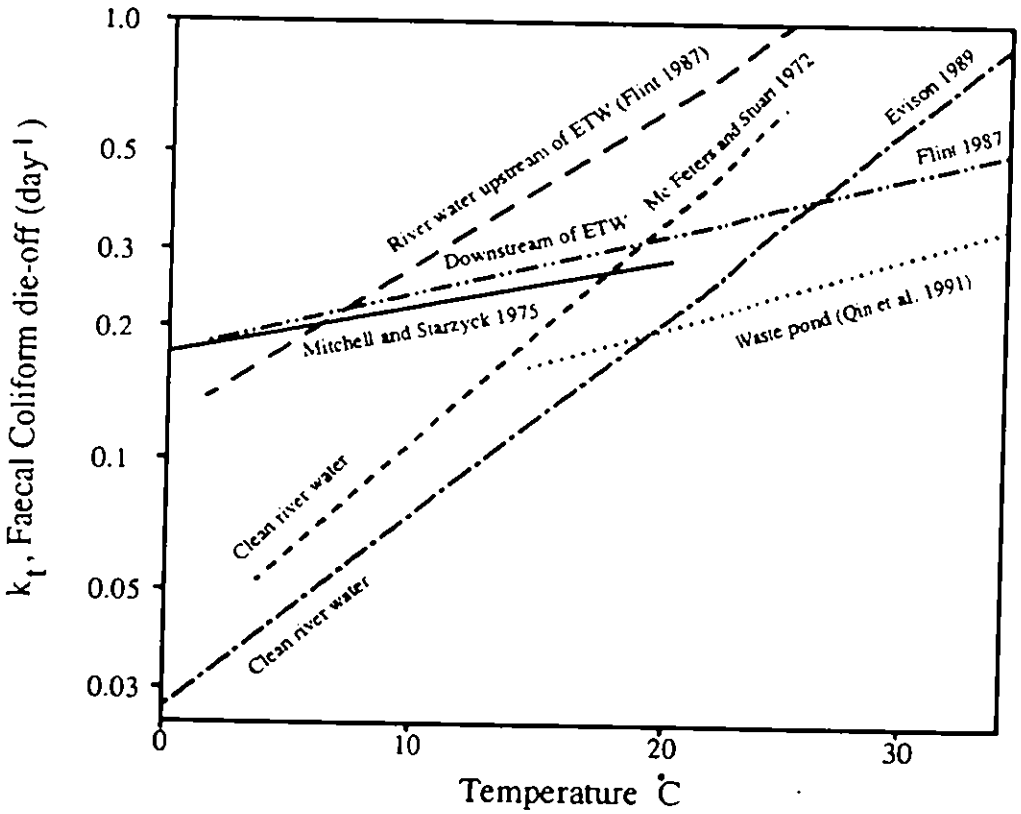

***Figure 2.2** Plots of faecal coliform die-off rate against water temperature.*

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

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

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_{T1} and k_{T2} are die-off at those temperatures. From the studies examined two distinct slope characteristics are apparent and appear to relate to clean and sewage contaminated waters. 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 calibration, the die-off coefficient will be set according to a range of influences, provided that an appropriate slope value is chosen the die-off rate will self-adjust appropriately (Table 2.2).

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

In general, faecal coliform die-off increases with temperature, this results 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.

2.3 The bactericidal effect of sunlight

The effect of sunlight on faecal coliform die-off is dramatic; T_{90} between 1-2 hours in bright sunlight and a number of days in darkness (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 2 hours in exposed containers (Kunkle and Meiman, 1968). The same effect might be expected on the catchment surface (Crane and Moore, 1986). Light has both direct and indirect effects, the direct effect being cell damage, indirectly the effect on temperature or moisture conditions In surface waters oxygen production and the excretion of algal toxins exerts a further stress on the contaminant organisms (Verstraete and Voets, 1972).

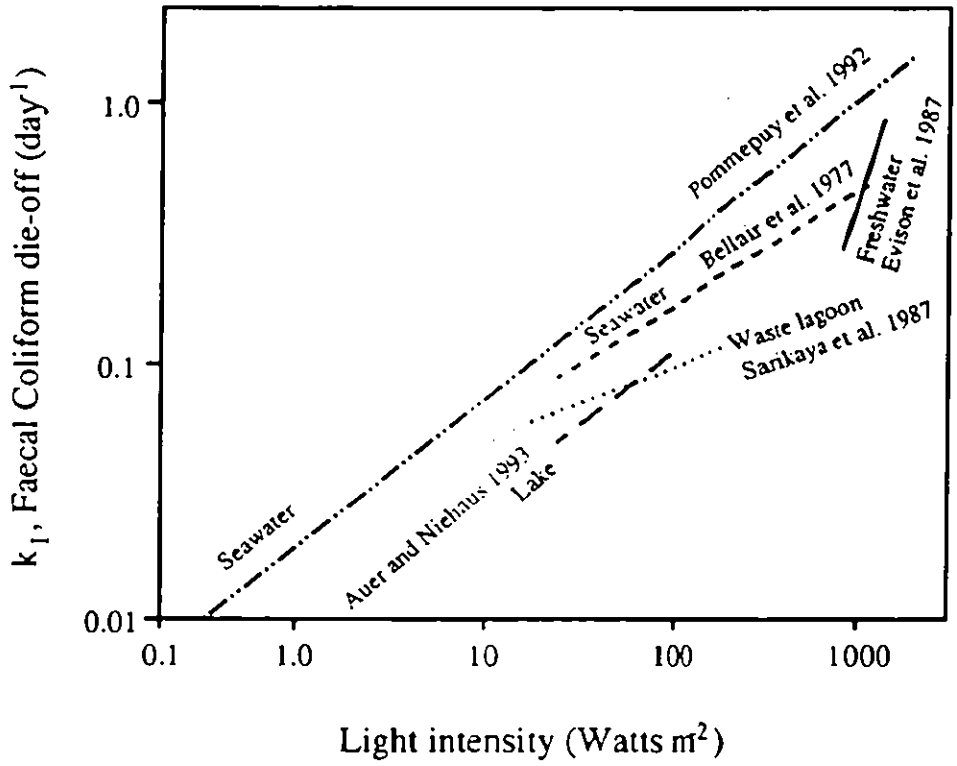

Figure 2.3 Plots of faecal coliform die-off rate against light intensity.

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. All 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).

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 to 50% of the observed *E.coli* variation. In clean waters, good light penetration will be possible leading to optimal die-off conditions; that is, the direct lethal effect of sunlight, adequate oxygen supply to the indigenous microfauna and production of toxins by algae. In poorer quality or perhaps deep waters, where light attenuation is significant, the benefits of sun-light will be reduced. Poor penetration, protection by adsorption to particulates and oxygen stress of the natural micro-community will result in extended

survival. On the catchment surface factors such as aspect, slope and vegetation cover will be important in determining the influence of sun-light on faecal coliform survival.

Table 2.3 Conversion factors for comparison of light intensity values.

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

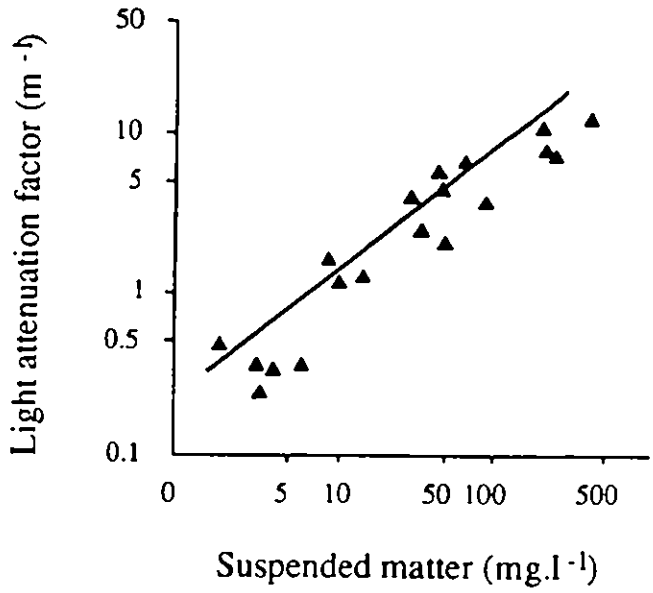

***Figure 2.4** Plot of light attenuation factor, η, per metre depth against suspended matter concentration (after Pommepuy et al. 1992).*

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_l.

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

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.

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

$I_{0,avg}$ 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;

$$\eta = 0.22\ SM^{0.78}$$

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

Solar radiation has a major impact on faecal coliform survival, in darkness the organisms may persist for many days, in bright sunlight, however, populations are destroyed in a number of hours. The effect of solar radiation is reduced in turbid waters where light penetration is reduced and the organisms are shield by an envelope of small particles (see Section 2.8).

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 al.*, 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.

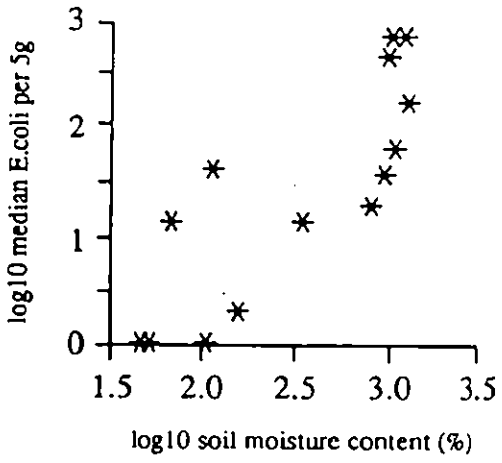

Figure 2.5 *Scatter plot of log_{10} median E.coli count (per 5g) against log_{10} soil moisture content (% of dry weight) for a catchment in the Yorkshire Dales (after Hunter and McDonald, 1991a).*

Moisture content and retention properties of the soil are in turn influenced by other factors which 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, 1991a).

Vegetation type and cover also influences soil moisture conditions. Dense sward, for example, can afford protection from adverse environmental conditions, limiting the effect of sunlight, wind and high temperatures (Chandler *et al.*, 1981; Zyman and Sorber, 1988). Waste application also influences moisture supply and retention. Liquid animal wastes have high moisture content and good moisture retention properties (Crane and Moore, 1986) and 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 1000mm. 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 100ml with mean dry matter content of 1.9%, the application rate approximated to a 30mm 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 6 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, by maintaining moisture levels, provide conditions suitable for extended survival.

T_{90} values of 3 days and 25 days have been observed in sandy and loam soils, respectively. These results are attributed to the higher 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) coat the outside of the much larger bacteria (a few microns in size) affording protection from microbial predators (Roper and Mitchell 1978) (see Section 2.8).

Faecal coliform survival is enhanced in moist soils and in locations which favour the continuity of cool moist conditions, ie. 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 significance of these effects may be lumped together with other model terms (for example Section 2.2, Table 2.2).

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 showing that different microbes dominate die-off in different environments.

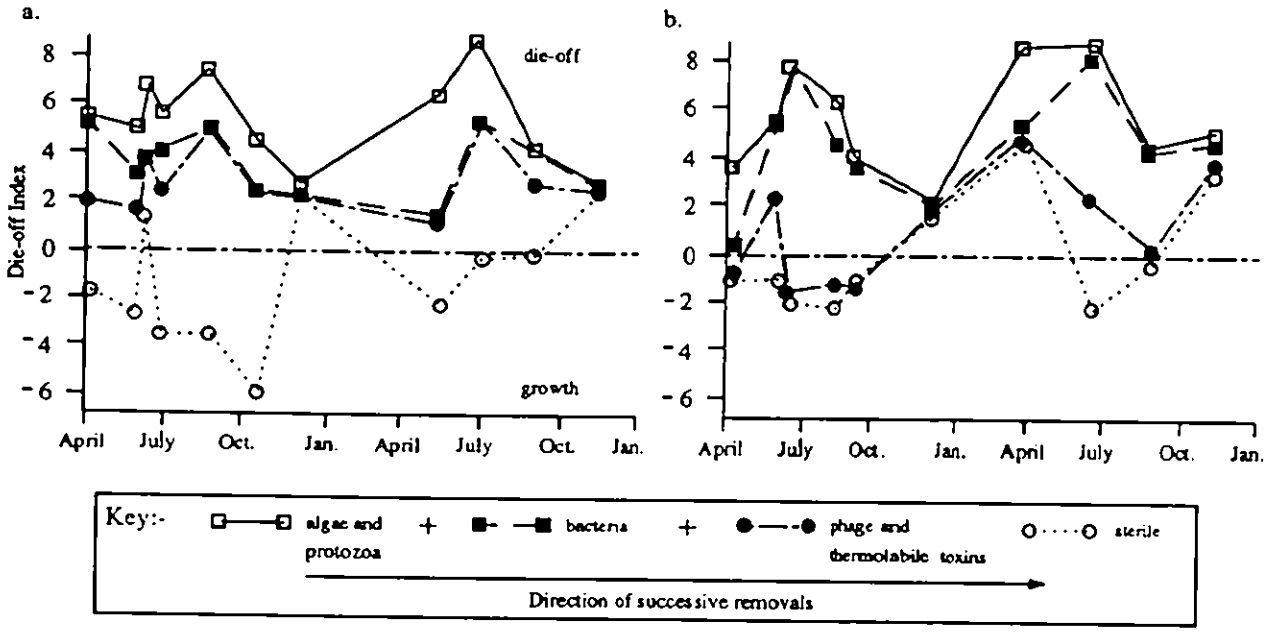

Figure 2.6 Die-off of E.coli in a. sewage contaminated and b. clean water bodies, following removal of; protozoa and algae (dashed line); protozoa, algae and bacteria (dot-dash); filtration at 0.45μm and autoclaving (dotted line); raw samples (continuous). After Verstraete and Voets (1972).

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 10 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 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

[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).

bacteriophages and thermolabile toxins, 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 identification of the nature and origin of the agents acting in the polluted water would require further research (Verstraete and Voets, 1972).

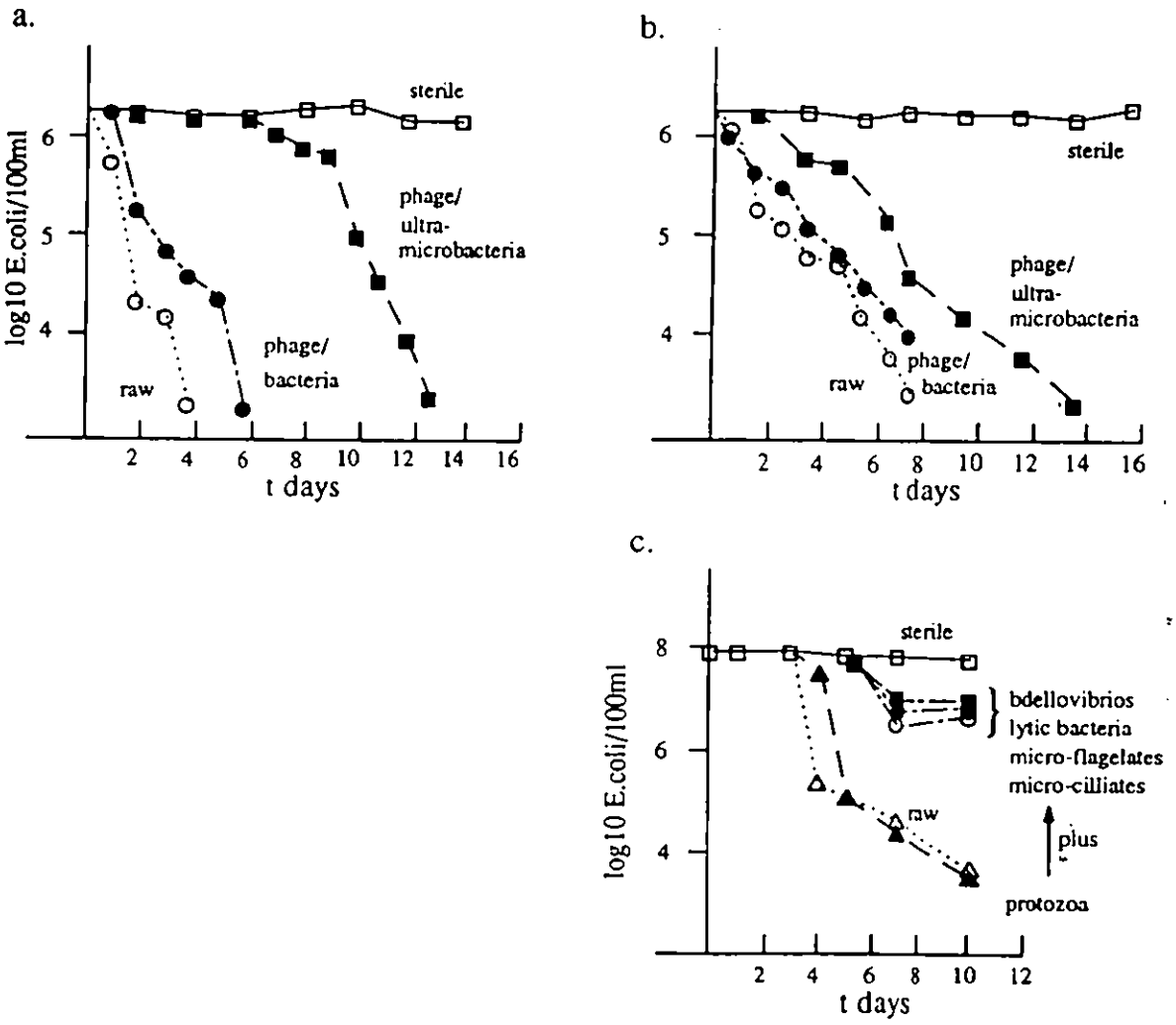

Figure 2.7 *The influence of successive removal of sections of natural microbial communities on E.coli die-off in a. "clean" river water, b. sewage contaminated river water (after Flint 1987) and c. estuarine water (after Enzinger and Cooper 1976).*

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 0.45µm Millipore membrane filters and finally by sterilisation in an autoclave. The filtrates and raw samples were then used in die-off experiments, results were 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 the raw and Whatman filtered samples which were 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).

