

# **Faecal Indicator Organism Modelling (FIO): Application to Motueka River**



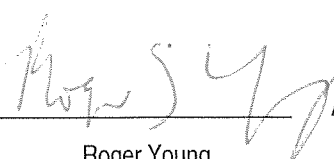
# Faecal Indicator Organism Modelling (FIO): Application To Motueka River

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# 1. MODEL BACKGROUND AND DESCRIPTION.

## 1.1. Introduction

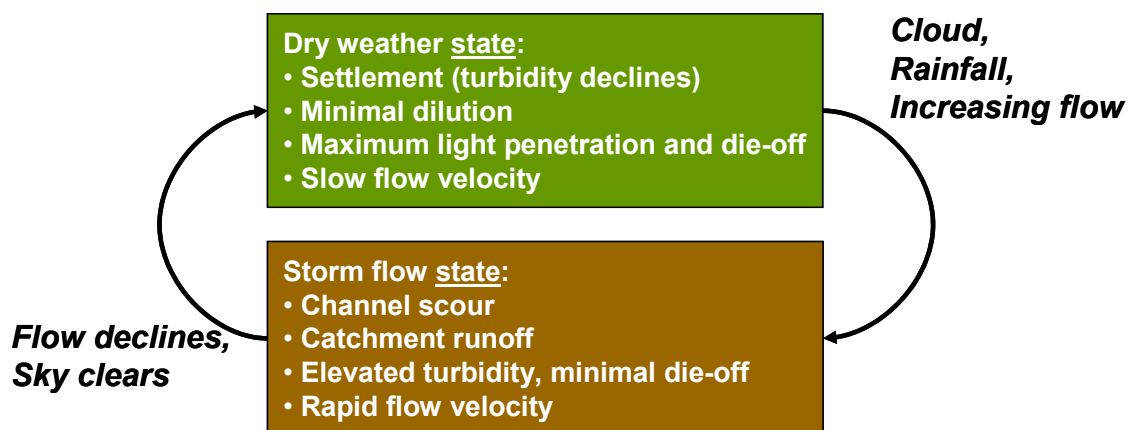
This report presents a model for faecal indicator dynamics simulation in rivers developed during doctoral research in the UK (Wilkinson 2001). The model has been applied to the Motueka River. Model setup and results are described.

## 1.2. Background

The original work to develop this model was funded by the United Kingdom Department of Environment in 1992 with the intention of determining the key sources of faecal contamination, drivers of faecal indicator die-off and transport, and to develop an improved model for faecal indicator concentrations in rivers.

### 1.2.1. Faecal indicator dynamics in rivers

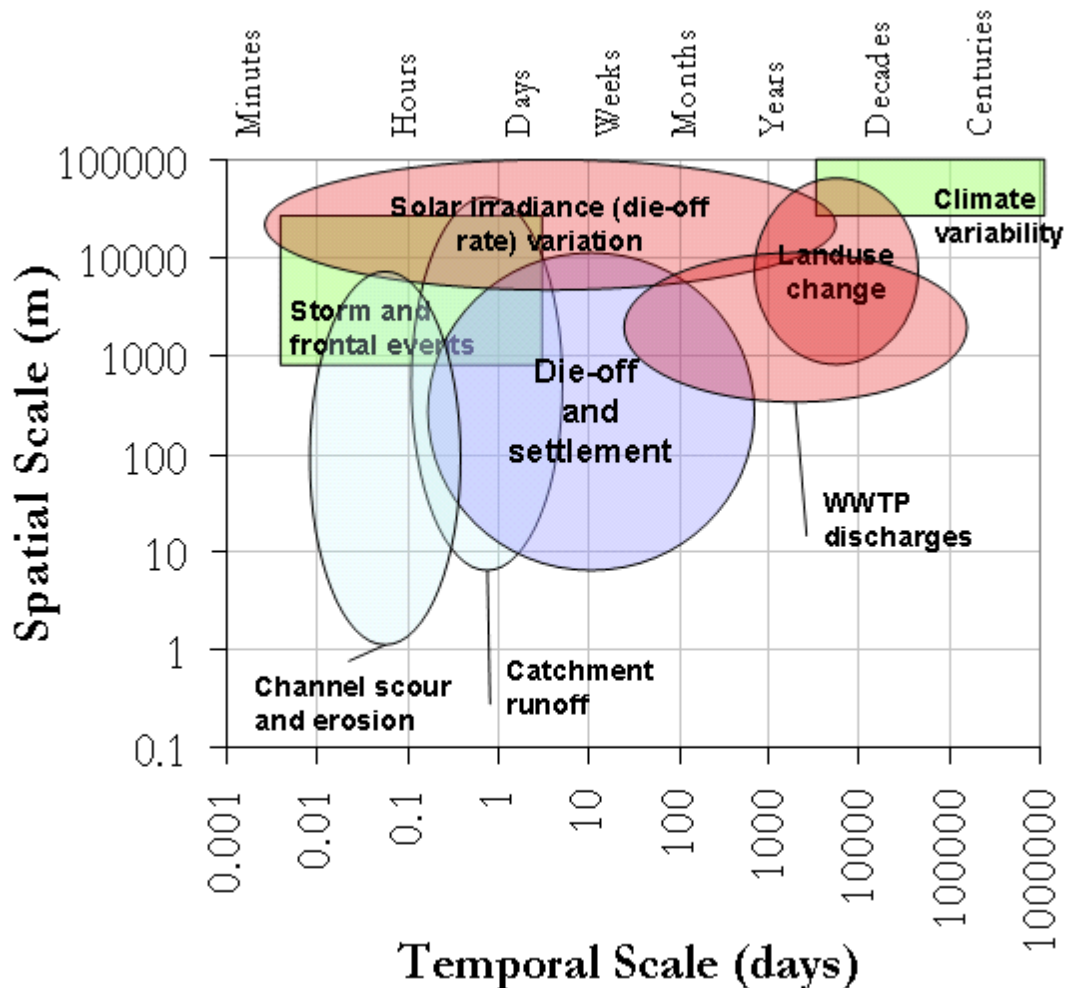
Faecal indicator dynamics in rivers are determined by two groups of processes associated with dry and wet weather conditions. Figure 1 summarises the two groups and their relationship as a temporal progression from dry to wet to dry conditions.



**Figure 1.** Schematic representing the transition from wet to dry weather processes associated with faecal indicator dynamics in rivers.

During dry weather transport of faecal matter is minimal, water tends to be shallow and less turbid than during wet weather, and water travel velocities are slow. Die-off dominates the variations in faecal indicators.

After rainfall, river flows are usually elevated and more turbid, there is cloud cover and hence reduced light. Travel times are enhanced and die-off is minimised. Transport mechanisms dominate the dynamic variations in indicators at this time.



**Figure 2.** A Stommel diagram adapted to represent the temporal and spatial scales associated contaminant microbial dynamics in catchments (from Wilkinson *et al.* 2006).

Figure 2 represents an indication of the temporal and spatial scales allied with the various processes associated with contaminant microbial dynamics. The most pertinent aspect of the diagram here is the difference between the temporal dynamic of channel scour and erosion and die-off/settlement mechanisms.