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%)

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 dominance of different microbial purification processes under different environmental conditions.

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 anti-biotics. Anti-biotic 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 anti-biotics 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; 0.22µm, filtrates were generally free of natural organisms and *E.coli* numbers in these samples

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

remained near constant; 0.45µm, only bdellovibrios were present in filtrates causing a slight increase in die-off rate. Results for 0.8 and 1.2µm filtrates were similar, these contained *bdellovibrio* plus one or two other bacteria capable of *E.coli* destruction. The 1.2 and 3.0µm filtrates both contained numerous *micro*-ciliates and *micro*-flagellates. Only the 3µm filtrates contained protozoa and die-off was most rapid in these. The lag-time before 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 the *E.coli* cells remained free-living and their reduction was dominated by flagelates, this was followed by rotifers and *K.cochlearis* the coliforms now being associated largely with particles. After 10 days macrozooplankton of the *Daphnia* species began to dominate. Phages were found not to be relevant. An assessment was made of the optimal target size spectrum, flagellates prefer sizes of 0.8 to 1.2µm which includes free living *E.coli*. Rotifers may have grazed on free living bacteria and particles. *K.cochlearis* and *Conchillus unicornis* can graze on *E.coli* and particles smaller than 20µm and *Daphnia* particles of less than 35µm in size.

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 behaviour occurs on the catchment surface and sub-surface and in stream beds, however, the effects of the natural biota effects may be insignificant compared with other influences. The relative impact of different indigenous microbes on bacterial survival in these environments is not well documented.

Self purification is a function of water quality itself, these results suggest that the poorer the water quality the lower the ability to eliminate contaminative organisms. The introduction of effluents automatically alters the receiving water quality resulting in conditions which are favourable to extended faecal coliform survival.

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. Most studies have examined the effects of acid conditions. 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.100ml^{-1} were observed in 2 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, pH2.9-4.5, die-off within a few

days, as opposed to several weeks in limestone soil, was observed. It is suggested that low pH not only affects 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).

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).

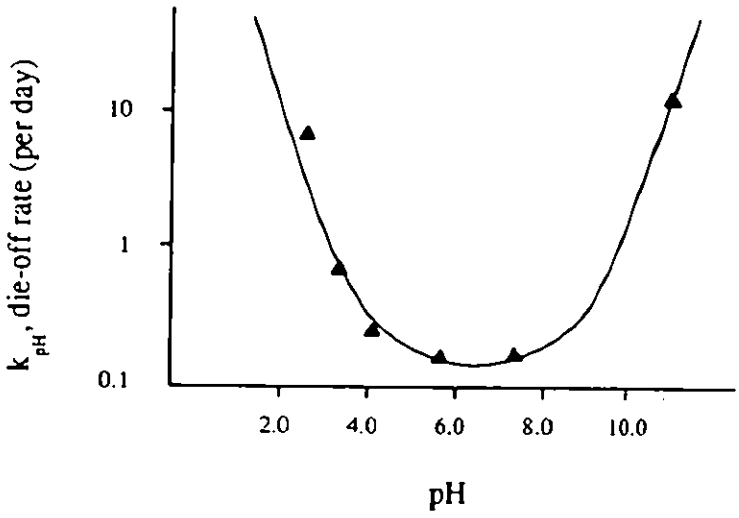

Figure 2.8 *Plot of observed (after McFeters and Stuart, 1972) and modelled (solid line) of E.coli die-off rate per day against pH.*

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;

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

Where, pH_{min} 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 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).

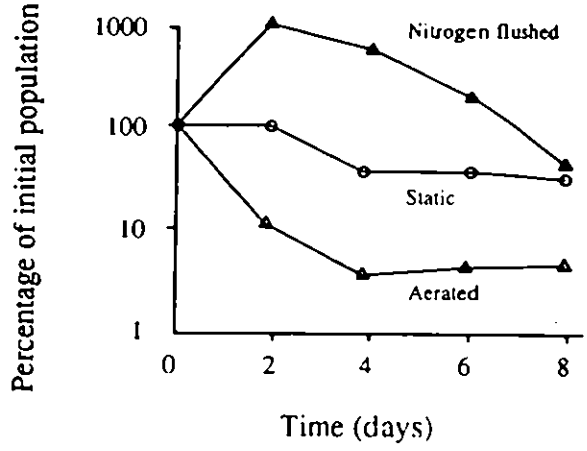

Figure 2.9 Survival of faecal coliform in raw river water at 20°C in aerated, static and N_2 flushed flasks (after Zerfas, 1970).

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 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 affording 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).

*Table 2.5 Bacterial survival in different sediments (Burton et al., 1987; *Sherer et al., 1992).*

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				$*0.666 10^{-3}$
14 : 12 : 74				$*0.791 10^{-3}$

Allen (1953) found that coliform survival in different mud samples was quite uniform and that the organisms were concentrated in the top 5cm 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 different (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 to 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 and the lack of 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), and enhanced survival has been observed in high conductivity water (McFeters and Stuart, 1972).

The results of 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 needed to simulate adequately field conditions were, however, only partly met. The water used to circulate over the sediment samples was sterile and reconstituted to the major ion concentrations observed in the field. 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) suggested an apparent self-protection mechanism in estuarine waters. In the absence of sediment particles *E.coli* were observed to aggregate forming an inner core protected from the osmotic stress caused by the saline 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 12.5mg.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 5mgSS.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 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.

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 affects of particulates and nutrient supply on faecal coliform survival are complex, the most tangible of these are 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 properties. Survival at the channel-bed/ water interface may be 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 (Cohen and Shuval, 1973; 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 being 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.

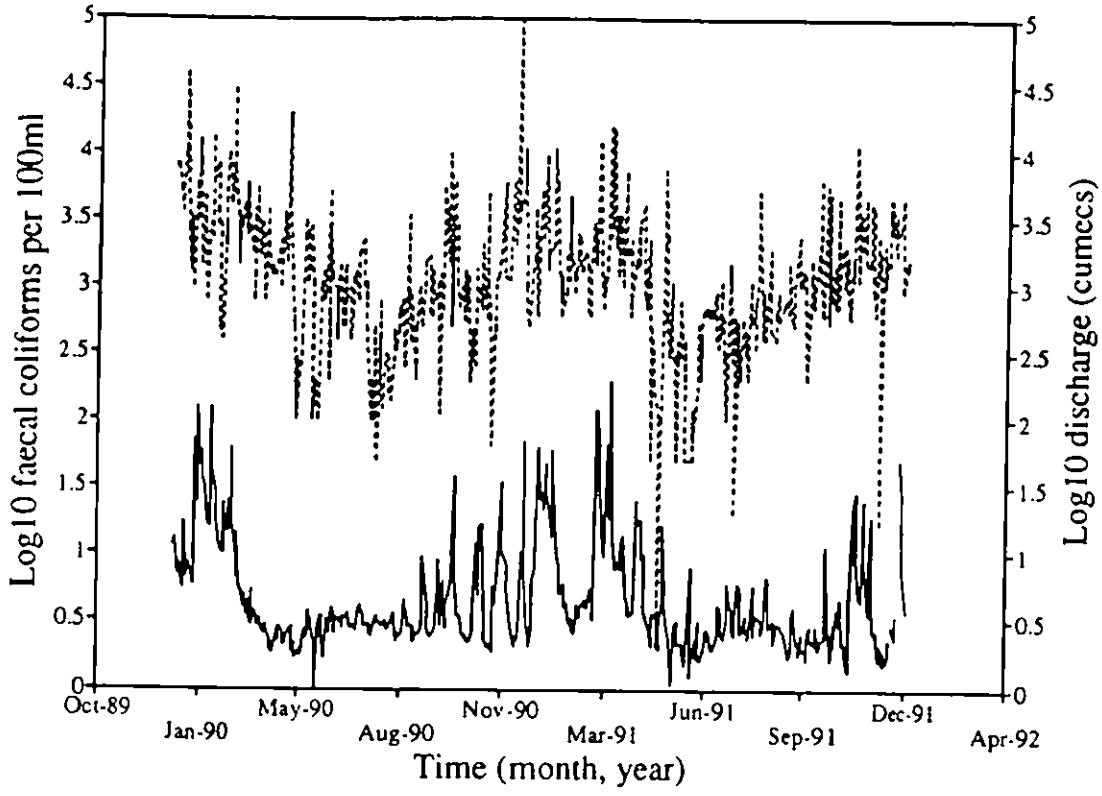

Figure 2.10 Plots of log_{10} faecal coliform concentration (----) and log_{10} discharge (solid line) for the River Dee at Huntington showing seasonal variation between the calender years 1989 to 1992.

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3. STORAGE AND TRANSPORT OF FAECAL COLIFORMS IN CATCHMENTS.

Section 2 of this report examined influences on faecal coliform survival in catchments and surface waters, this section examines 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 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 upto 4 months after being deposited (Evans and Owens, 1972). In lowland areas, transport by water infiltrating the soil mass may account for 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 baseflow 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 dependant on supply and die-off processes.

The significance of different sources of faecal contamination in a river system 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. In disturbed soils organisms will tend to be retained as a result of straining at pore margins and attachment to particles.

Downstream, point sources and resuspension from storage within the channel take on greater significance in the supply of contaminative organisms. The attachment to particulates and clumping of organisms enhances settlement within the channel leading to the accumulation of organisms at the water/channel-bed interface.

During flow events both surface runoff and entrainment from channel-bed storage results in major discharges of organisms. Point source inputs that discharge into the river system will maintain bacterial numbers in the water column during lower flows.

Organisms derived from different supply pools i.e. from point or non-point sources may behave differently on reaching the water column, for example, organisms derived from sewage effluents, being largely devoid of settleable solids, may remain in suspension for a number of hours, but will lack the protection afforded by particulates and suffer more rapid die-off. Catchment derived inputs, more likely to be associated with turbid runoff, may settle more readily and survive for longer.

The transient movement of organisms to and from storage within a stream or river channel bed makes the modelling of this behaviour relatively complex in comparison to purely aqueous phase contaminants. The description of this transfer is the subject of Section 5 of this report.

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, ie. 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 Mitchell, 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.

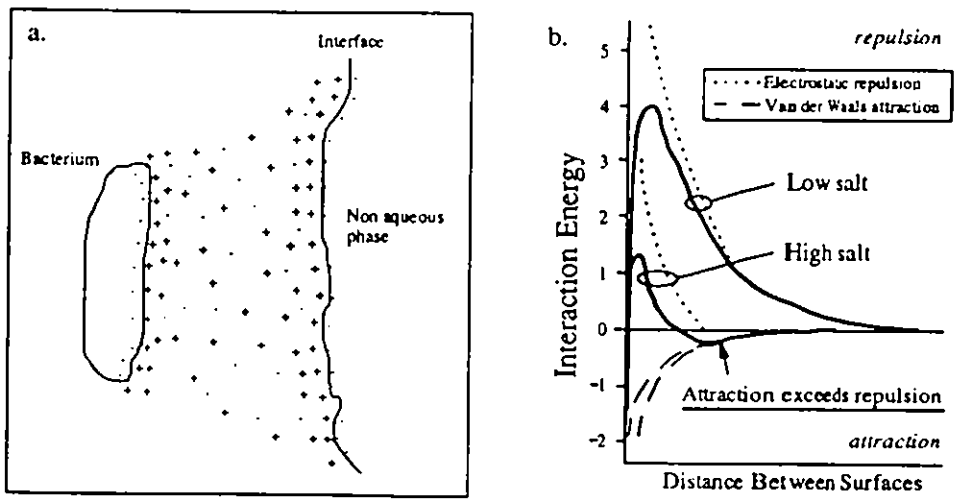

Figure 3.1 a. Schematic representation of the interaction between potentially overlapping cation clouds accompanying a negatively-charged bacterium as it approaches a negatively-charged interface, b. idealised curves showing the potential energy of interaction as a function of distance between a bacterium and an interface in solutions of different salt concentration (after Marshall 1979).

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 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 the 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 reduction of pH below 8.0, the addition of divalent cations, ie 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 a range of 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 of the review therefore 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 un-disturbed 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 through-flow 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. Bacterial transport is certainly enhanced in the saturated zone (Hagedorn *et al.*, 1981).

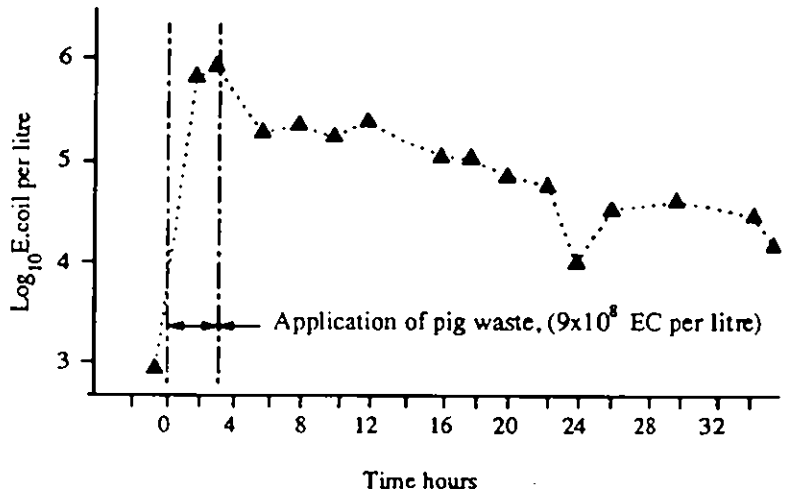

Figure 3.2 Response of E.coli concentration in water draining an extensive land drainage system to an application of piggery waste (after Owens and Evans, 1972).

Hunter *et al.*, (1992) have recently examined the relative contributions of inputs from overland flow, matrix through flow, and non-matrix through flow to stream bacterial numbers in an upland experimental catchment in Yorkshire. Matrix through-flow was found to produce a small stream bacterial loading in comparison with non-matrix through-flow and overland flow, which was due to low concentrations and low flows. It was suggested, however, 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 described in the literature 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 3 and 5 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 bacteria from the catchment surface on the land 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.

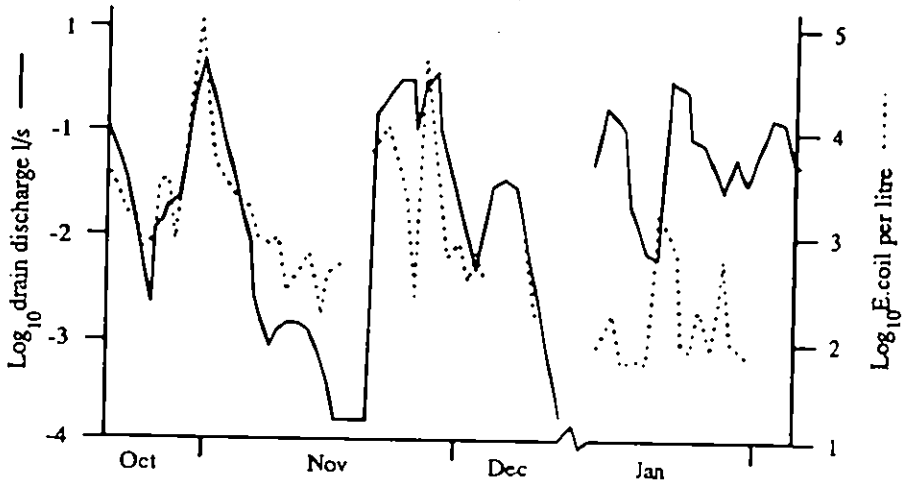

Figure 3.3 Variation of E.coli concentration in water draining an extensive land drainage system in the absence of recent inputs of faecal bacteria (after Igneous and Evans, 1972).

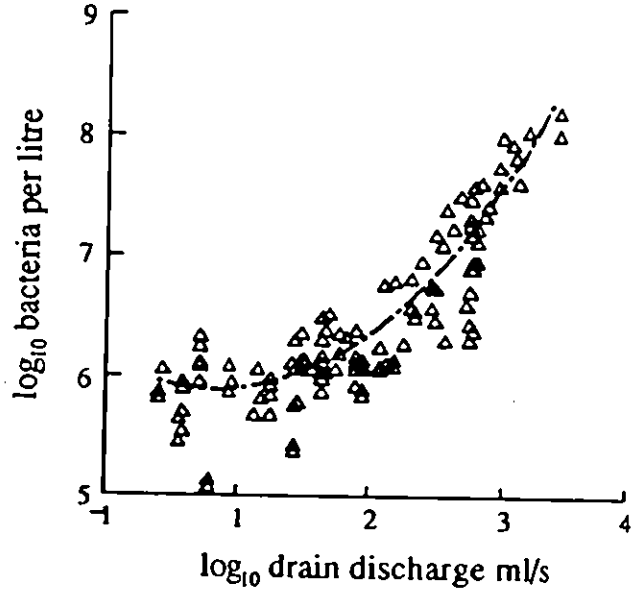

Figure 3.4 Curve showing relationship between viable bacteria and land drain discharge (after Owens and Evans, 1973).

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 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, ie. less at lower flows; the pore water velocity, greater detachment of bacteria from soil particles 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 through-flow 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 and 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 have recently 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 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 concurred 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).

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 preceeding sampling (mm)
RFT2	Rainfall in the 24HRS preceeding sampling (mm)
RFT3	Rainfall in the week preceeding sampling (mm)
TRF1	Time elapsed since daily rainfall > 1mm (days)
TRF2	Time elapsed since daily rainfall > 3mm (days)
TRF3	Time elapsed since daily rainfall > 10mm (days)
NRF	No. days in preceeding 10 when rainfall > 1mm
ST	Stage Hgt. at time of sampling
TST	Time since stage hgt. > 0.23m (≈ baseflow)
RSD	Relative sheep stocking density
TMP	Mean air temperature for the day preceeding sampling

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/100ml) in:	
	Overland flow (Mean FC concentration per 100ml)	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

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 runoff originating on saturated areas which built-up along channel edges. It was therefore suggested, that due to the runoff 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 throughflow and non-matrix throughflow, at 11 sites along the channel. The faecal coliform load contributed by overland flow was 5 orders of magnitude higher than matrix through flow and 2 orders of magnitude higher than non-matrix throughflow. Overland flow was a major contributor to stream bacterial load during both base and stormflow 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.

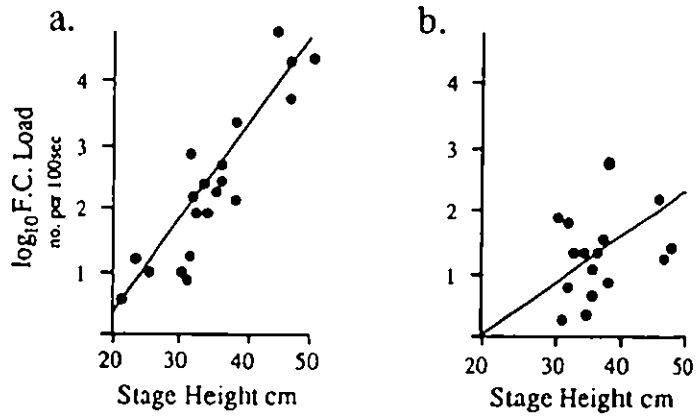

Figure 3.5 *Scatter plots of $\log_{10}$ faecal coliform load against stage height for inputs from a. overland flow and b. non-matrix throughflow (after Hunter et al., 1992).*

These studies indicate that faecal coliforms 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 dominate 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.

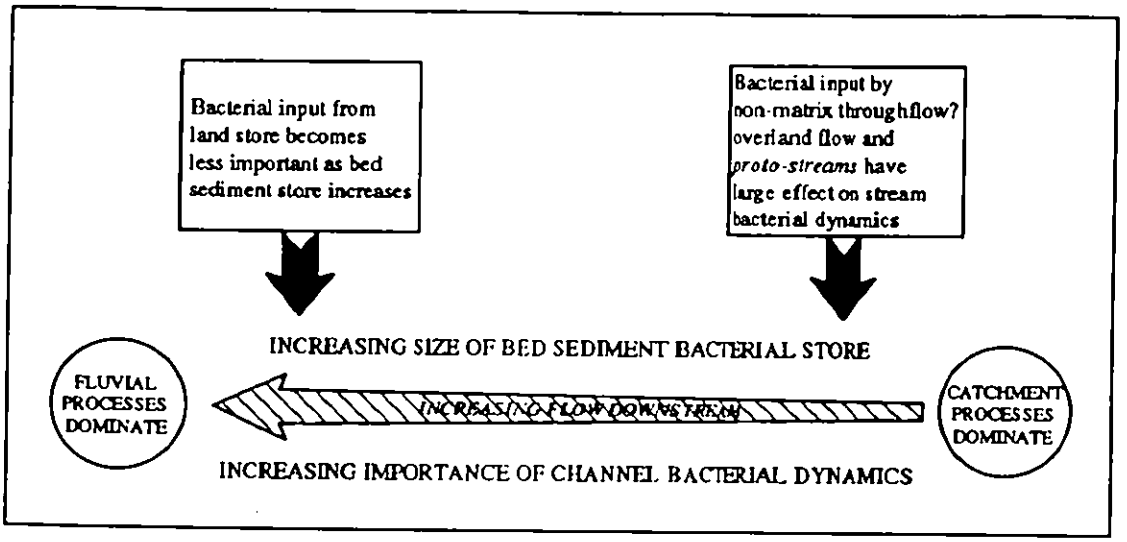

Figure 3.6 The relative importance of bacterial input rates from the catchment land store and the channel sediment store to stream bacterial dynamics, with distance downstream (after Hunter et al., 1992).

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 resuspension 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). Resuspension of these stored organisms may result from the disturbance caused, for example, by a recreational activity or in response to a flow event. In near static water bodies such as lakes and reservoirs the potential for transport of sediment bound bacteria beyond the main body of water is limited as die-off in the water column and settlement will dominate their behaviour.

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 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, however, shown 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 30mm was not a function of time and no significant alteration in concentration occurred after 3hr. 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 $5x10^4$ to $7x10^4$ *E.coli* per 100ml 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. Upto 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 30µm diameter, or whose settling velocity was equivalent to that of mineral sediment grains of less than 30µm 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 $1.2m.day^{-1}$ 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, however, 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, 1967; 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 hydrograph events.

An intensive programme of sampling for coliform organisms both during periods of constant flow, to establish background variations in concentrations, and storm events, in order 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 100ml). During the rising limb of all hydrograph 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 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).

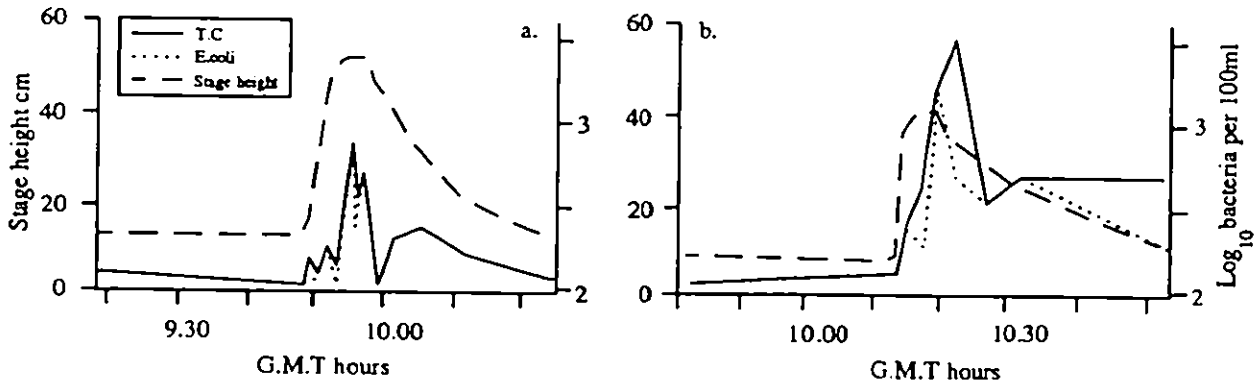

Figure 3.7 Propagation of artificially generated hydrograph with response of total coliforms (TC) and E.coli concentration at locations (a) 400m, and (b) 2500m downstream of the hydrograph source (after McDonald et al., 1982).

To confirm the above results a release was made immediately after a major natural flow event that should have flushed the channel bed 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).

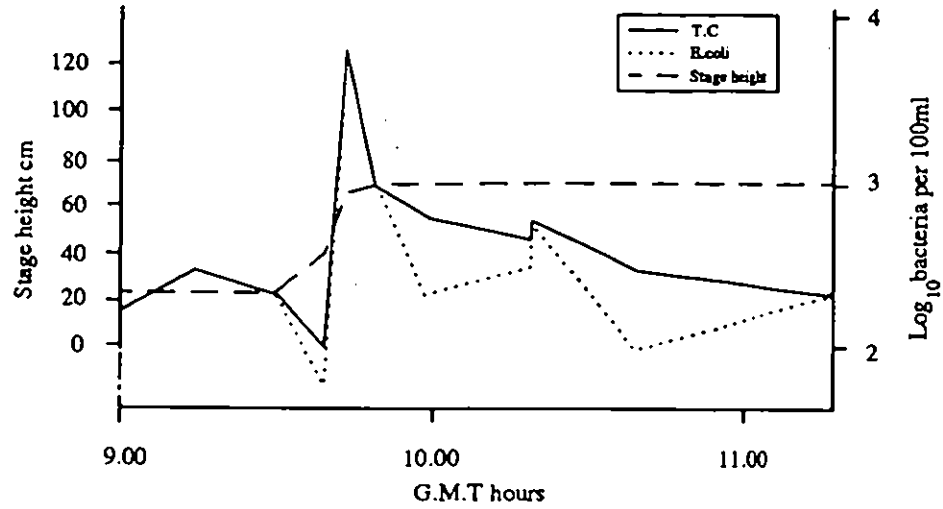

Figure 3.8 *The response of total coliforms (TC) and E.coli concentration to a step change in stage height (after McDonald et al., 1982).*

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 resuspension may occur at the margins of the water body. Resuspension of bacteria by wind-wave action was thought to be responsible for faecal coliform concentrations of upto 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 runoff 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 32km 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.

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4. EXISTING MODELS FOR FAECAL COLIFORM TRANSPORT AND SURVIVAL

Models used to describe faecal coliform dynamics are wide ranging.

Multivariate statistical approaches relate bacterial concentration to a number of driving variables using simple statistical relationships. This purely "black-box" approach pays little attention to the physical nature of the system or its internal processes.

Simple deterministic first order decay functions have been used for describing the exponential die-off of some initial bacterial population. This approach is only appropriate to closed systems; for example a beaker in a laboratory subject to constant environmental conditions. Minor variants to this type of model use different form curves for the die-off characteristic and may be extended with coefficients for temperature effects.

Model applications to lakes, estuaries and rivers apply partial differential equations to describe fluid mixing processes in two or three dimensions. They might also include momentum equations for describing flow hydraulics. Terms are then added to describe the non-conservative behaviour of faecal coliforms, such as the first order decay functions mentioned above.

Section 3 of this report demonstrated the importance of the transfer of organisms to and from storage at the water/channel bed interface. This concept was first incorporated into a mass balance model by Jenkins (1984).

The new model presented in Section 6 of this report 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.

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).

$$\frac{N_t}{N_0} = 10^{-kt} \ldots\ ie.\ N_t = N_0 . 10^{-kt} \qquad (4.1)$$

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 $t > t_1$, the time delay, after which decline commences (Figure 4.1, curve 2).

$$\frac{N_t}{N_0} = 10^{-k(t-t_1)} \qquad (4.2)$$

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

$$\frac{N_t}{N_0} = a.10^{-k_a . t} + b.10^{-k_b . t} \qquad (4.3)$$

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).

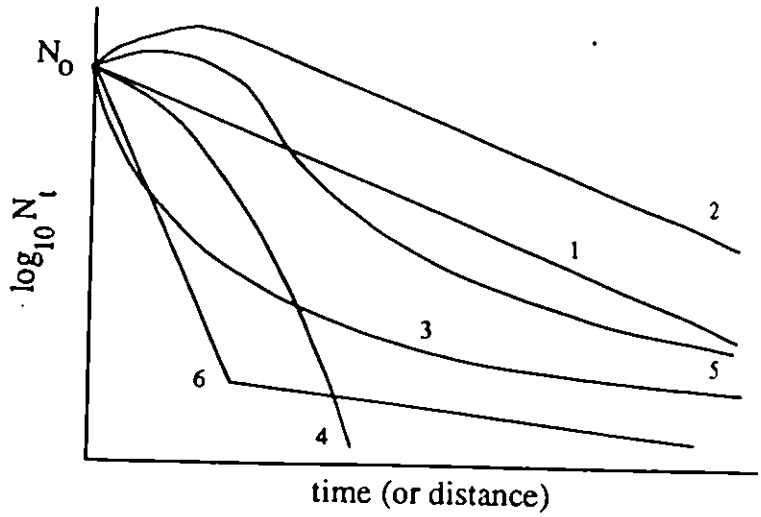

Figure 4.1 *Bacterial die-off curves as predicted by models in the literature (after Crane and Moore, 1986).*

$$\frac{N_t}{N_0} = a\,10^{-k_a t} + b\,10^{-k_b t} + \ldots\ldots n\,10^{-k_n t} \qquad (4.4)$$

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).

$$\frac{N_t}{N_0} = \frac{1 - 10^{-k' t}}{2.3\,k'.t} \qquad (4.5)$$

Where k' is the die-off coefficient.

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

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).

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

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.

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

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;

$$y = A\, e^{-\beta \ln x} \tag{4.9}$$

from which the following equation was derived;

$$lnC = a + b[\ln (t + 1)] + \varepsilon \tag{4.10}$$

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 with in 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 which 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, 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 it's cross-section, a one-dimensional formulation is used (Equation 4.11).

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

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

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 a pulse of contaminant entering a point x_o in a river at time t_o changes through time. Figure 4.2a, shows how the model works if $U=0$, ie 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, but used 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 all of the following models and is applied to salinity variation, sediment and bacterial transport and flood-wave propagation. 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.

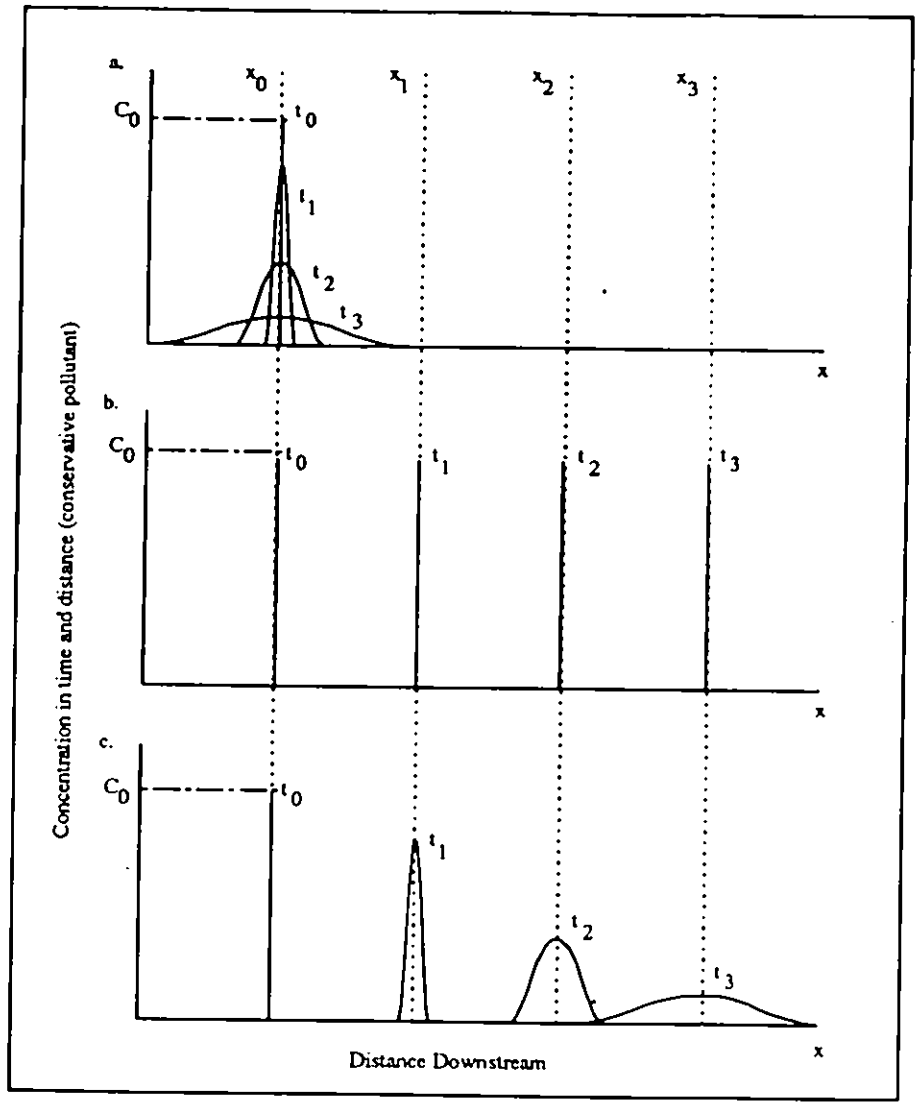

Figure 4.2 A graphical representation of modelling the dispersal of a conservative contaminant under steady flow conditions using the ADE.