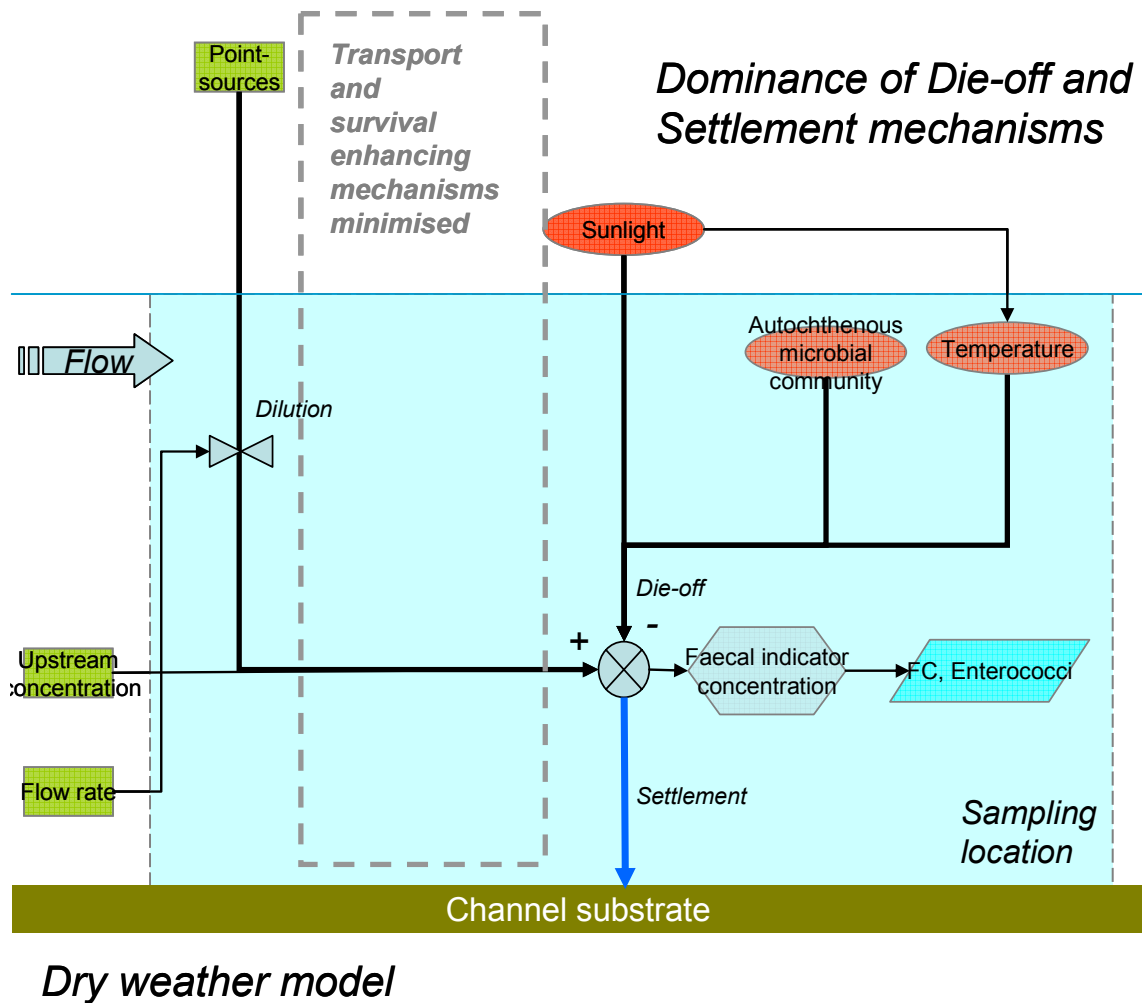
### **1.2.2. Faecal indicator die-off**

The model used in this study embodies the main processes indicated above. As already stated die-off dominates dry weather behaviour (Figure 3). This section describes the background to the die-off model that is used. Appendix 1 provides an extract from the PhD thesis of Wilkinson (2001) that outlines the components of the die-off modelling used here.

The die-off model is primarily driven by solar radiation. The secondary components of the die-off model are:

- Water temperature - moderates darkness die-off;
- Turbidity – reduces light penetration into water column and coats bacterial cells with protective clays (Brettar & Höfle 1992);
- Water depth – reduces average light intensity;
- Velocity – faster flows, reduced residence time.

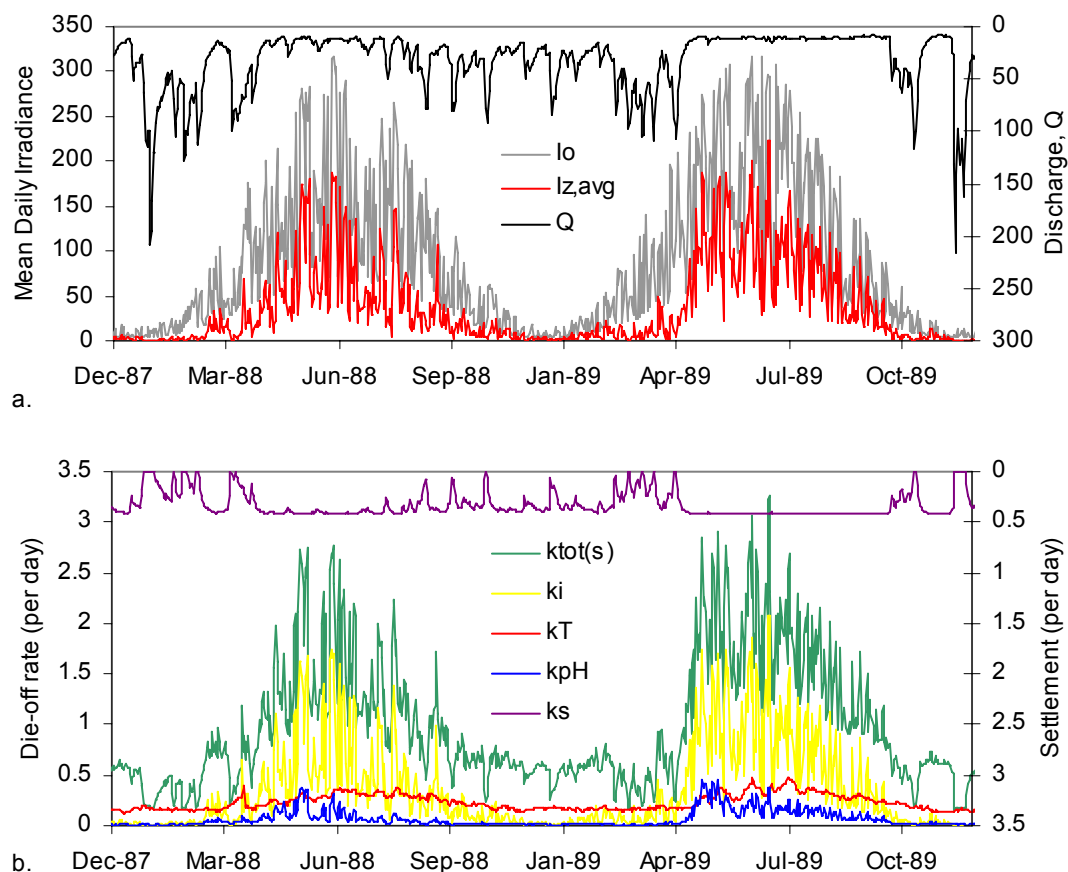
Essentially die-off is maximised during dry weather and minimised during storm event river flows. During and following flow events settlement of particle-associated bacteria removes organisms from the water column. In the sediments they can persist for extended periods and can be resuspended by subsequent flow events (*e.g.* Wilkinson *et al.* 1995, 2006; Muirhead *et al.* 2002). Settlement acts as an “apparent” die-off. Organisms appear to have died because they are no longer present (see Appendix 1 for settlement equations).



**Figure 3.** Dominant processes driving faecal indicator dynamics during dry weather.

In the model application to the Motueka River, solar radiation for the Nelson Airport weather station is used. Other applications of the model have shown that meteorological data from relatively distant sites can still provide adequate modelling results (Wilkinson 2001), although local micro-meteorological influences such as local topographic effects mean that data from nearby sites are desirable.

The model die-off rate is effectively a time-variable parameter. The time series of  $k_{\text{tot}}$  (total die-off) is setup in a separate spreadsheet. Water surface die-off is calculated initially, and the progressive influences of water depth and turbidity are applied to adjust for depth-averaged conditions (Figure 4).



**Figure 4.** Examples of the variation in faecal coliform die-off in the Welsh River Dee, UK, a. estimated mean daily surface irradiance (top),  $I_0$ , and depth averaged irradiance,  $I_{z,\text{avg}}$ , (*i.e.* reduced by water depth and attenuation by particulates). Discharge is plotted to demonstrate the timing of the maximum reduction in light penetration (Irradiance values are in  $\text{W.m}^{-2}$ , discharge is in  $\text{m}^3.\text{sec}^{-1}$ ). b. Estimated die-off,  $k_{\text{tot}}(s)$ , and the component die-off time series,  $k_i$ ,  $k_T$ ,  $k_{pH}$  and  $k_s$ .

Die-off is treated as a global variable in the model. Water surface die-off is used for the land surface and is arbitrarily doubled with the intention of accounting for drying-out (this is not validated, and cow-pats can maintain viable organisms for extended periods). The absolute numerical accuracy of the die-off rate is not greatly important. It is the dynamical variation in die-off that accounts for the dry weather dynamic variation in faecal indicator organism (FIO) numbers at a point of interest.