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 ($m^2.sec^{-1}$), ρ is density ($kg.m^{-3}$), A is cross sectional area of flow (m^2), 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 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).

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

and,

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

and,

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

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

For deposition $u_* < v_d$,

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

and for erosion $u_* > v_e$,

$$E = \left(\frac{u_*^2}{v_e^2} - 1\right) W_b\,\chi \ \ldots for \ \ u_* > v_e \qquad (4.18)$$

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

$$u_* = \left|\frac{Q}{A}\right| \cdot \frac{k}{\log_e \{ 30.2 \frac{H}{k_s} \}} \qquad (19)$$

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 grainsizes, 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, according to the transport characteristics of the particulates with which organisms tend to be associated.

Sediment transport model

Historically, the behaviour of suspended solids in open channels has been described in a two dimensional context i.e vertical and horizontal. Consequently, a two-dimensional steady-state formulation of the ADE (Equation 4.2) has been utilised to describe the transport and distribution of suspended solids and associated pollutants, including bacteria, in a steady turbulent river channel (Uchrin and Weber, 1979, 1980, 1981).

$$\frac{\partial C}{\partial t} = -U\frac{\partial C}{\partial x} + D\frac{\partial^2 C}{\partial x^2} + W\frac{\partial C}{\partial y} + D\frac{\partial^2 C}{\partial y^2} \qquad (4.20)$$

where, C is the concentration of suspended solids, x is distance downstream, y is water depth, W is particle settling velocity, U is time averaged velocity, assumed to be the same for ambient fluid and suspended solids and D is the turbulent diffusion coefficient for particles.

If a uniform vertical distribution of particles is assumed an upstream boundary condition and an upper water surface boundary condition can be formulated, assuming there is no transport through that surface. A generalised bottom boundary condition may be formulated to account for the possibility of complete settling or re-entrainment of particles given by;

$$D\frac{\partial C}{\partial y} = -iWC \qquad (4.21)$$

where, i is a re-entrainment coefficient ranging between 0, for complete settling, and 1, for complete re-entrainment. W is determined as a function of hydraulic and particle characteristics using an empirically based analytical method. Particle settling velocity is treated as a randomly distributed function, generated from quiescent column settling data, which can be calibrated to account for turbulence. Given these boundary conditions and definitions, the two-dimensional partial differential equations (PDE) can be solved numerically. The resultant

solutions can then be used to generate a rate law for specific water quality parameters of the form;

$$\frac{dC}{dt} = k.C^{n} \qquad (4.22)$$

where t is the time of travel and k and n are constants relating to particle and hydrodynamic conditions. For uniform particle fall velocity, n has been found to approach unity.

This suspended solids model has been incorporated into a lake system model, which utilises a finite section approach, and applied to a recreational lake in South East Michigan in the United States. In application of this model bacterial removal from the water column, other than by sedimentation, i.e. predation, die-off, etc., was lumped into a first order function parameterised from field data.

4.3 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;

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

and,

$$N = S - W - D \qquad (4.24)$$

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 land-surface, W is washout from bed sediments, S is loss to sediments (settlement) and D is net die-off as a result of environmental stresses.

Sediment behaviour is determined by the flow characteristics of the stream ie, turbulence and velocity as related to discharge. The on-set of washout is assumed to occur at some threshold discharge below which the washout term is zero. It is 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 runoff components. The baseflow component comprises, soil throughflow and groundwater flow, responsible for maintaining inputs during dry periods. The quickflow component resulting from rainfall events over the catchment, resulting in surface run-off and non-matrix throughflow. 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 the effect of turbulence which is assumed to reduce the rate of settlement. The model uses a threshold discharge value at which bacterial settling halved (Equation 4.26).

$$S = P_{set} \frac{Q}{1 + Q/P_Q} C = p.C \tag{4.26}$$

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 also 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).

$$\frac{dN}{dt} = \frac{P_{set} \frac{Q}{1 + Q/P_Q} C}{v.w} = \frac{S}{v.w} \tag{4.27}$$

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;

$$v.w = \frac{Q}{Q^{0.5}} \tag{4.28}$$

Substituting Equation 4.27 into 4.28 and rearranging gives;

$$\frac{dN}{dt} = P_{set} \frac{C}{1 + Q/P_Q} Q^{0.5} \tag{4.29}$$

This modification causes a slight decrease in the number of bacteria settling into the sediment store once $Q > P_Q$. Entrainment or wash-out of bacteria from the bed sediment store is assumed to follow a suspended sediment function based on a velocity/discharge relationship whereby;

$$Bacterial\ Load = a.Q^b \tag{4.30}$$

It is suggested that values for b fall in the range 2-3. The entrainment of organisms at some threshold discharge Q_T is given by;

$$W = a(Q^2 - Q_T^2)N \tag{4.31}$$

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

$$a = \frac{T_{H2}}{2.Q_T + 1} \tag{4.32}$$

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

This allows a to be conveniently formulated in terms of perceived system behaviour or field observations. Equation 4.31 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

4.31, N is replaced by;

$$f(N) = \frac{N.\ Sat}{N + Sat} \tag{4.33}$$

Where Sat is the maximum value of $f(N)$, which only approaches Sat for values of N much greater than Sat (Figure 4.3).

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;

$$C_t = C_0\, e^{-kt} \tag{4.34}$$

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;

$$\frac{dC}{dt} = -K_c.C - p.C + \frac{a(Q^2 - Q_t^2)N}{Q} + \frac{I_B}{Q} + I_Q \tag{4.35}$$

$$\frac{dN}{dt} = -K_n.N + p.Q.C - a(Q^2 - Q_t^2)N \tag{4.36}$$

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

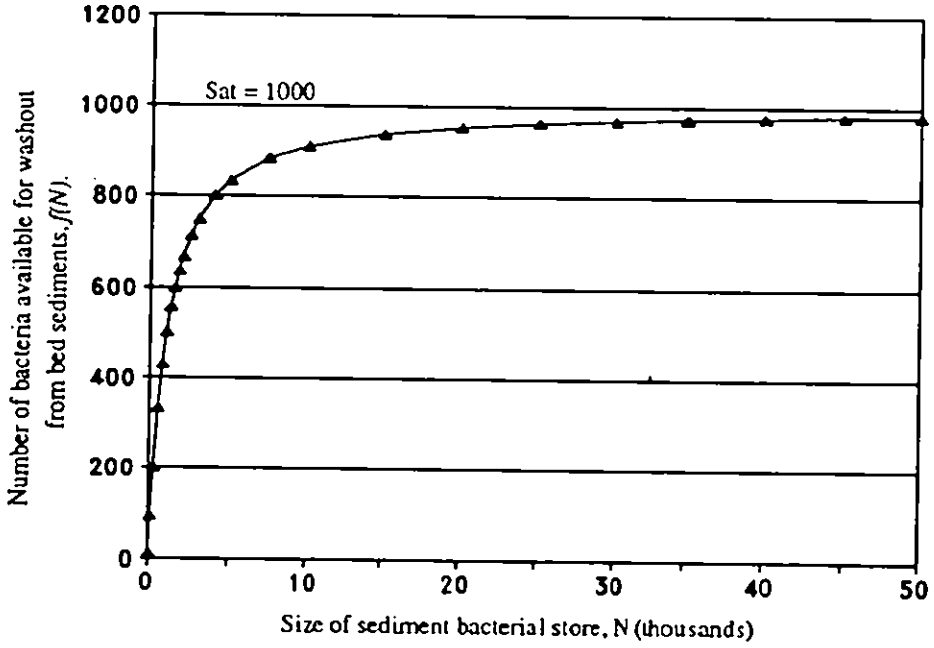

Figure 4.3 Plot of washout limitation function, for Sat=1000, as given by equation 4.18.

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 was needed to produce the observed coincidence of bacterial peaks with respect to the hydrograph peaks (Jenkins *et al.*, 1984).

This model uses a total of five parameters in describing the transfer of organisms to and from bed-storage. There is a single threshold for entrainment, the channel bed store is treated as a

single contiguos unit and there is no dispersion term in the model mass-balance structure. Section 5 of this report shows that the transfer of organisms to and from bed-storage requires only two parameters. The new model described therein has what is in effect a floating entrainment threshold discharge and the mass-balance structure is based-on Aggregated Dead-Zone dispersion theory (Henderson-Sellers *et al.*, 1988).

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5. TOWARDS A NEW MODEL OF FAECAL COLIFORM STREAM DYNAMICS

Faecal coliform concentrations in rivers are influenced by many factors; environmental and microbiological effects determine survival in the hostile aquatic environment. Settlement to, and resuspension from, storage within the stream channel means that the organisms cannot be modelled in the same manner as a simple aqueous phase.

Field experiments to determine the nature of the model internal structure i.e. the relationship between flow and channel storage have enabled the development of a new model.

The model structure and operation incorporates the following assumptions. The channel-store is distributed across the entire channel and that 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 be assumed once the flow recession commences. Further higher flows will still release organisms from storage.

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

The model sucessfully reproduces the field data and it's structure is more conceptually acceptable than any previous model.

Further developments of the model might consider extending it's scope to modelling other particle associated contaminants such as metals or hydro-carbons.

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. Alternative statistically based approaches use multivariate analysis to develop models of coliform concentrations related to a number of physical and/or chemical influences (Mahloch, 1973) 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 it's significance to water quality has been recognised for some time (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, extending the work of Jenkins (1984), which lead to the development of a new conceptualisation of faecal coliform storage within a stream or river channel.

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

Experiments were carried-out at three sites where the flow could be augmented and hence the channel faecal coliform response to flow changes without inputs from the adjacent catchment. 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 releases were provided by a hydroelectric scheme. The river was sampled 10km downstream of the Cwm Rheidol reservoir in the catchment flood plain. The reach is characterised by 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, there is also a small sewage treatment works.

The Afon Clywedog in the Upper Severn was sampled at either end of a 4km 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, catering 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 1.5km reach was sampled.

Sampling and analysis techniques

Sampling was commenced prior to the releases in order 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 400ml 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 100ml and 1ml were filtered and enumerated within six hours of collection, the results are expressed as colony forming units (cfu) per 100ml. 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 a 1.73 times improvement in accuracy approximately. 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 inputs were minimal. The observed responses are therefore assumed to result

from inputs to the flow from channel storage. Indeed, Figures 5.1 and 5.2 and Table 5.1 demonstrate the cumulative nature of the entrainment process with distance downstream during sampling on the Clywedog and Washburn. The Washburn results demonstrate the finite nature of the channel-bed supply of organisms.

Table 5.1: *Geometric mean faecal coliform concentrations (cfu per 100ml) for the three experimental release sites showing differences in concentration during hydrograph rise and recession and the accumulation of organisms with transport downstream.*

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

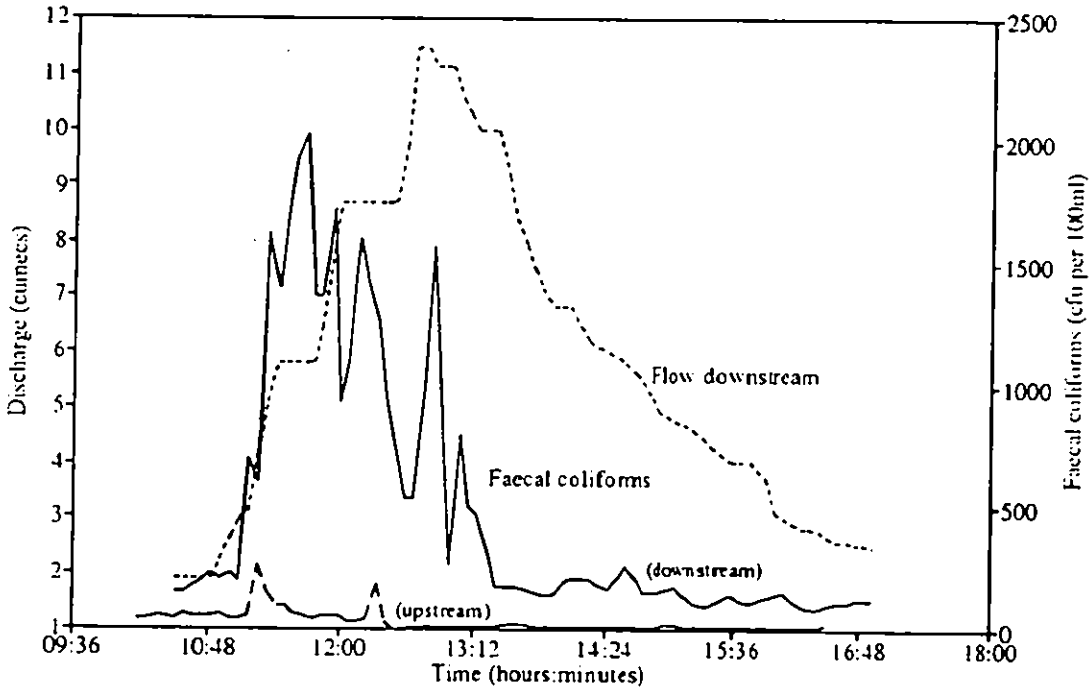


Figure 5.1 *Raw faecal coliform and flow data for the experimental flow release in a 4km reach of the Afon Clywedog downstream of the Clywedog reservoir (28.5.93).*

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 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.

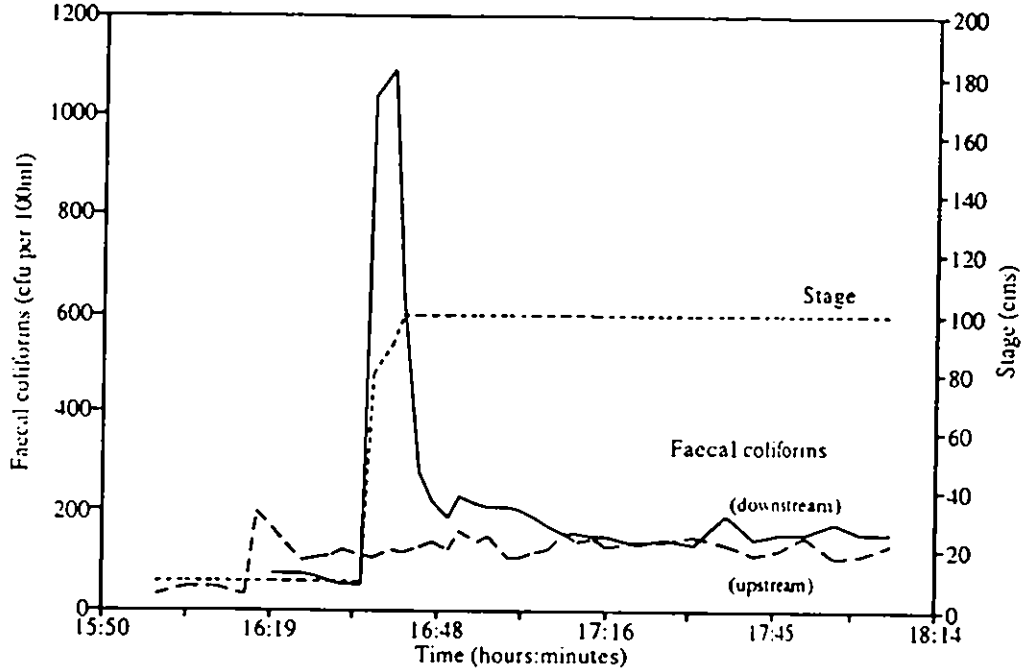


Figure 5.2 Raw faecal coliform and flow data for the experimental flow release in a 1.5km reach of the River Washburn downstream of the Thruscross reservoir (26.5.93).