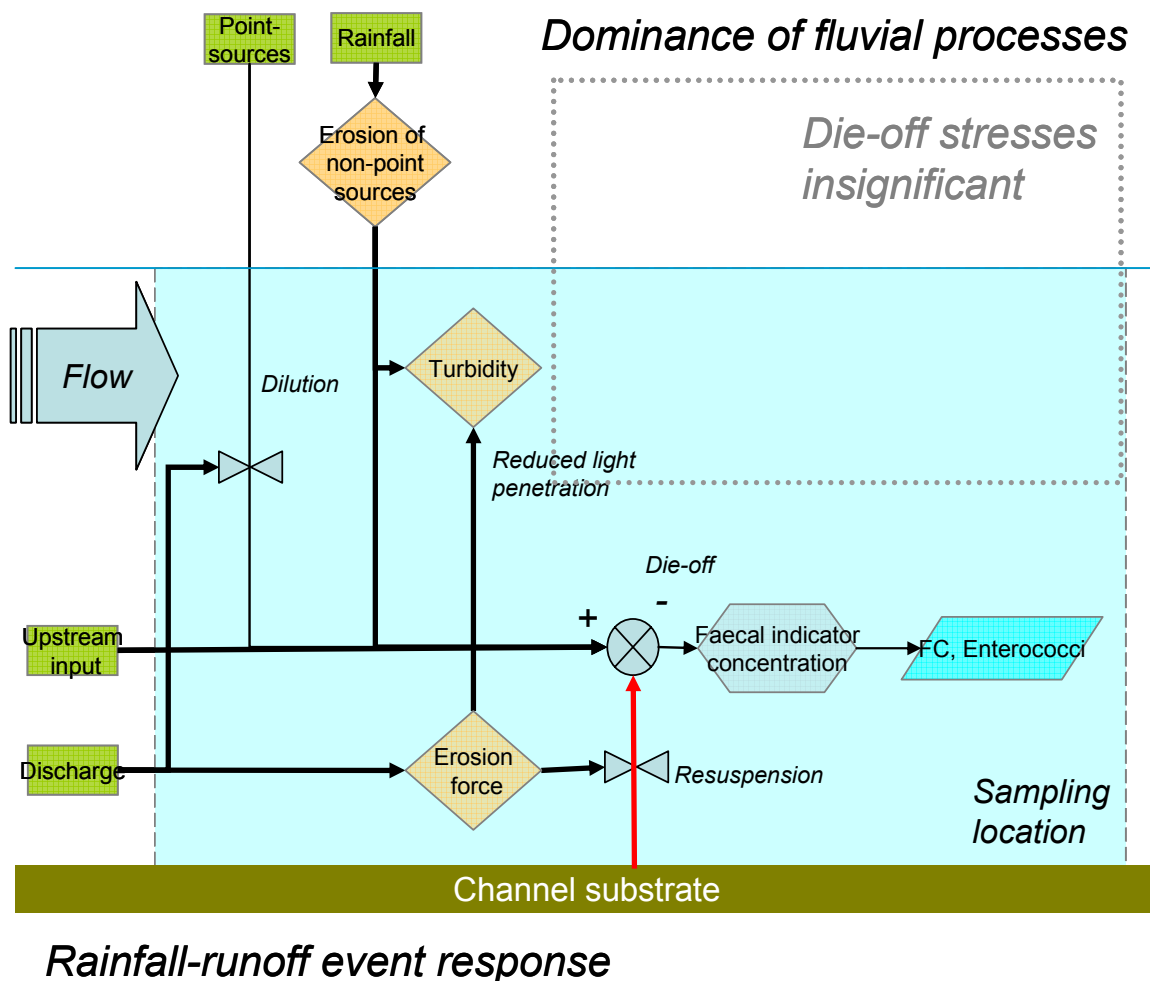
This approach may seem inadequate to the purist, but it provides a pragmatic and successful approach to this difficult issue and good simulation results of faecal indicator numbers can be achieved.

### 1.2.3. *Faecal indicator wet weather dynamics*

As suggested above with wet weather there is a switching from the dominance of die-off encouraging mechanisms to fluvial processes that enhance flushing of stored organisms and minimise die-off to the extent that it is negligible (Figure 5).

Early studies in the United States (Morrison & Fair 1966; Kunkle & Meiman 1968; McSwain & 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 & McDonald (1980) examined the nature of the changes in concentration during individual flow events and the significance of different sources of organisms in greater detail. An intensive programme on the River Washburn in Yorkshire showed background total coliform concentrations to vary widely (2-600 MPN per 100 ml) and significant increases in concentration during the rising limb of all hydrograph events. Artificial hydrographs, generated between impoundments on the Washburn demonstrated that the release of organisms from within the channel could produce coliform peaks in the order of those observed during natural flow events. Using a velocity hydrograph, to eliminate bank sources, the source of organisms was shown to be a direct bacterial or indirect sediment/bacterial bed supply mechanism (McDonald *et al.* 1982). Further releases demonstrated accumulation of organisms in the flow with propagation and the finite nature of channel supply of organisms.

Wilkinson *et al.* (1995, 2006) suggested that there were three types of bacterial flushing mechanisms associated with steeply rising flood flows: wave-front entrainment; kinematic overrun lift and burst/sweep erosion associated with river turbulence at high flow. Modelling results (Wilkinson 2001) also suggested that it was near-field bacterial entrainment that contributed most to the observed bacterial peaks at a point in a river *i.e.* material entrained further away did not appear to contribute greatly to the concentration at the point of interest. This suggestion appears to be supported by modelling results for the Motueka River and is discussed further below.



**Figure 5.** Dominant processes driving faecal indicator concentrations in rivers during storm events.

#### 1.2.4. Mixing and dispersion in relation to microbial transport

Wilkinson *et al.* (1995) briefly consider the process of river mixing in relation to faecal indicator dynamics. Recent research into mixing processes in rivers has demonstrated that the advection-dispersion model does not provide a realistic representation of mixing processes, and moreover due to numerical dispersion is prone to errors in its application (e.g. Henderson-Sellers *et al.* 1988). The mixing volume/zone approach to dispersion modelling in rivers gives a better fit to tracer data and does not suffer from numerical losses, and is computationally efficient (see Wallis 1993; Green *et al.* 1994). The mixing volume model is used for pollution spill location prediction usually during quasi-steady flows. In this instance the mixing volume principle is applied to dynamic flow situations. Green *et al.* (1994) demonstrate that dispersive action is slightly reduced at high flows, and the full mixing volume equations for dynamic flows are significantly complicated (Price *et al.* 2000; Wilkinson 2001):

$$\frac{dX_t}{dt} = \frac{Q_t}{Ve_t} U_{t-\tau} - \left\{ \frac{Q_t}{Ve_t} + \frac{1}{Q_t} \frac{dQ_t}{dt} + \frac{2}{Ve_t} \frac{dVe_t}{dt} \right\} X_t$$

and the following simplification provides a satisfactory approximation:

$$\frac{dX_t}{dt} = \beta(U_{t-\tau} - X_t)$$

and in its finite difference backwards approximation this becomes

$$X_k = bU_{k-\delta-1} + aX_{k-1}$$

where, for a conservative solute  $a = 1 - b$ , *i.e.*  $a + b = 1$ , and for most river mixing applications  $a \gg b$ .  $\tau$  is the advective time-delay for upstream transport.

The mixing volume model is effectively a “low-pass filter” or “smoothing algorithm” *i.e.* it simulates the stirring and mixing of a flowing river such that in a large river variations in a solute will be relatively small over short time scales. With faecal indicators the variability between samples taken over short time intervals is often a function of the enumeration method and this variability can be reduced by a factor of 1.73 by using triplicate enumeration (Fleischer & MacFadden 1980; Fleischer *et al.* 1990). In the current application to the Motueka River, small scale variations in FIO numbers do appear to be smoothed out by river mixing.

### 1.3. Model formulations for the Motueka River

The preceding sections gave an overview of some of the key components of the faecal indicator model. This section outlines the model formulations investigated for the Motueka River.

Wilkinson (2001) presents the full FIO model with all of its components in an application to the Welsh River Dee in the United Kingdom. Appendix 2 reproduces material from the PhD thesis and demonstrates the progression from the full model to a much simplified version of the model which is applied here to the Motueka River. A number of model variants have been investigated which operate on similar principles but with slight variations on the configuration of components.