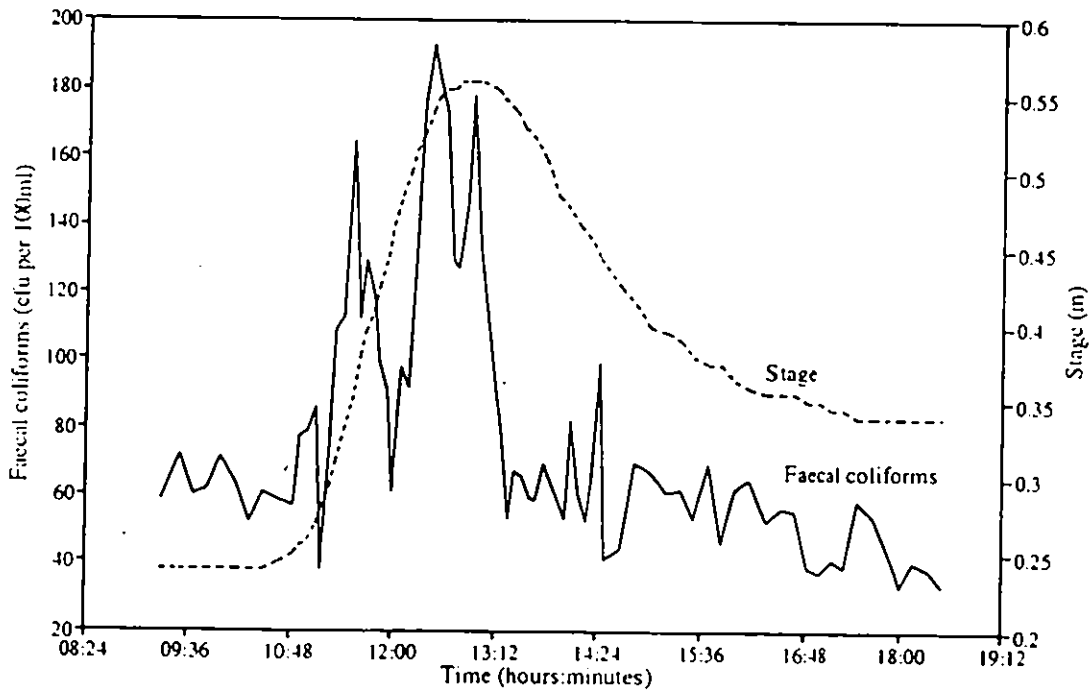


Figure 5.3 Raw faecal coliform and flow data for the experimental flow release on the Afon Rheidol at a site sampled 10km downstream of the Cwm Rheidol Dam (17.2.93).

Examination of data for the Washburn release (Figure 5.2) shows that a 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 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. One factor affecting entrainment and the observed bacterial response is likely to be 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).

A more recent study in the River Severn demonstrated a large-scale dead-zone feature with an inferred residence time of around 25 days (Reynolds *et al.*, 1991). Areas of low velocity are ideal for the preferential accumulation of faecal coliforms, 90% of which have been shown to be associated with particulates of settling velocity 1.2m per day (Gannon *et al.*, 1983; Auer and Niehuas, 1993). 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 irregular entrainment. Entrainment is modelled purely as a function of flow. This assumption appears to be valid for the Clywedog and Washburn data, but not the Rheidol.

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. 90% of faecal coliforms in storm flow sediments have been found to be associated with particle sizes of between 0.45-10μm (Gannon *et al.*, 1983; Auer and Niehuas, 1993) and it might be expected that the release of such fines and associated organisms is related to the disturbance of sediments of increasing grain sizes. Jenkins (1984), however, 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. Areas of bed storage are assumed to occur randomly over the channel bed and to undergo entrainment over a wide range of flows. In the new model, entrainment from the many areas of storage is lumped into 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 flow, more organisms become "available" for rapid entrainment 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 $N_{t,j}$ organisms in each store;

$$N_{t,j} = \frac{NT_t}{j} \quad and \quad \Delta Q = \frac{Q_m - Q_0}{j} \tag{5.1}$$

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;

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

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;

$$\frac{dNT_t}{dt} = V.k_s.n_d.x_t - w \sum_{j=0}^{n_s} N_{t,j} \tag{5.3}$$

Where, x_t 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;

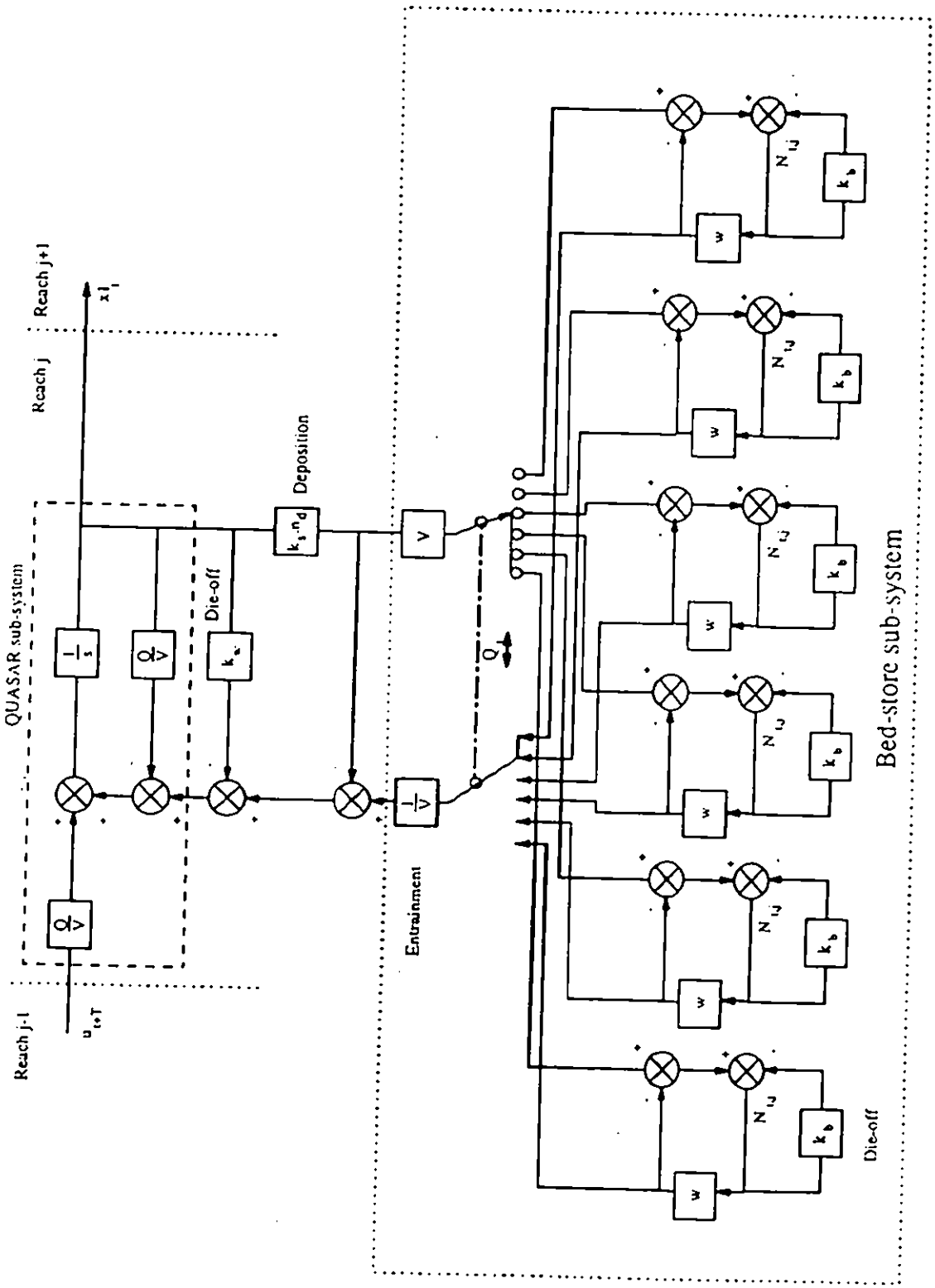
$$\frac{dN_{t,j}}{dt} = -(w + k_b).N_{t,j} \tag{5.4}$$

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;

$$\frac{dN_{t,j}}{dt} = V.k_s.x_t - k_b.N_{t,j} \tag{5.5}$$

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).*



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_t. If u_t is greater than the concentration in the dead-zone, x_t, that concentration rises. If u_t is less than x_t, x_t 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;

$$\frac{d\,V.x_t}{dt} = Q.u_{t-\tau} - Q.x_t \qquad (5.6)$$

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

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

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:

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

and,

$$Ne_t = \sum_{j=0}^{n} N_{t,j} \qquad (5.9)$$

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 6 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;

$$k_w = k_w + \Delta k_I + \Delta k_T + \Delta k_{pH} \tag{5.10}$$

where;

$$\Delta k_I = \alpha . I_{z,avg} \tag{5.11}$$

is the change in die-off rate due to solar radiation, α 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.

$$\Delta k_T = k_w \{ 1 - 10^{\theta.(T_2 - T_1)} \} \tag{5.12}$$

In Equation 5.12, θ, is the slope of the die-off/temperature response curve. T_1 and T_2 are temperature in °C at time step t-1 and t, respectively. The derivation of this equation and values of θ 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 θ were around 0.045 in non-sewage impacted waters and 0.013 in sewage contaminated waters.

$$\Delta k_{pH} = k_w \{ 1 - \cosh (a [pH_{k_{w,min}} - pH]) \} \tag{5.13}$$

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 Section 5 of the report 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;

$$k_w = k_{w,min} + k_{kw} (1 + \cos [\pi + \omega t]) \tag{5.14}$$

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 Earths orbit of the Sun in radians/day and t is the time step in days starting arbitrarily at either the shortest day or the New Year.

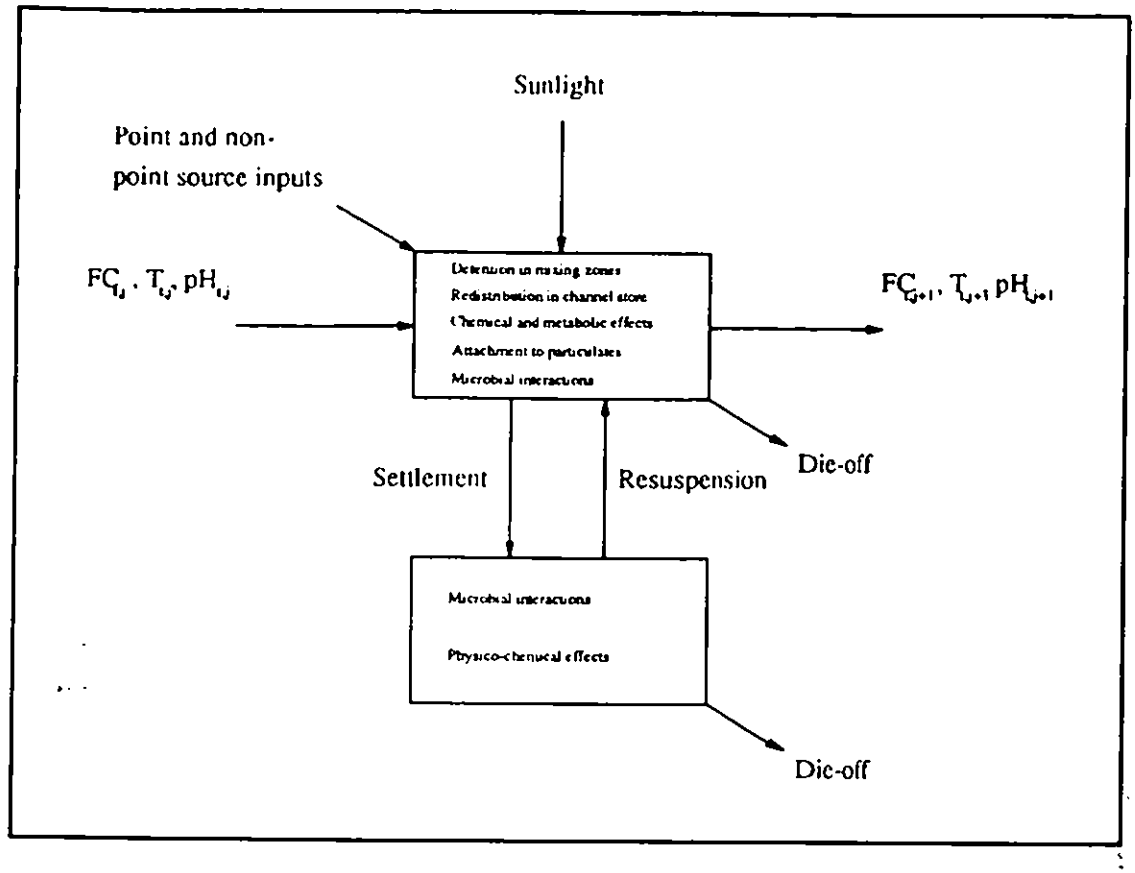


Figure 5.5 Two-box conceptualisation of the faecal coliform model as applied to a single river reach.

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 occur are large relative to the speed a parcel of water travels through a reach. It was also necessary to provide the initial total number of 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 which 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.

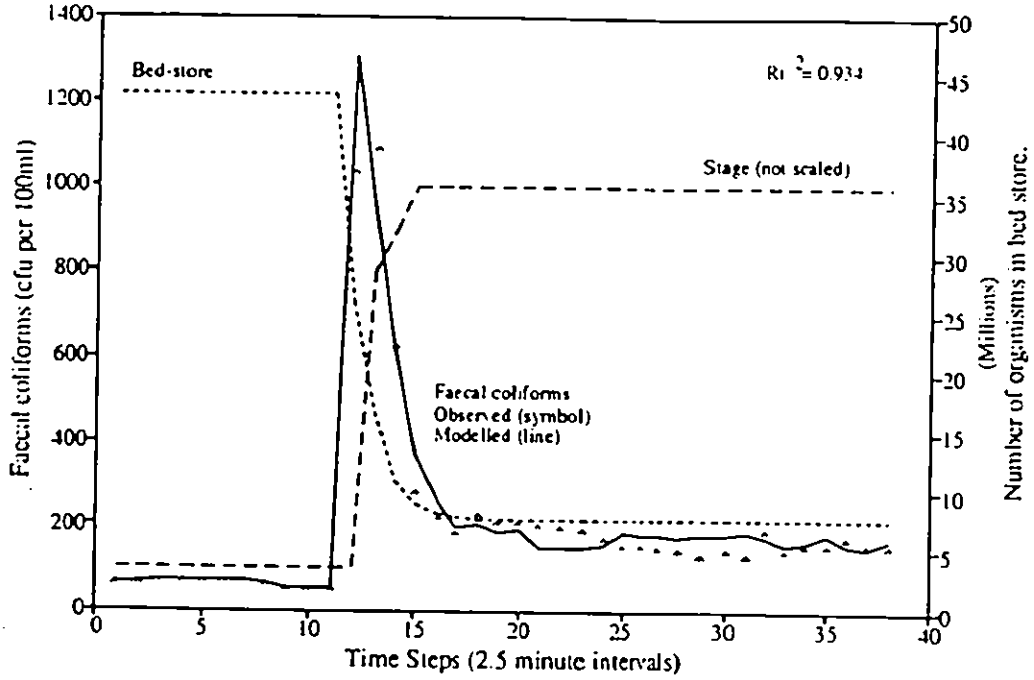


Figure 5.6a Observed and modelled faecal coliform concentrations in response to a step change in flow on the River Washburn. Rt^2 is a goodness of fit criterion and should be approaching 1.

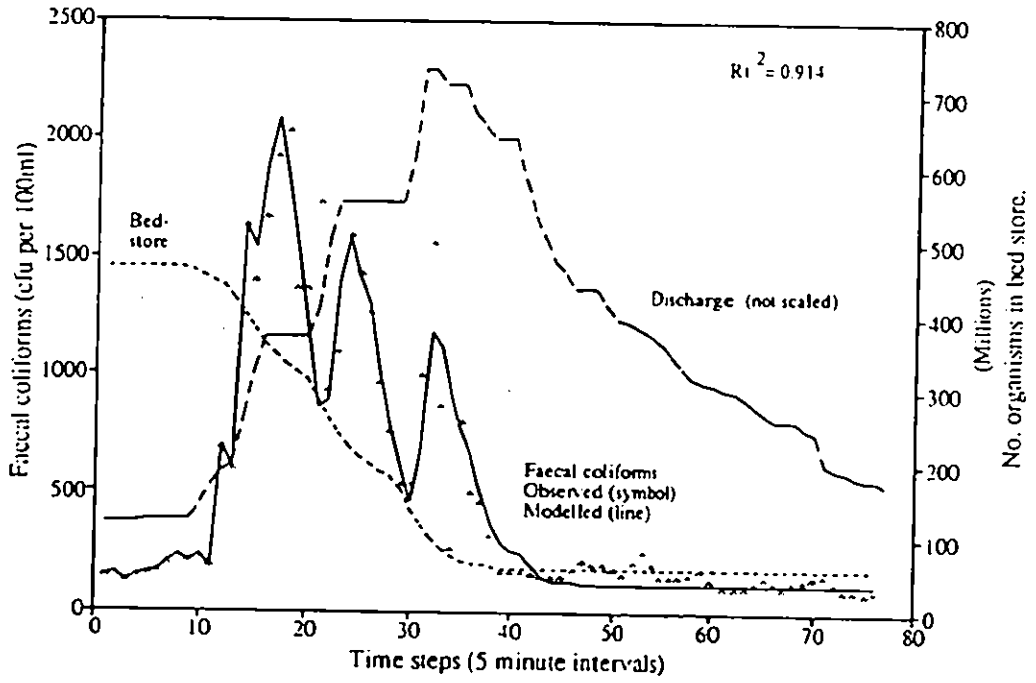


Figure 5.6b Observed and modelled faecal coliform concentrations in response to a stepped artificial flow event on the Afon Clywedog.