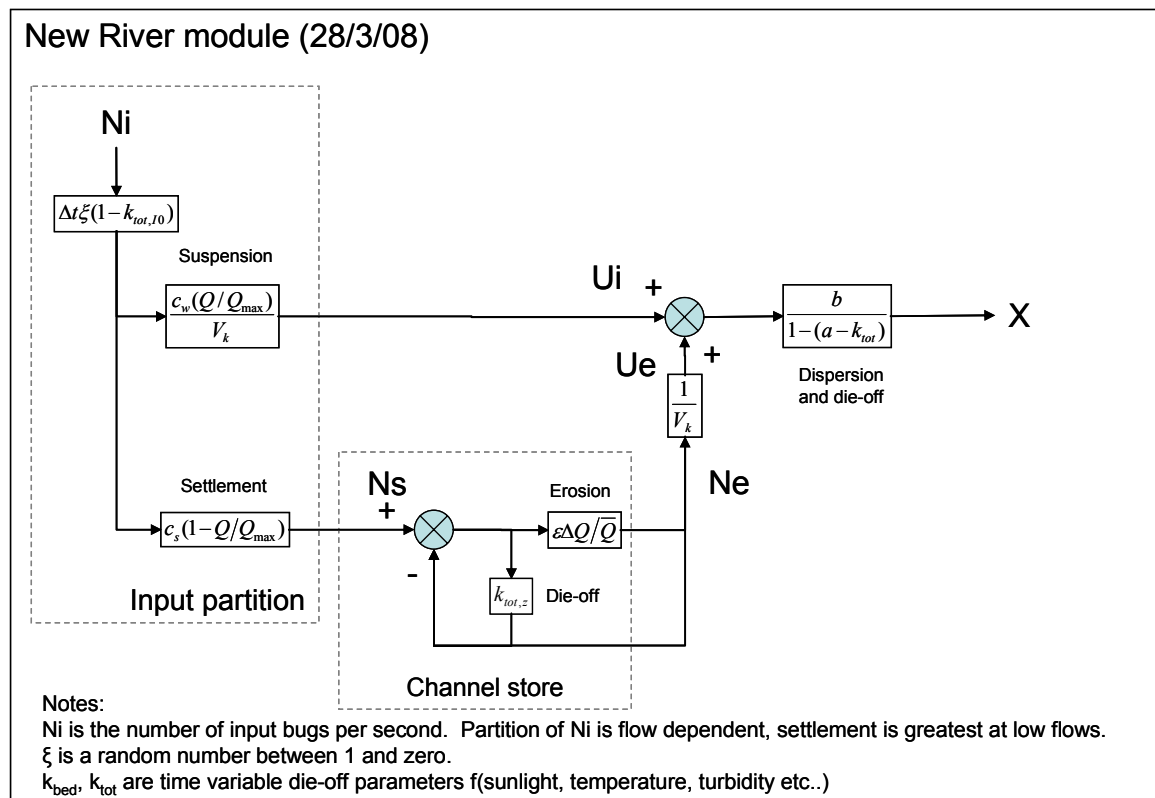
#### 1.3.1. Modelling objective

Woodman’s Bend on the Motueka River was the location that received the most attention. The purpose of the modelling exercise was to generate a realistic time series of faecal indicator data to feed into Tasman Bay in order to investigate the impact of microbial contamination on shellfish harvesting areas. The FIO discharge data was linked with a particle tracking model

that simulates the transport, dispersion and die-off of indicator organisms from a storm flow plume as it passes out into the bay.

In addition to this primary objective, a secondary objective has been to investigate FIO modelling in relation to potential source areas within the catchment. For this purpose a semi-distributed version of the model was set-up and is described below. It is worth mentioning that the modelling results indicated that the majority of FIO entrainment and delivery by storm events was associated with flushing of the channel in the lowest reaches of the catchment. The tendency for channel bed supply dominance in low lying river reaches was suggested by Hunter *et al.* (1992), and the observations of Nagels *et al.* (2002), Collins (2004) and Muirhead (2004) have tended to confirm this. A similar conclusion was drawn from the modelling results presented in the PhD thesis by Wilkinson (2001).

### 1.3.2. Model structure and configuration



**Figure 6.** Mathematical schematic of the FIO river flushing model for the Motueka River at Woodman's Bend.

Figure 6 shows the simplified model used to simulate FIO (*E. coli*) time series of concentrations at Woodman's Bend. The model has a single input term for numbers of organisms entering at each time interval, this constant number is put through a filter to randomise the value and apply die-off. The resulting numbers are then split into those that

settle to the channel store and those that remain in suspension. The proportion that settle-out is flow dependent. The greater the flow the larger the proportion that remains in suspension.

Organisms that have settled into the channel store are subject to die-off appropriate to channel floor conditions, and are resuspended when a suitable magnitude storm event passes by. Resuspended organisms are diluted into the water column and mixed with those already in suspension. Die-off is applied throughout the water column. It is important to remember that FIO concentrations are conventionally represented as the numbers of organisms per 100 ml volume, so an extra factor of 10,000 division has to be applied when expressing output concentrations. This is easily overlooked.

A multiple reach version of the model has been built and this further enhances the model fit and utility, in that it is possible to investigate the influence of upstream changes in faecal inputs. Further work is needed to refine details of this model. The model fit, however, is excellent (see Section 2.3).

## 2. MODEL CALIBRATION AND RESULTS

The FIO flushing model used here has been applied to the Welsh River Dee in the United Kingdom and the Motueka River. The model is calibrated in a manner which is subtly different to conventional modelling approaches. The temporal dynamic variations in indicator numbers are found to be determined by a combination of fluvial and die-off dynamics. The die-off variability is predominantly a function of solar radiation, turbidity and water depth (Figure 4). The system model is simply constructed, and driven by these environmental variables. There are few parameters and the input number of bugs is effectively one of the parameters.

### 2.1. Parameterisation

The model has fixed parameters and adjustable parameters. The fixed parameters are associated with hydrology and channel geometry *i.e.* relating to reach volume, water depth, velocity and so on, and also with die-off mechanisms. The die-off models and their parameters are described in Appendix 1. The die-off model is effectively a fixed system, assumed to be reasonably well characterised, and hence these parameters and those for river flow and channel geometry are not varied when fitting the model to the observed data.

The parameters that are varied are:

- $N_i$ , input of bugs to the system;
- $c_s$ , input settlement coefficient (and hence  $c_w = 1 - c_s$ );
- $\varepsilon$ , the erosion coefficient;



- $a$ , the dispersion coefficient ( $b = 1 - a$ );

Wilkinson (2001) investigated a range of drivers for erosion/entrainment of organisms from the channel store. The change in river discharge is the key driving variable, but the way flow is used can determine the response of the model. Flow-related variables that were investigated included velocity and stage (water depth). River discharge  $Q$  was found to give the best results and was applied by testing the change in discharge relative to the median flow.

$$\varepsilon \frac{Q_k - Q_{k-1}}{Q},$$

This tends to bias the entrainment towards greater flows and provided a better fit of the entire time series from March 2001 to July 2004. The events for which there are observations occur on some of the highest flows during the period. With the entrainment function presented, few flow events other than those which were actually monitored result in a significant model response (Figure 7). This is either a happy coincidence, or the model is under-representing *E. coli* flushing on the smaller events.

Table 1 presents the model parameter values used for calibration. The parameters have a physical interpretation. The input partition coefficient ( $c_s$ ) determines the proportion of input organisms that settle-out to channel storage during dry-weather flow; the value 0.9 indicates that 90% of input organisms settle-out, and 10% remain in suspension (during low flow). In Figure 6 the flow-driven function that partitions input organisms to settlement or suspension is shown to be a function of discharge  $Q$ , in reality the values used are  $\log_{10}Q$  and  $\log_{10}Q_{\max}$ .  $\log_{10}$  transformed discharge is used because this linearises the function, since discharge tends to vary in a log-normal manner. Untransformed discharge was used initially, but this was found to be too unreactive across the range of observed flows, and the function only operated at higher flows. The energy required to maintain particles in suspension is far less than that required to overcome boundary layer shear stress, hence it seems appropriate that the entrainment function operates differently than that which merely reduces settlement from the water column.

**Table 1.** Model parameter values for calibration of *E. coli* flushing events in the Motueka River between March 2001 and July 2004.

$N_i$ , input numbers (organisms per second)	$c_s$ , input settlement coefficient (0 to 1)	$\varepsilon$ , erosion coefficient	$a$ , dispersion coefficient (0 to 1)
6500,000,000	0.945	0.02	0.91

### 2.1.1. The calibration procedure

This is a conceptual model which is semi-physical in nature, in that the structure and parameters are based on the simulation of perceived or observed physical phenomenon. Purely data-driven models have an abstract structure and parameters without physical interpretation. Often in modelling a numerical approach is used for model optimisation with an automatic

routine that tunes parameters to achieve a minimum of some objective measure of model fit, this is often the case with hydrological models. This kind of optimisation may produce a wide range of differing calibration parameter combinations that result in a similar model fit, the combination of model parameter values may make sense relative to the understanding of the system function, or they may seem quite unlikely and disconnected from perceived reality.