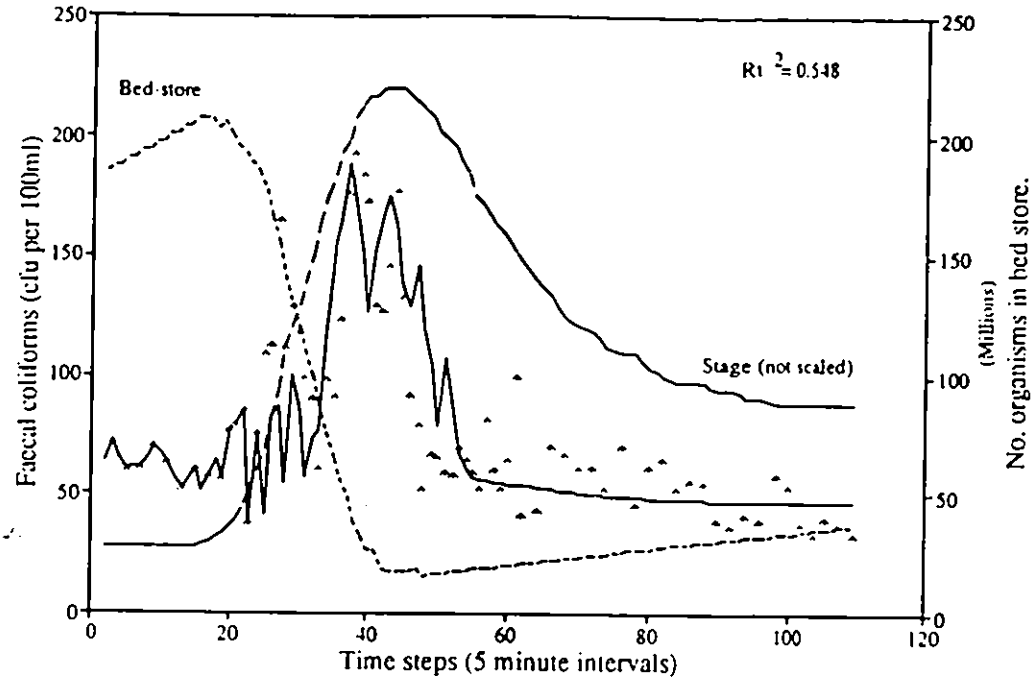


Figure 5.6c Observed and modelled faecal coliform concentrations in response to a stepped artificial flow event on the Afon Rheidol.

The successful model presented above gives a good conceptualisation of faecal coliform dynamics in stream being able to reproduce field observations where existing models would fail. Further developments of the model might consider extending it's scope to modelling other microbiological determinands (eg. faecal streptococci) and particle associated contaminants such as metals or hydro-carbons.

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6. MODEL APPLICATION AND VALIDATION

This section of the report describes how data derived from the routine monitoring programmes of the Water Companies and National Rivers Authority were modelled with the dynamic model described in Section 5.

In the application described, the combined seasonal effects of temperature and solar radiation on faecal coliform die-off are simulated with a simple cosine function. This gave a valuable reduction in the data requirements of the model. A series of scenarios were run to examine the impact of changes in faecal coliform inputs to the modelled reach, as well as, looking at the impact of worst case changes in flow regime resulting from climate change projections for the year 2050.

The operation of the model is demonstrated with graphical representations of the changes in bacterial concentration at discrete distances along the modelled reach, plots of entrainment data and of the numbers of organisms stored at the water/channel-bed interface.

The faecal coliform model was successfully applied to a reach of the River Exe in Devon for the years 1990 and 1991. Values of parameters chosen for the calibration were found to be comparable to those determined during the field experiments described in Section 5 of the report.

In general the model is seen to operate well for 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. This 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 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 it 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.

Future field-programmes might seek to improve or confirm the models calibration with detailed monitoring of all relevant variables over a period of several months.

6.1 Data availability for model application and validation.

The data used in these analyses were derived from routine sampling programmes undertaken by the National Rivers Authority, the Water Companies. The completion of a detailed questionnaire by the responsible laboratories established the use of common analytical standards and practices (HMSO, 1983) thus establishing that it would be acceptable to use faecal coliform data from these different sources in the model application described. Where possible a broad suite of other water quality determinands were included with these data, certain of the sites coincided with flow gauging sites, the latter being available from the IH National Water Archive. The microbiological data are stored on the IH data-base and have been used in an analysis of relationships with land-use (see Section 7).

The faecal coliform model described in Section 5 of this report requires input/output (i/o) data, it has a mass-balance structure 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

and flow data for one or both of the sites. Data for solar radiation and turbidity would be beneficial for calibration of the die-off component of the model, however, given certain assumptions it can be made into an implicit function of the model as will be shown later.

Much of the data available to this project is 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.

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 is available. If the data is to be treated as an input series, the modeller can use best estimates of parameter values from knowledge of existing applications or the literature, but can have no idea of the accuracy of the model prediction. It simply cannot be compared with actual data. Similarly, if the data were to be treated as an output time-series, it may be possible to reproduce the observed faecal coliform concentrations, however, it would be difficult to establish the source of organisms resulting in the observed bacterial concentration and care would be required in choosing realistic parameter values.

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 to the whole system.

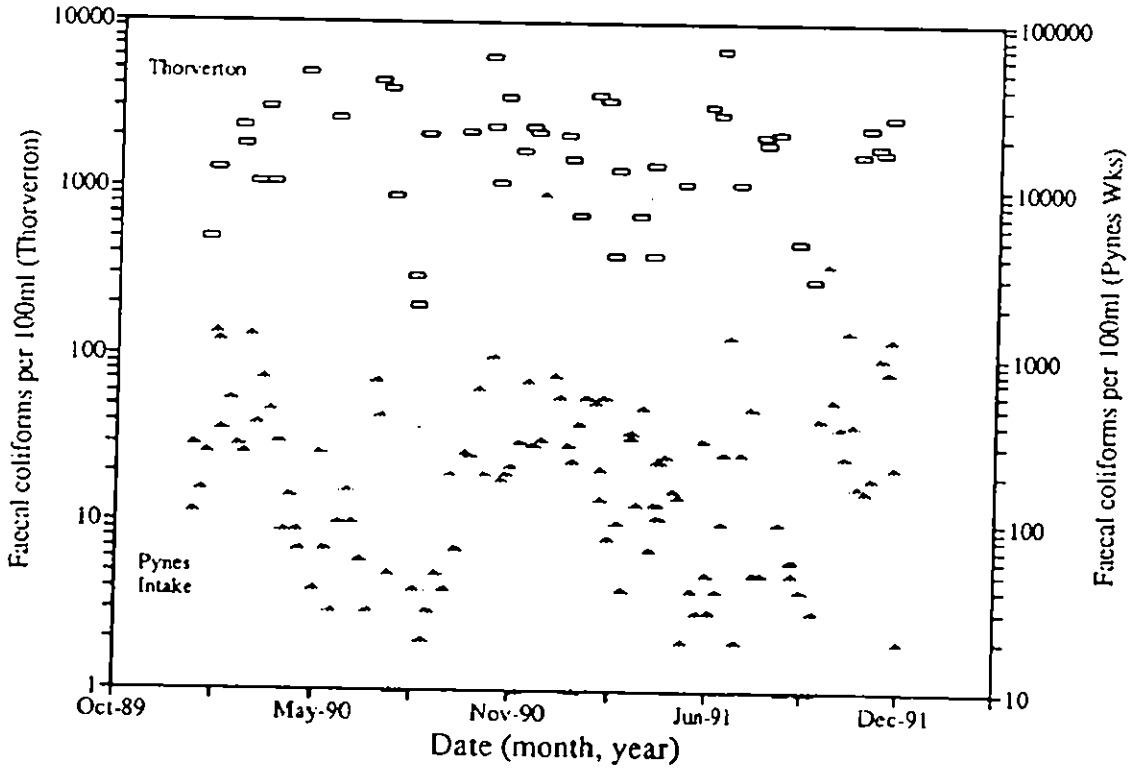
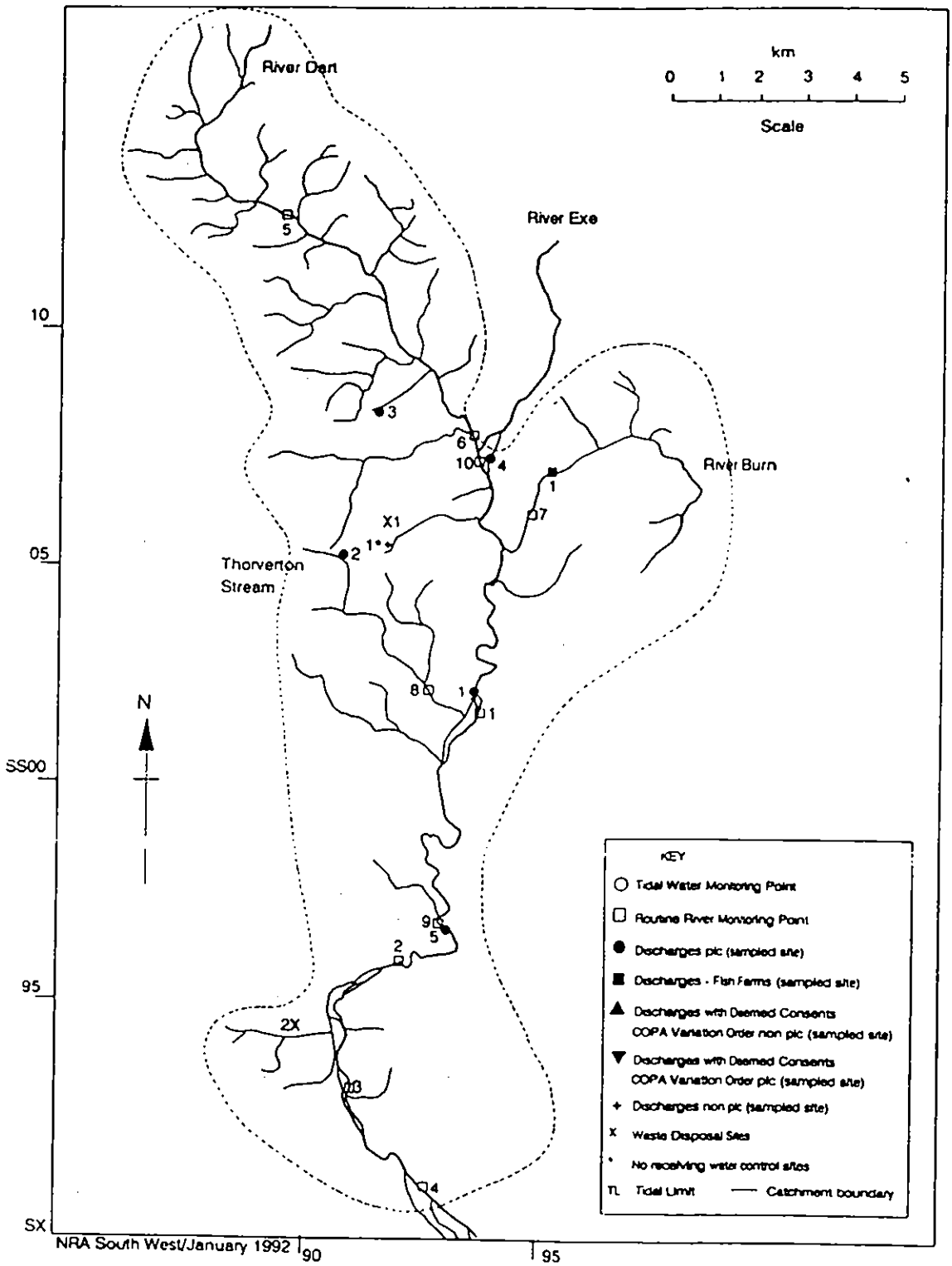


Figure 6.1 Time series of faecal coliform concentrations at Thorverton STW and the Pynes water intake on the River Exe in Devon; data are presented in their raw "as sampled" state.

Map 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 6km reach between these sites.*



Of the many sampling sites in England and Wales for which faecal coliform data is held, only six river systems had data with potential for model application. Of these, the Tyne and the Thames had too few data points, 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 6km reach (Map 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 can 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.

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. In order 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. Daily flows were retrieved for Thorverton from the IH National Water Archive. The faecal coliform time-series for Thorverton, the upstream site, was, as has been suggested, only sampled every two to three weeks and as a result inadequate for modelling purposes. It was, therefore, necessary to generate a new daily time-series for Thorverton based on the available sampled data.

The time series for Pynes intake consists of regular weekly samples taken as part of the routine raw water monitoring undertaken at that works (Figure 6.1). The data for the Pynes Intake demonstrates a seasonal trend, it is, however, difficult to discern such a trend in the data for Thorverton. As a result it was 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. In order to simulate the high variance observed in most faecal coliform time-series a random noise function was superimposed onto the seasonal trend from which the new data was derived (Figure 6.2).

Comparison of Figures 6.1 and 6.2 shows that the time-series of faecal coliform concentrations for Thorverton and Pynes Intake exhibit opposite seasonality. The explanation for this apparent anomaly 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 (refer to 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. Hence the opposite seasonal trend observed at Pynes Intake.

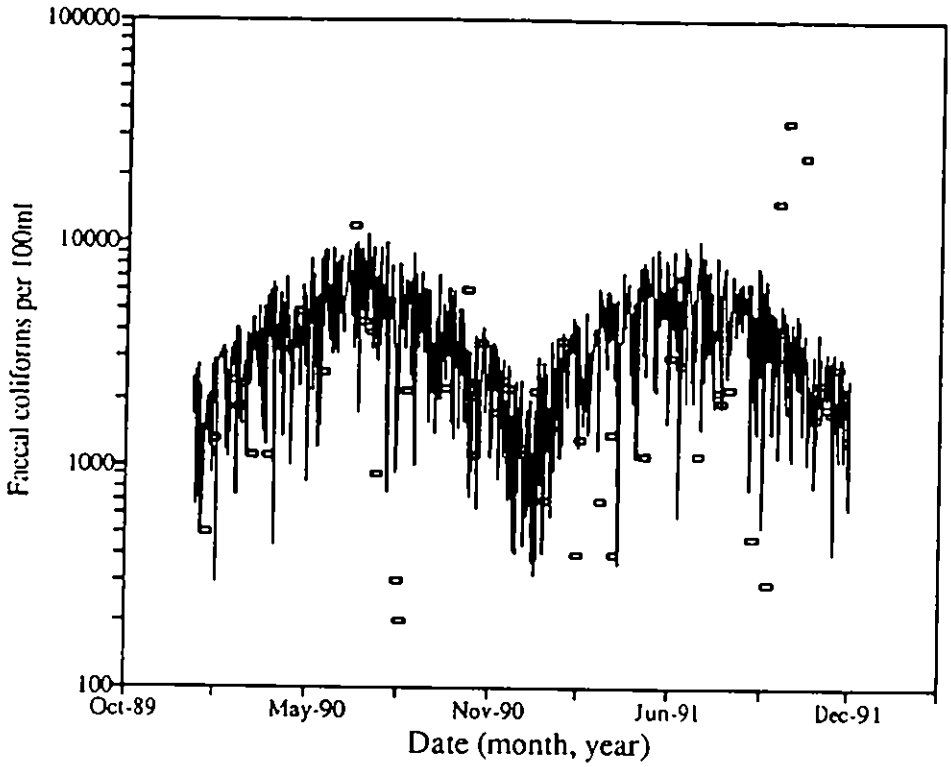


Figure 6.2 Raw data (□) and artificially generated time series (solid line) of faecal coliform concentrations for the River Exe at Thorverton.

6.3 Model calibration

Model calibration can be broken down into a number of discrete stages: Setting-up the flow routing equation and dispersive component. Setting the die-off terms in the channel-bed store and water column and finally the washout rate and settlement velocity.

The model version used in these runs uses a QUASAR formulation which gives a slightly different treatment of dispersion processes. In this formulation dispersion is achieved by treating each modelled reach as a number of continually stirred tank reactors connected in series. These cells are referred to as QUASAR blocks. The stretch of river between the two sites was divided into 6 x 1 km reaches, or 6 QUASAR blocks.

Flow routing within QUASAR is achieved using a simple power relationship; $velocity=a.Q^b$ where Q is discharge. Values of a & b for the River Thames gave successful results. Future field studies should be aimed at providing velocity measurements over a range of flows for the calibration of a and b.

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);

$$k_w = k_{w,min} + k_{rw}(1 + \cos[\pi + \omega t]) \qquad (6.1)$$

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 to good effect (Figure 6.3b).

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

The values of parameters were, as far as possible, derived from ranges of values found in the literature (Auer and Niehaus, 1993; Evison, 1989; Flint, 1987; Sherer *et al.*,1992), these were adjusted manually to achieve the model fit (Table 6.1). The value for settlement velocity chosen ($0.04m.d^{-1}$) is far lower than that found by Auer and Niehaus (1993). Their value of $1.17m.d^{-1}$ was for particulates in the range 0.45-10µm, 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 being the fact that the stronger mixing in river flow will result in a lower net accumulation compared to that in a lake. Although the actual gravitational settlement rate is unchanged, forward and upward motion within the flow will result in less settlement.