In the case of the model applied here, there are only a small number of parameters to adjust, and the influence of each parameter is immediately obvious and has an easily understood physical interpretation. The fit of the *E. coli* model is measured using  $Rt^2$  which is a statistic of goodness of fit used in time series modelling (see Young & Benner 1991; Price *et al.* 2000) and tends to highlight errors in fitting the peak concentrations and the timing of the faster dynamics of the response;

$$Rt^2 = 1 - \frac{\sigma(Y_k - \hat{Y}_k)^2}{\sigma(Y_k)^2}.$$

$Y_k$  is the observed value and  $\hat{Y}_k$  For the best fit  $Rt^2$  approaching 1 should be aimed for. An  $Rt^2$  of greater than 0.95 is considered a good fit. Only pairs of values where there is an observed record can be tested, in this case only 271 hourly aggregated observations (data were averaged to the nearest hour) out of approximately 30,000 records.

A second measure of fit is provided and is specifically intended for the improvement of the fit of the low flow period (Jakeman *et al.* 1993), however since there are so few values for the Motupiko River this measure is almost meaningless. The measure is the Relative Mean Absolute Error (RMAE) and is given by;

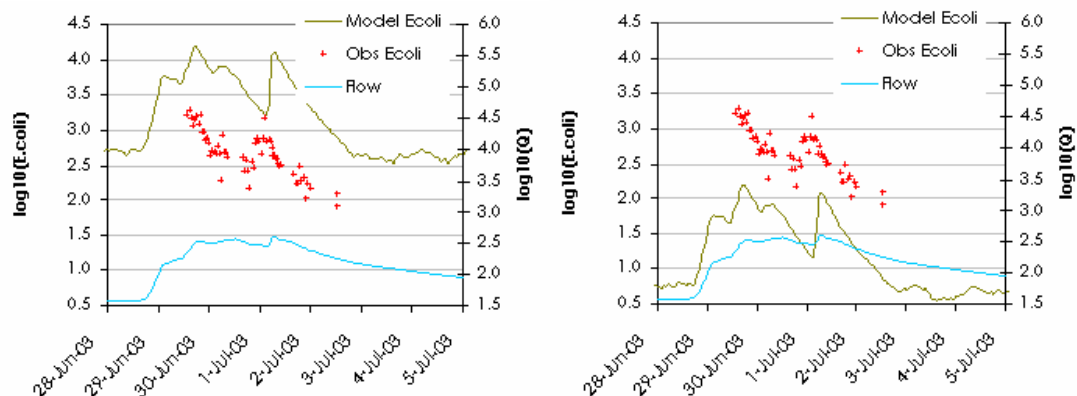
$$RMAE = \frac{1}{n} \sum \left| \frac{Y_k - \hat{Y}_k}{Y_k} \right|$$

RMAE should be minimised for the best fit.

Manual optimisation is undertaken and an indication of the effect of each parameter is given below:

### Input loading

The only influence of adjusting the input loading is to raise and lower the overall output distribution on the y-axis. The model was calibrated with  $N_i=6.5$  billion cfu per second (Bcfu), Figure 9 demonstrates the impact of choosing  $N_i=6.5$  Bcfu and  $N_i=0.65$  Bcfu.



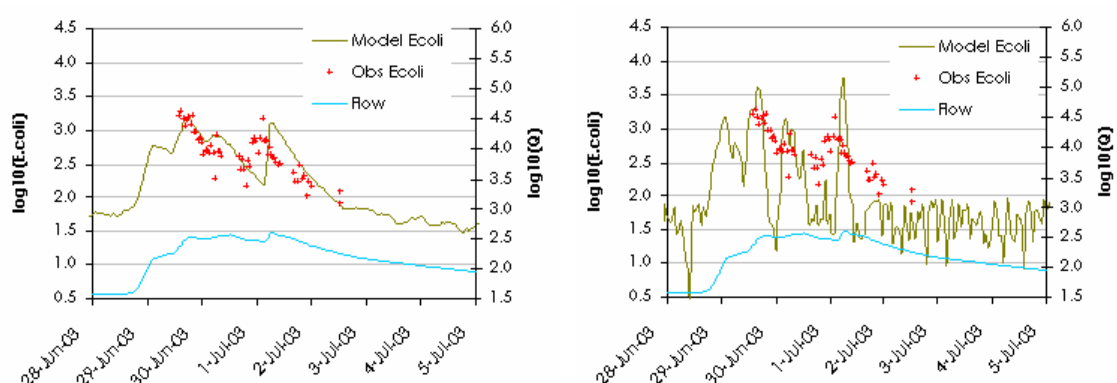
**Figure 7.** Comparison of varying the input faecal loading between  $N_i=65\text{Bcfu}$  and  $N_i=0.65\text{Bcfu}$ .

### Random Input Switch

In order to view the model results with and without random variation of the input faecal numbers a switch has been added with the calibration parameters. For random input the switch is set to 1, for non-random input the switch is set to 0. Since the random “variable”  $\xi$  is effectively “white noise” varying between 0 and 1, it has an average value of 0.5, therefore when random input is switched on, a value of  $2\xi$  has to be used to achieve the same output for a given input value  $N_i$ .

### Mixing and dispersion

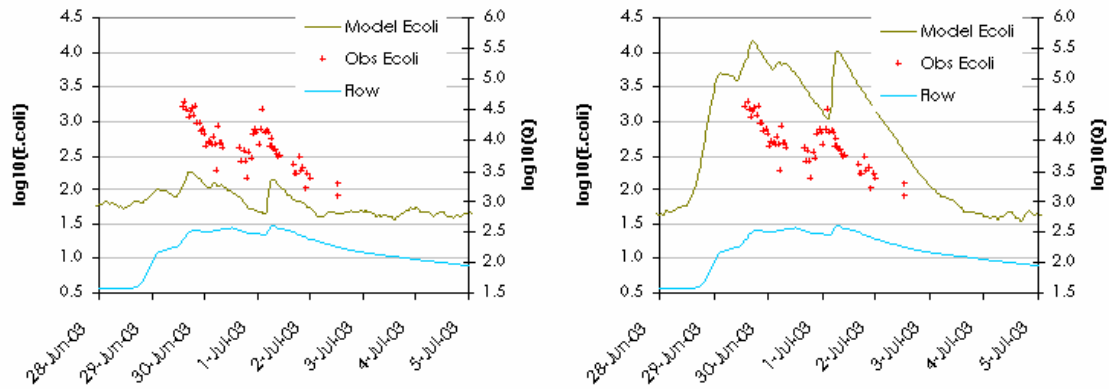
The mixing and dispersion parameter  $a$  adjusts the smoothing of the output time series. The model was calibrated with  $a=0.91$  which is the kind of value expected for a large river system,  $a=0.2$  indicates the response of a small scale system with a relatively quick time-constant and little mixing or dispersion *e.g.* a small stream with many cattle crossings and direct faecal deposition immediately upstream of the monitoring site.



**Figure 8.** Comparison of varying the mixing and dispersion parameter between  $a=0.91$ ,  $a=0.2$ .

### Entrainment coefficient

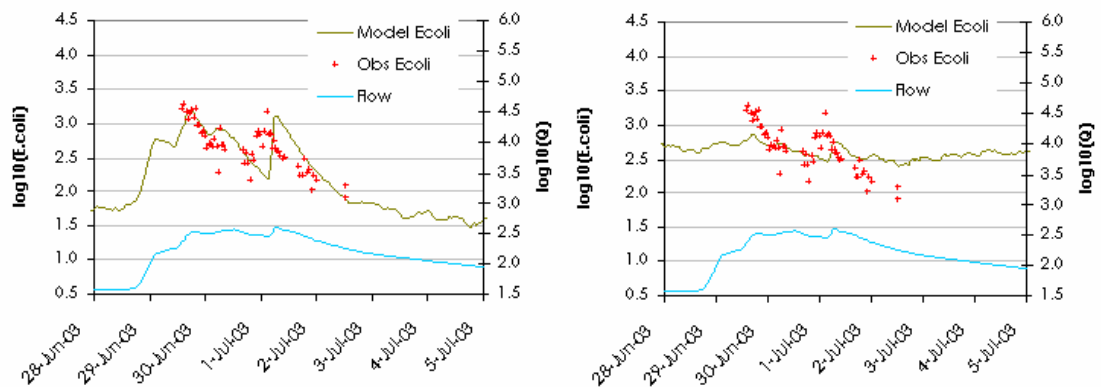
Adjusting the entrainment coefficient up or down amplifies or attenuates the response of the model to flow variation and erosion/entrainment of organisms stored on the channel bed, relative to the existing ambient/background water concentration. The model was calibrated with  $\varepsilon=0.02$ , Figure 11 demonstrates the impact of raising or lowering the  $\varepsilon$  by a factor of 10.



**Figure 9.** Comparison of varying the entrainment coefficient between  $\varepsilon=0.002$ ,  $\varepsilon=0.2$ .

### Input partition coefficient

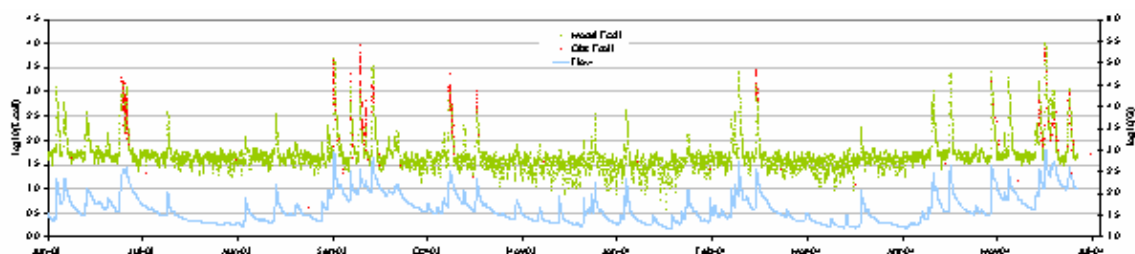
The input partition coefficient adjusts the proportion of input organisms settling or remaining in suspension. The model was calibrated with  $c_s=0.945$ . A much lower value of  $c_s=0.2$  means that there is minimal settlement to channel storage and the input bugs remain in suspension, consequently the overall *E. coli* concentration is higher and there are few bugs to be flushed from channel storage when flow events pass (Figure12).



**Figure 10.** Comparison of varying the entrainment coefficient between  $c_s=0.945$ ,  $c_s=0.2$ .

## 2.2. Modelling results

Figure 13 presents the modelling results for the full year when event response and ambient sampling data are available. As was found for the Welsh River Dee, the model performs well throughout the extended period of observation without recalibration ( $R^2 = 0.993$ ,  $\text{RMAE} = 0.143$ ). As investigated by McKergow (NIWA, pers.comm.), the variation in peak storm *E. coli* numbers can be accounted for by the variability or heterogeneity of rainfall distribution across the major Motueka catchment (Figure 14). Despite this, the mechanistic model presented here out-performs a simple load-discharge relationship when it comes to predicting *E. coli* concentrations on a peak-by-peak basis. Since the fluvial aspects of the model are driven solely by one time series of river discharge (at Woodman's Bend) it is not currently possible to account for the influence of rainfall heterogeneity.

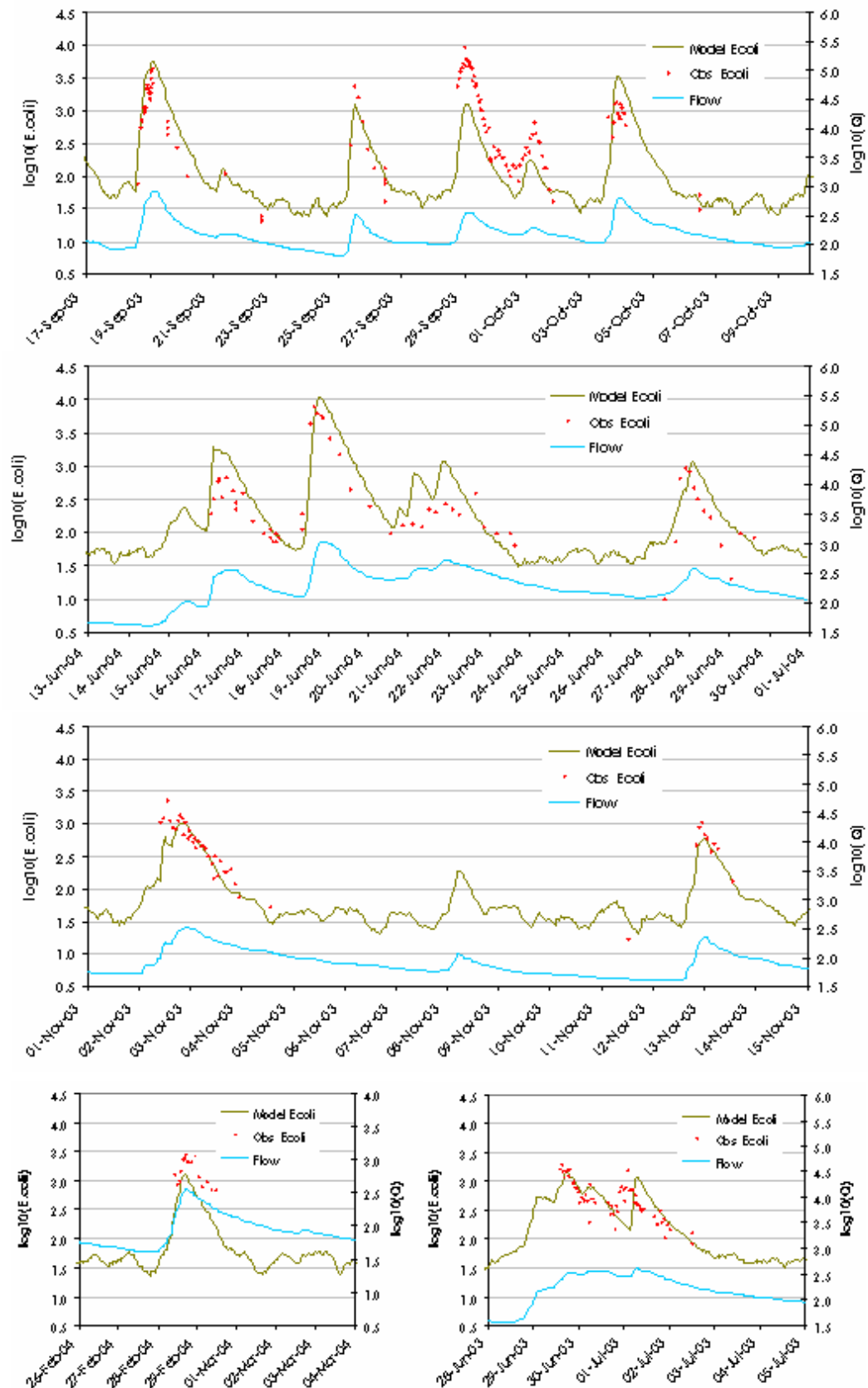


**Figure 11.** FIO flushing model results for the full year of event monitoring at Woodman's Bend on the Motueka River (observed data largely provided by Davies-Colley and McKergow), individual events are presented below.

The storm events that have been monitored by NIWA as part of the ICM programme cover a range of periods through the year; mid-winter, late winter and spring, and one in mid-summer. Figures 13 and 14 present a single model calibration for the full period of data and demonstrate that the model reproduces the observed storm-event responses to a high degree of accuracy ( $R^2 = 0.993$ ,  $\text{RMAE} = 0.143$ ) in terms of timing and the magnitude of peak concentrations (as was found by Wilkinson *et al.* 1995). It is worth restating that the model uses a single input value and that no account is taken of outdoor stocking variation on the input faecal concentration. The only seasonal variation is in die-off (and whatever variation in flow occurs). The dry-weather variability in *E. coli* concentrations is caused by die-off enhancing processes as discussed above, and summer die-off is greater than in winter. There are sparse data on ambient *E. coli* levels in the Motueka River and the input rate and partition was chosen to produce dry-weather *E. coli* variation that covers the few observed points.