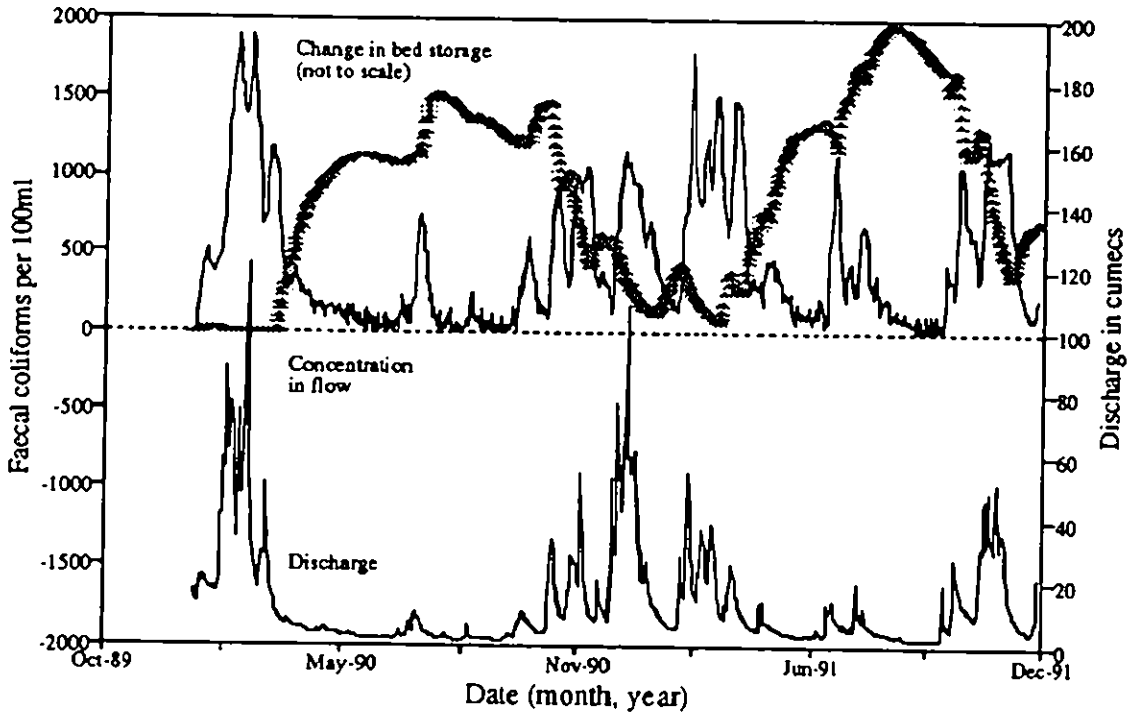


Figure 6.3a Time series of flow and modelled faecal coliform concentration and channel storage for the River Exe at Pynes water intake.

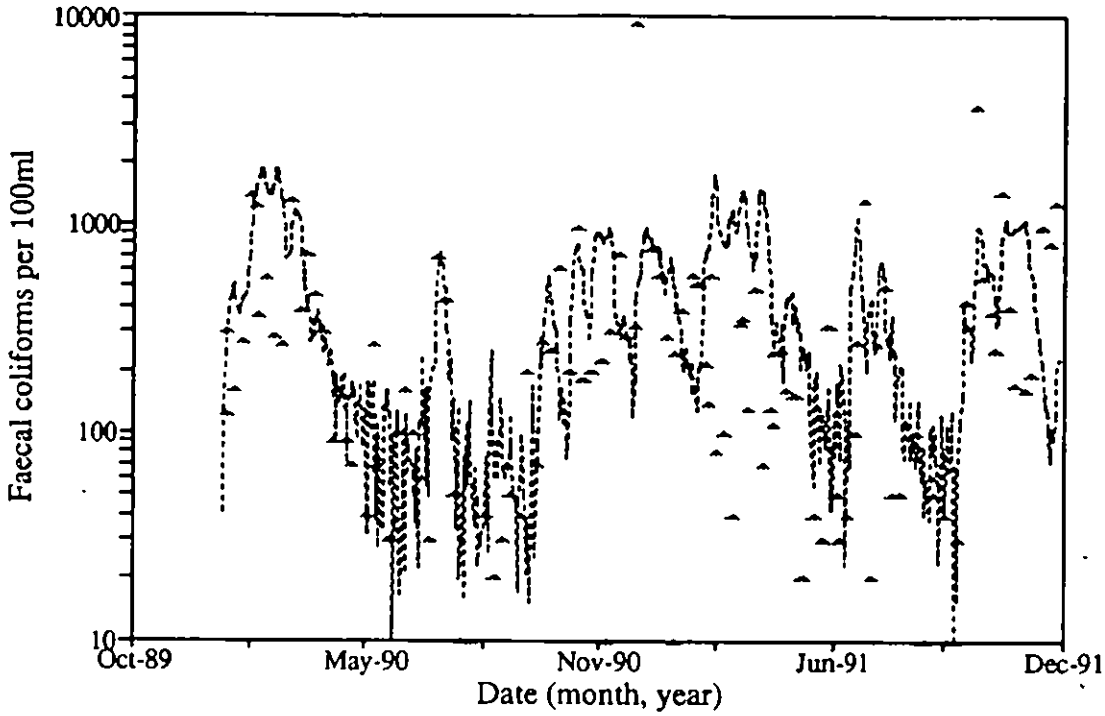


Figure 6.3b Modelled and observed time series of faecal coliform concentrations for the River Exe at Pynes water intake.

6.4 Model operation

Figure 6.3 a&b show the model fit to the observed data and time series of flow and change in channel storage. Figure 5.3a 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 6.4 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 6.5 shows the rapid reduction in channel storage with distance from the effluent discharge at Thorverton reflecting the reduced concentrations in the overlying water. Figure 6.6a&b show episodes of entrainment from channel storage these coincide with the flow events in the river.

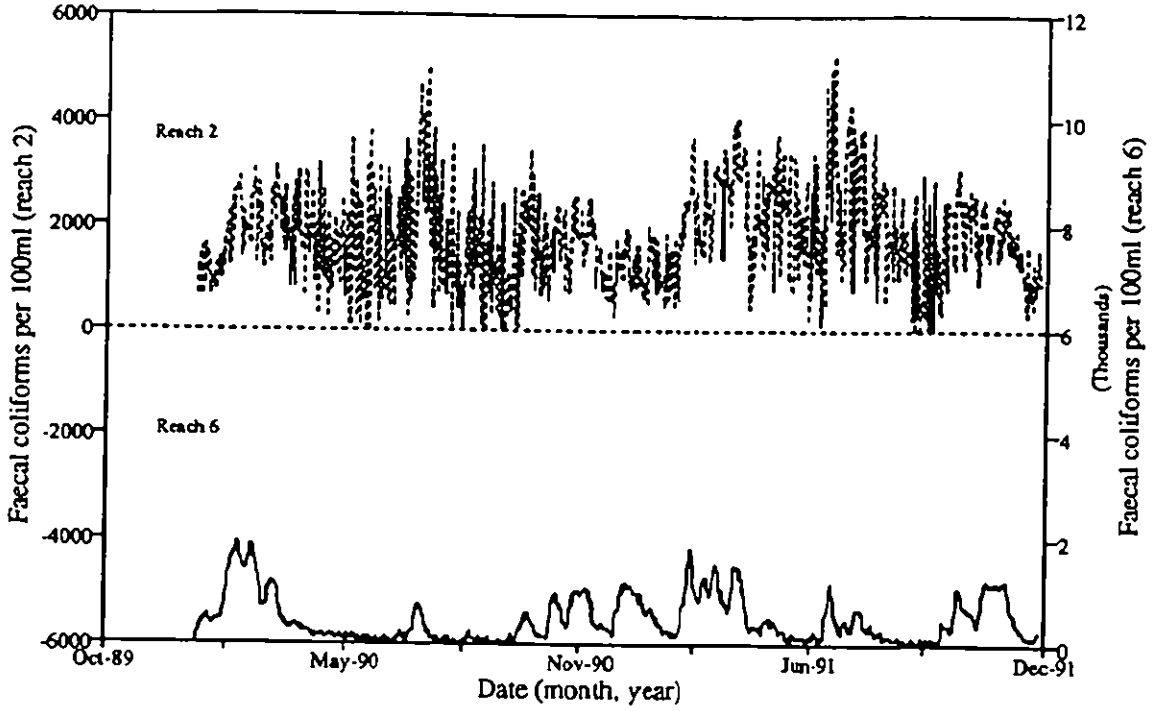


Figure 6.4 Time series of modelled faecal coliform concentration at the output of model reaches 2 and 6; each reach is 1km in length.

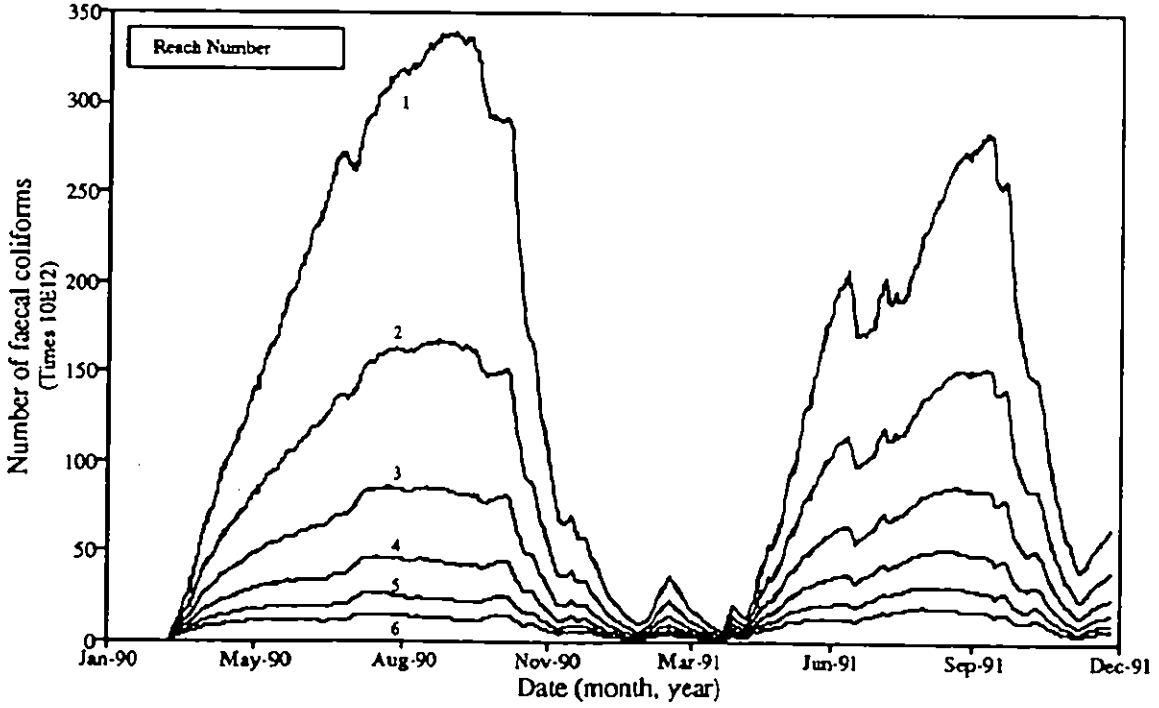


Figure 6.5 Time series of modelled faecal coliform numbers in channel storage for reaches along the River Exe between Thorverton STW and Pynes water intake.

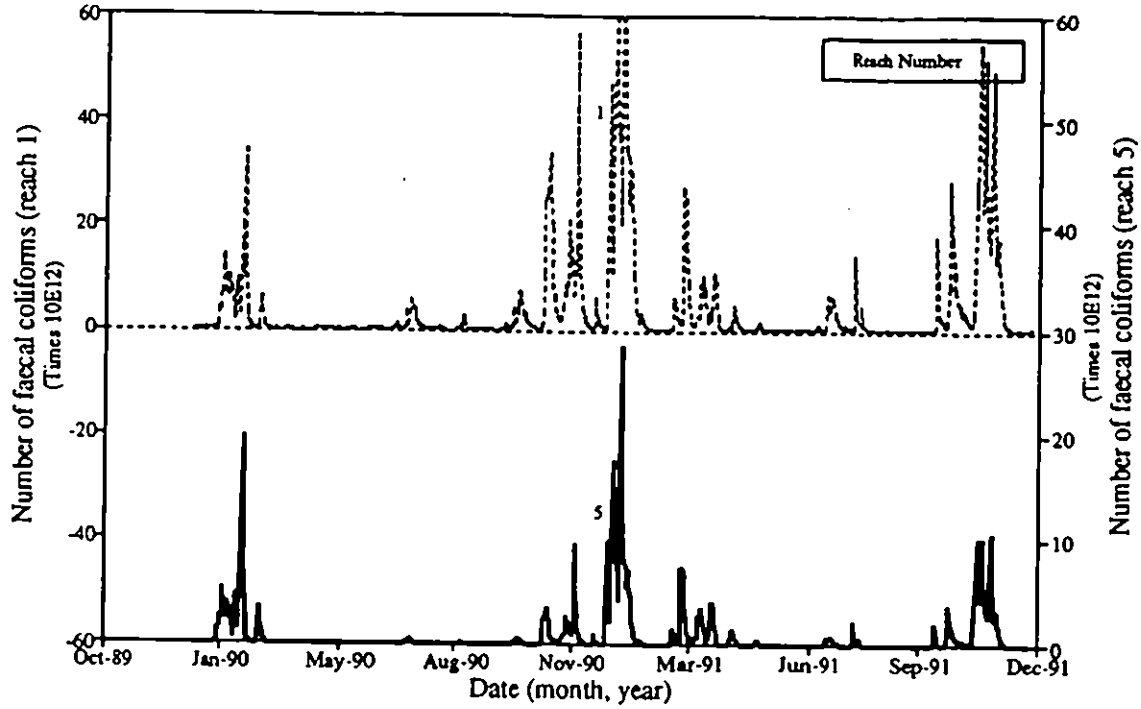


Figure 6.6a Time series of modelled faecal coliform entrainment episodes from channel storage during flow events; reaches 1 and 5 are shown.

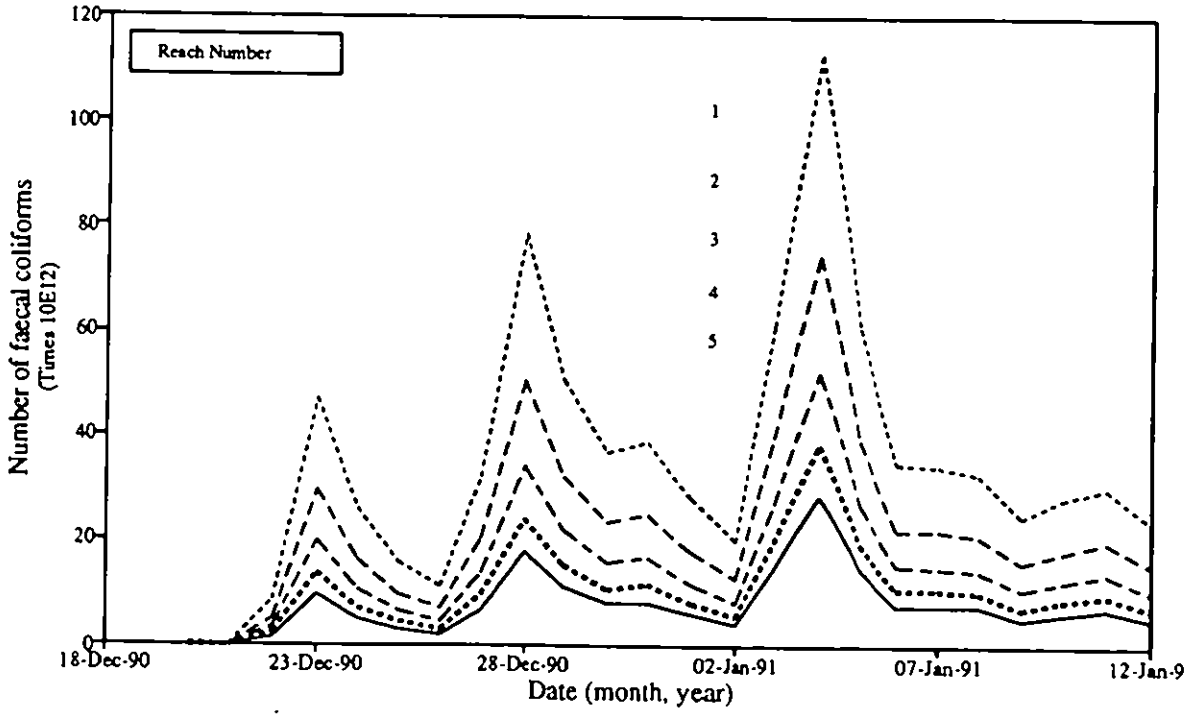


Figure 6.6b Time series of three modelled faecal coliform entrainment episodes over the New Year period 1990-91 in model reaches 1 and 5 of the River Exe between Thorverton STW and Pynes water intake.

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 6.7 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 Percentage change in $\log_{10}$ mean faecal coliform concentrations from the modelled values for various scenarios.

*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

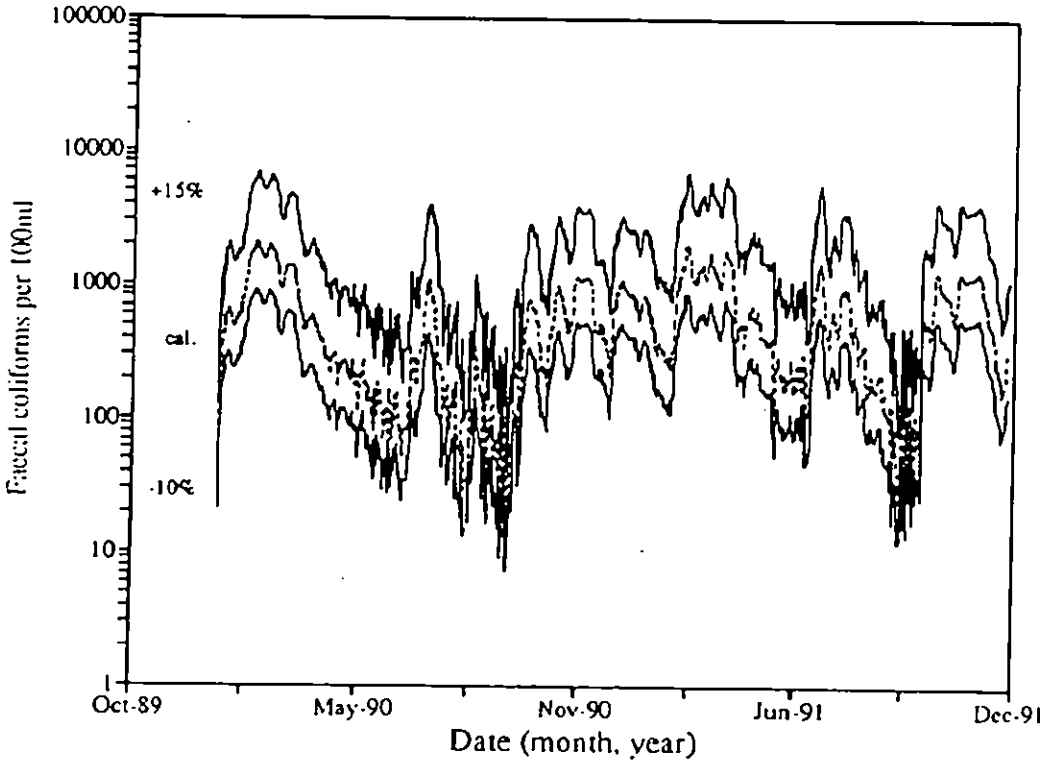


Figure 6.7 Time series of modelled faecal coliform concentration at Pynes water intake showing the impact of increased and reduced faecal coliform loadings.

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 faecal coliform concentrations has been examined. The tests examine only the impact of the change in flow on the modelled reach; the two scenarios examined are 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 upto 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 6.8). The flow series is adjusted using a simple transfer function of the form:-

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

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 input faecal coliform concentration 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 6.9 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 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 in this report suggests that higher temperatures and more sunshine hours will result in more rapid die-off. Drier conditions lead to more rapid die-off on the catchment surface and in soils. Conversely, wetter conditions will

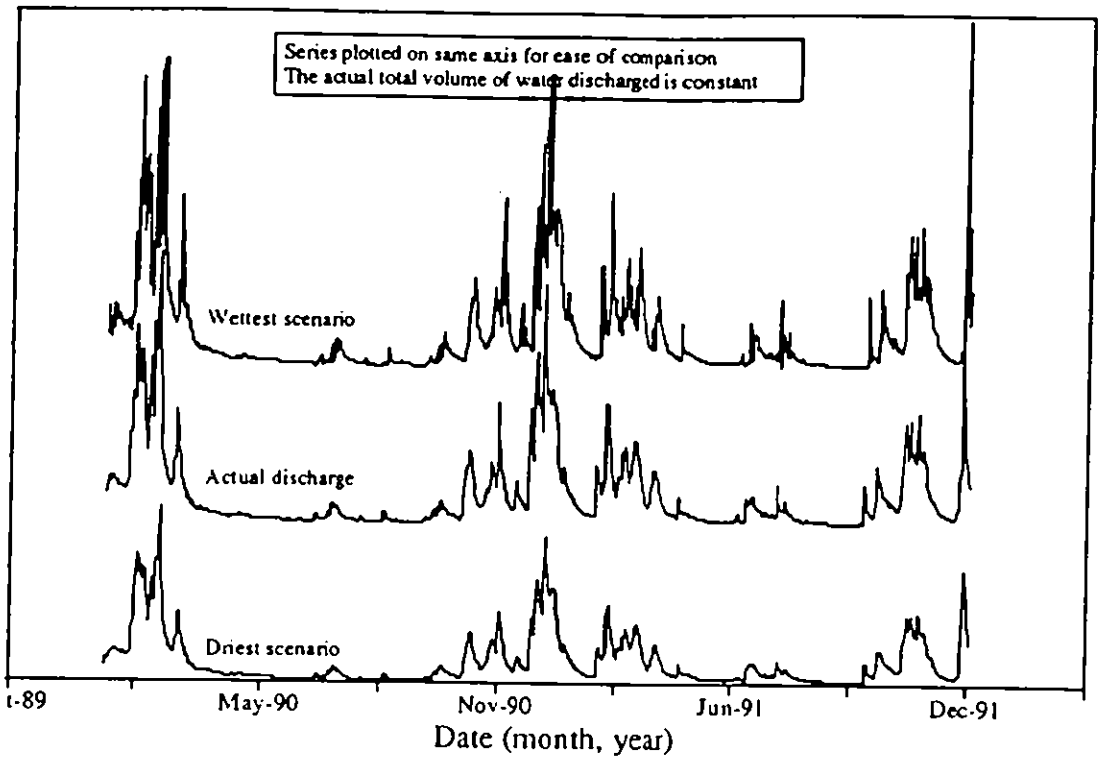


Figure 6.8 *Time series of actual and modified flows in the River Exe at Thorverton. The flows have been modified on the basis of CCIRG projections for the year 2050 (Arnell, 1992).*

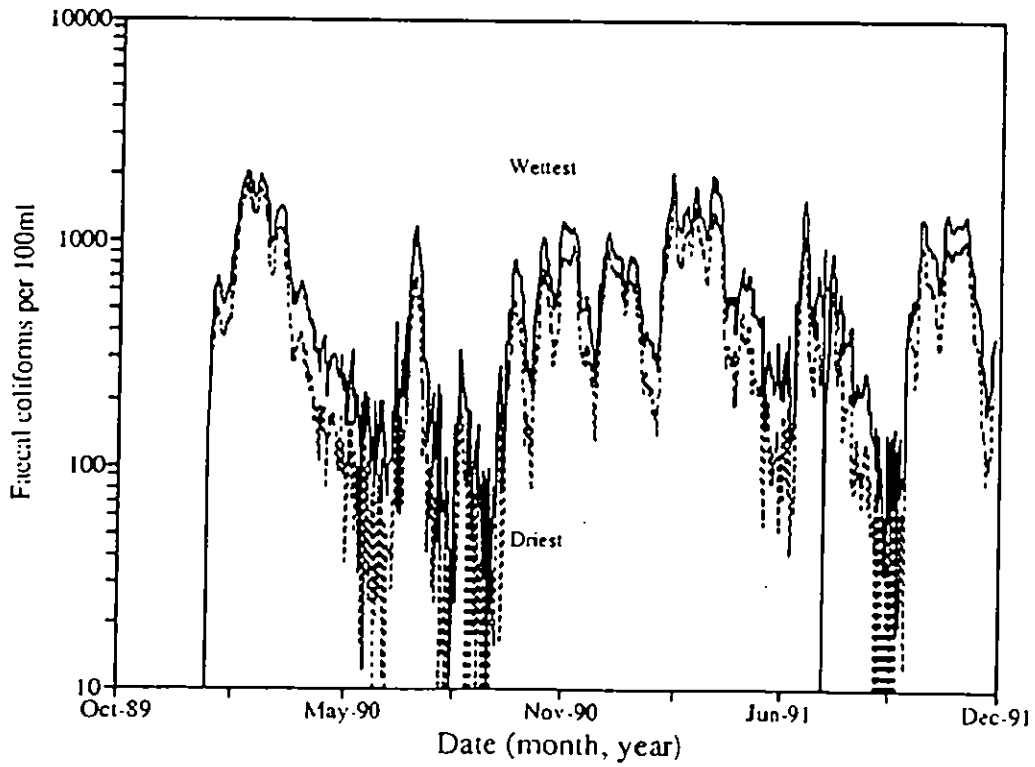


Figure 6.9 *Time series of actual and modified flows in the River Exe at Thorverton. The flows have been modified on the basis of CCIRG projections for the year 2050 (Arnell, 1992).*

result in lower river residence times resulting in less die-off and moister soils will improve survival and may lead to more rapid transport through the catchment. Increased cloud cover will result 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. This 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 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 it 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.

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7. RELATIONSHIPS BETWEEN FAECAL COLIFORM CONCENTRATIONS, LAND-USE AND CATCHMENT CHARACTERISTICS.

Examination of relationships between land use and faecal coliform concentrations in 25 UK catchments has shown consistent patterns.

Faecal coliform concentrations in 13 upland Welsh catchments and data from ADAS land use maps and derived from surveys of stocking practices and fertiliser use, showed that catchments with higher proportions of improved agricultural land, higher fertiliser use and livestock densities produce higher geometric mean faecal coliform concentrations than forested catchments. This 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 landuses. Regression analysis demonstrated a statistically significant negative relationship between the geometric mean faecal coliform concentration and the sum of % moorland and % miscellaneous natural woodland in the catchments. A significant positive relationship was found with the square root of the total area in km^2 of cultivated land, leys and broadleaf woodland. This may reflect the nature of the transport processes within the catchment, the length value produced by square rooting the areas relating to the contact length between the stream channel and the land areas delivering the contaminant organisms. This suggests the importance of near channel areas and future studies should be aimed at pursuing this interesting line of research.

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 and the study of travel-time effects, this would represent valuable enhancement of the results already presented.

Introduction

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 faecal coliform data for a set of upland catchments in Wales. The relationships between faecal coliform concentration and ADAS land classification data as well as information derived from a farm questionnaire survey of fertiliser use and animal populations in the target catchments are investigated. The second study examines the relationship between faecal coliform concentration and the ITE 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 for both 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) farm questionnaire survey following initial contact via NFU and FWU offices. The latter survey included variables such as fertilizer use (Limestone and NPK) and populations of sheep and cattle. Stock in 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 baseflow 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 was 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 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.

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

Catchment	Area (km^2)	Rainfall (mm)	Forest (km^2)	Grade IV (km^2)	Grade V (km^2)
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 Hamog	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

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 summarises fertilizer and stock data from the farm survey.

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 Hamog	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 $\alpha < 0.05$

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.

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

Table 7.4 Summary statistics for faecal coliform concentrations (count 100 ml^{-1}) in the study catchments

Catchment	Geometric Mean	Log_{10} Std 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 Hamog	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	1047.13	32

Table 7.5 Pearson correlation coefficients (r) (and significance (p)) between faecal coliform parameters and landuse variables

Variable	Geometric Mean	Log_{10} 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 $\alpha < 0.05$

Moderate geometric mean concentrations (10 - 100 FC.100ml^{-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 wildlife populations resident in such catchments. 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, $\log_{10}$ 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 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 1km 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; good water quality data, flow data collected at the same point in the river network and for ease of selecting coverages of land class data. The catchments 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 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 (HMSO, 1983).

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

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 $\log_{10}$ 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.

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

Number	Catchment	Area (km^2)	Rainfall (mm)	Mean flow ($m^3.sec^{-1}$)	Area under cultivation (km^2)
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.

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

Catchment	N	N log_{10} FC	Maximum	Arithmetic Mean	Geometric Mean	Std Dev	Log_{10} Std Dev	Skewness	Log_{10} 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

The most significant relationship found was that between maximum and mean faecal coliform 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 some aspect of cultivated land management generates a significant numbers of faecal coliforms.

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)

This result prompted a similar conversion of the other land cover types, but only produced improvements in fit of 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 discussed. 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 ($r^2 = 0.7251$, $p = 0.0004$) with no outliers.

Table 7.10 Correlation coefficients (r^2) (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	r^2 (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. [a,b](see Figure 7.1).

Land cover groupings:-
Group A1 = cultivation + leys + broadleaf woodland + built-up area (km^2)
Group A2 = cultivation + leys + broadleaf woodland (km^2)
Group B1 = moorland + miscellaneous natural woodland (% of catchment area)
Group B2 = moorland + miscellaneous natural woodland + heathland + bogs (% of catchment area)

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.

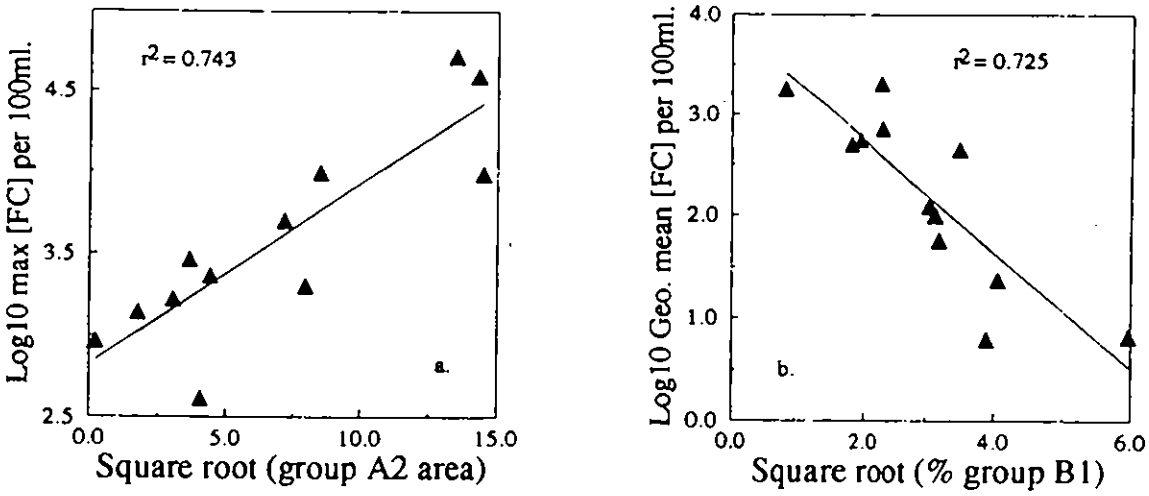

Figure 7.1 Plots of faecal coliform parameters against positively (a.) and negatively (b.) correlated groupings of land cover descriptors in 12 UK catchments.

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, as suggested in Section 7.1, the faecal coliform numbers derived from such areas being caused by wildlife. 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 head-water areas may be largely agricultural.

Having established these landuse 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 should 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 runoff originating on saturated areas which built-up along channel edges. It was therefore suggested, that due to the runoff 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 landcover map for the UK with 25 metre resolution. This comprises a total of 25 sub-classes and differentiates between land types ie, rough pasture, pasture, meadow, tilled land etc., and represents yet another possible enhancement to the results already presented.

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8. SUMMARY, CONCLUSIONS AND FURTHER WORK

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; and to develop an integrated predictive model of faecal coliform concentrations from point and non-point sources". The work can be integrated with research in areas 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 contained 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; to assist in the assessment of changes in effluent discharges or land-use on, for example, the health risk posed to recreators by a given river reach; assess the loadings of faecal contaminants to the marine systems and hence the impact on compliance of local bathing waters; and to assist drinking water abstractors to predict the timing, duration and magnitude of events of peak bacterial concentration in order to prevent the intake of large loadings of faecal contaminants.

A literature survey examined the key influences on the survival of faecal coliforms in streams and rivers. These were found to be light and turbidity, temperature and pH. Although faecal coliform survival is influenced by a wide range of other interactions and mechanisms; competition from, and predation by, the natural microbial community, particle interactions and nutrient effects. 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, moister catchment surface, shorter residence times in river reaches and protection from light and predation by particulates results in enhanced survival. Supply and transport factors also tend to result in higher winter concentrations; more rapid transport from the catchment surface, frequent operation of storm sewage overflows and scouring of settled organisms. In the summer months the effects are reversed, die-off through-out 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.

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 throughflow 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.

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 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.

Field experiments carried-out during the project provided enough information to improve considerably upon the results of Jenkins. Although the eventual model structure is similar to that used by Jenkins, fewer parameters are required, the model gives a better description of the physical processes occurring within the channel and simulates observed faecal coliform concentrations where existing models clearly could not.

The model structure and operation incorporates the following assumptions. The channel-store is distributed across the entire channel and that 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 be assumed once the flow recession commences. Further higher flows will still release organisms from storage.

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

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. 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 it also has scope for application to other water quality determinants.

Further developments of the model might consider extending it's scope to modelling particulates or other particle associated contaminants such as metals, hydro-carbons or radio-nuclides, as well as other microbiological determinand 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.

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 fertiliser use were analysed. Catchments with higher proportions of improved agricultural land, with higher fertiliser 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 landuses.

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 valuable enhancement of the results already presented.


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