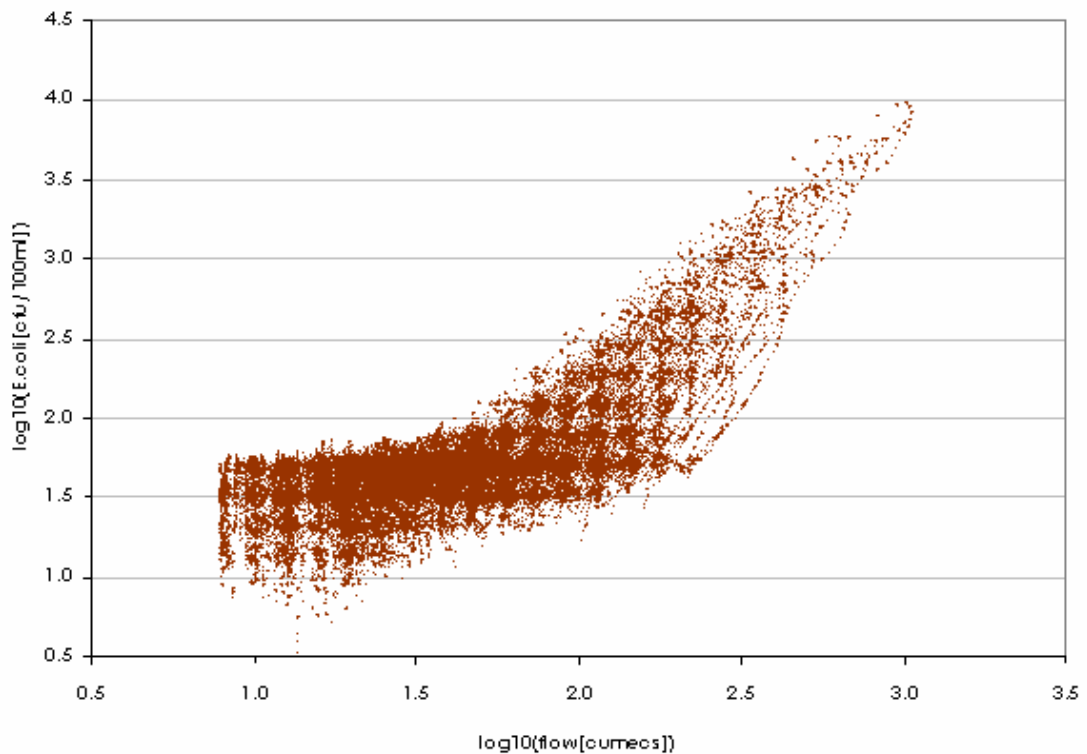
The model appears to match the observed data effectively and offers the potential for “real-time” estimation of FIO loads into Tasman Bay. The model has not yet been directly compared with a load-discharge relationship. Muirhead (Ag. Research, pers. comm.) suggests that catchment wash-off is not a significant direct contributor to storm faecal loadings. Accumulated contaminated material in the channel is considered to be the major rapidly “available” source of contaminated material. From a dynamic modelling perspective this has

certain implications. It implies that there is a disconnection between catchment sources and observed downstream microbial peak concentrations associated with storm flows. The entrained material observed in any bacterial peak will have been flushed by the passage of the storm-flow front on the rising limb of the hydrograph and this would suggest that even if rainfall is very focused in one area of the catchment, the material flushed may not all be sourced from that area.



**Figure 12.** Highlighted event modelling results from Figure 7 – a single calibration of the FIO flushing model.

Figure 15 presents the X-Y relationship between flow and modelled *E. coli* numbers. With this calibration of the model *E. coli* numbers remain below 100 cfu per 100 ml for flows up to approximately 40 cumecs and from here start to rise towards 1,000 cfu per 100 ml by around 250 cumecs. The peak model *E. coli* concentration is around 10,000 cfu per 100 ml and flows of around 1,000 cumecs.



**Figure 13.** Plot of river  $\log_{10}$  discharge versus  $\log_{10}$  (*E. coli*).

Table 2 provides a summary of statistics for paired observed and modelled *E. coli* values. The observed *E. coli* concentrations have a median value of 450 cfu per 100 ml, with a 95<sup>th</sup> percentile value of 3,695 cfu per 100 ml. The 5<sup>th</sup> percentile value which is representative of ambient dry-weather samples is 25 cfu per 100 ml. The model achieves a good fit to these statistics (Table 2), and similar results can be achieved with slight variations in the four key parameters. The random input function of the model means that the statistical errors can vary up to +/- 12%. It is possible to switch off the random function and recalibrate the model with fixed un-randomised input.

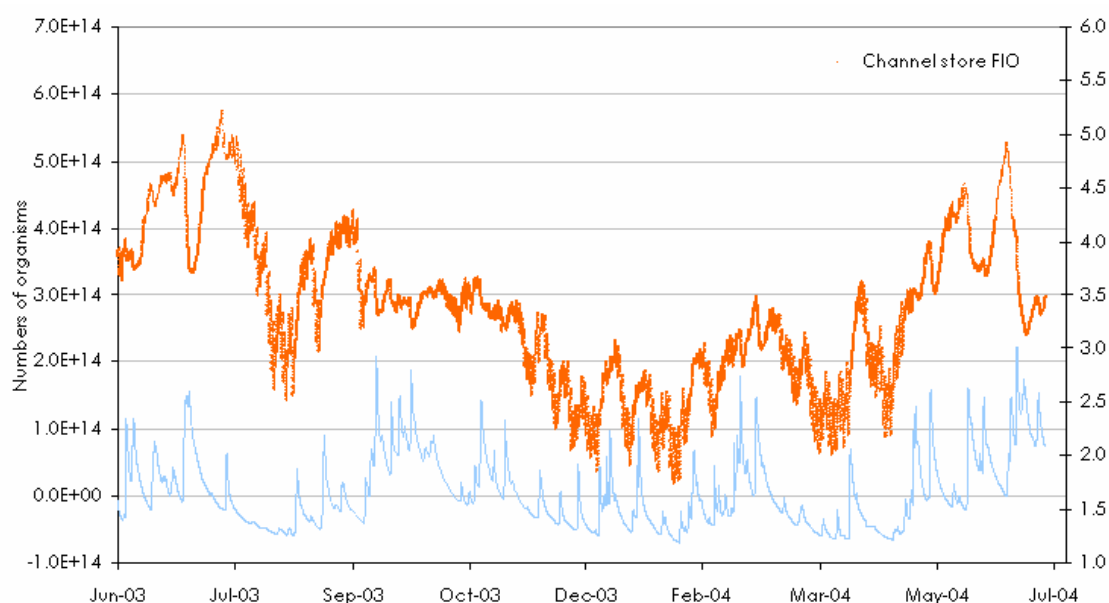


**Table 2.** Statistics for observed and modelled *E. coli* concentrations values in cfu per 100 ml (only for paired values,  $n = 271$  out of 28,880 hourly records).

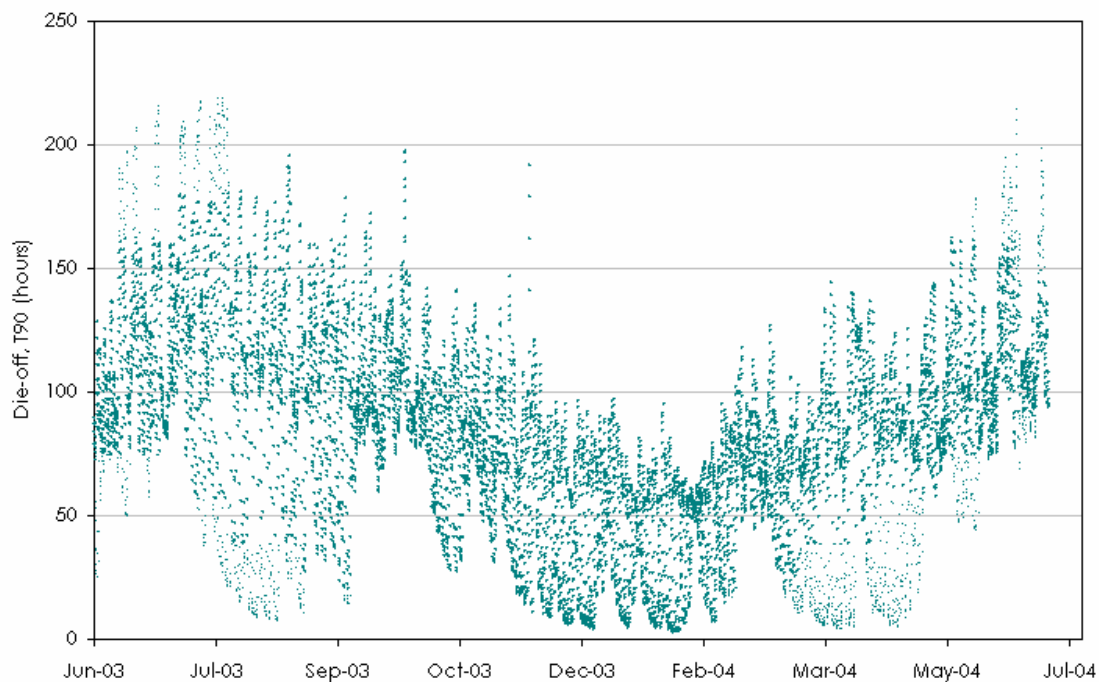
	Median	95 percentile value	5 Percentile value
Observed data	450	3695	25
Model results	446 (err. -0.84%)	3724 (err. 0.79%)	27 (err. 8.32%)

### 2.2.1. Channel store dynamics

The FIO model channel store is an internal function/variable of the model and it keeps track of the organisms that have settled out into the notional river reach available for later resuspension by a rainfall runoff flow event. Figure 16 shows a time series of the numbers of organisms in the channel store for the Motueka River application.



**Figure 14.** Plot of *E. coli* (FIO) numbers in the model “channel store”.

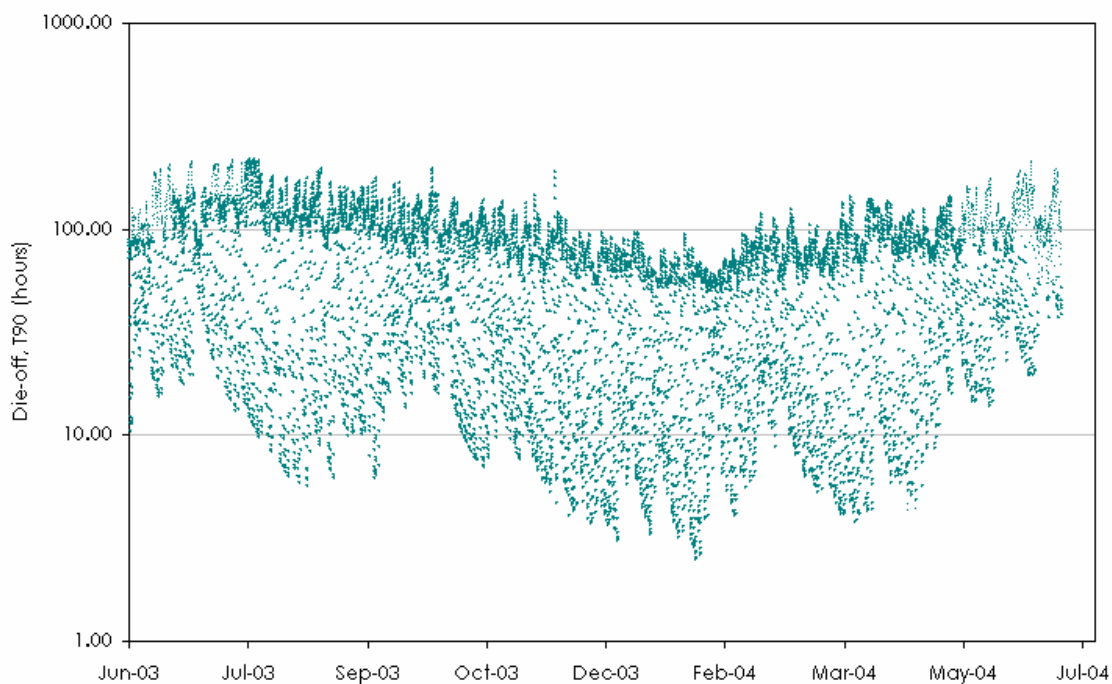


**Figure 15.** River bed die-off expressed as  $T_{90}$ .

Figure 17 shows modelled die-off in the river bed store. The annual cycle of bed store faecal numbers tends to follow the general trend of die-off. This is not surprising since the die-off annual cycle also affects the input of organisms to the river system (Figure 6). The shorter time scale variations are a function of entrainment forces. The major flow peaks can be seen to result in flushing from the channel store (sharp reductions in store numbers associated with the bigger storm peak flows). In the period of hydro-graph recession the bed-store is replenished briefly, until die-off increases and starts to reduce the input numbers and die-off at the river bed also increases.

### **2.2.2. Die-off and river faecal connectivity**

Die-off in the water column differs only slightly from riverbed die-off (Figure 18). During mid-summer dry weather the  $T_{90}$  (time for 1 log cycle reduction in numbers, 90% reduction time) values can fall to as much as a few hours during bright midday sunshine. At night the die-off is reduced to a  $T_{90}$  of around 80 hours. Table 3 summarises the water column die-off in both  $T_{90}$  and  $k_{tot}$ . The peak bright sunlight  $T_{90}$  is equivalent to a rate of 0.35 (or 35% reduction per hour). In winter die-off can still reach values of  $T_{90}$  around 15 hours, and the modelled darkness die-off can be as slow as 160 hours.



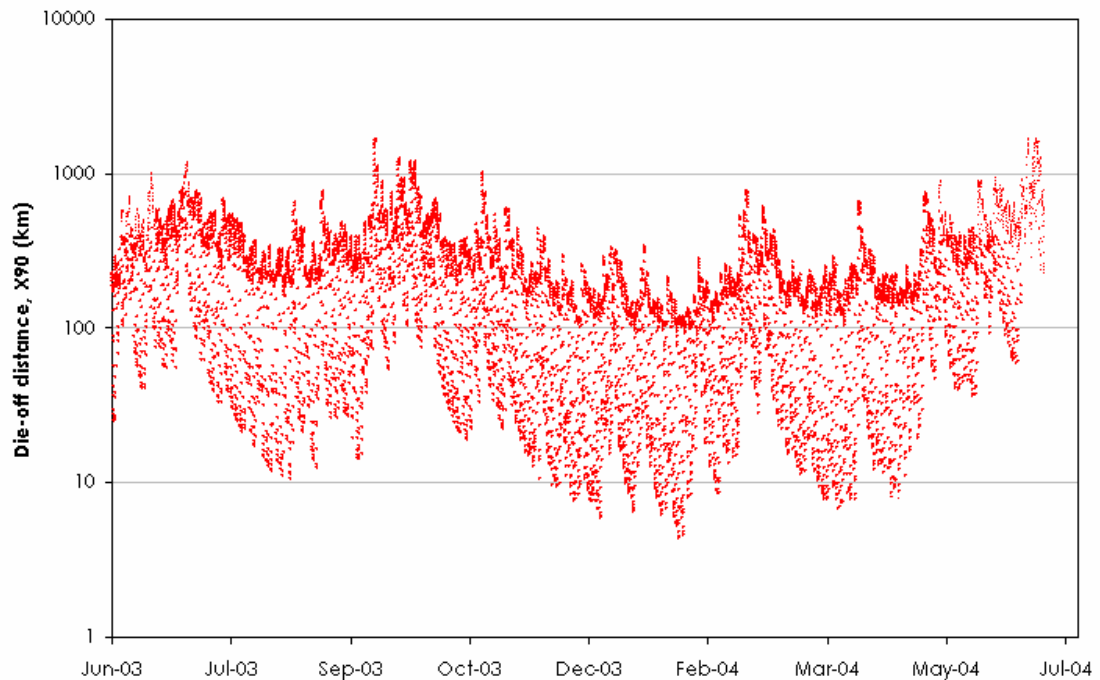
**Figure 16.** Annual cycle of hourly die-off rates expressed as  $T_{90}$  (hours).

**Table 3.** Model die-off statistics for the Motueka River Application.

Die-off measure	Max	Min	Mean	5 <sup>th</sup> %ile	95 <sup>th</sup> %ile	Median	SD
$T_{90}$ (hours)	2.8	217.8	24.3	160.7	5.9	65.5	48.9
$k_{tot}$ (per hour)	0.354	0.005	0.041	0.006	0.169	0.015	0.056

Wilkinson (2001) introduced an alternative measure of die-off that takes into account travel velocity, in an attempt to relate die-off to the faecal indicator connectivity in a river network. During dry weather when river velocity is slow, water clarity is high and settlement is enhanced, organisms entering the river channel may either settle or die-off in a location near to their point of entry. The term used to express this is  $X_{90}$  and in words is the “90% die-off travel distance” and is calculated from  $X_{90} = v / k_{tot} = T_{90} \cdot v$  where  $v$ , is the mean travel velocity.

When investigating up-catchment influences the  $X_{90}$  offers utility, in that it gives an indication of whether organisms from a given location have a high probability of reaching a site downstream under prevailing conditions (given a reasonable degree of confidence in the die-off rate and estimated velocity). Figure 19 presents estimated  $X_{90}$  for the Motueka River based on the discharge velocity relationship available from the river and  $T_{90}$  presented above. Table 4 summarises the results.



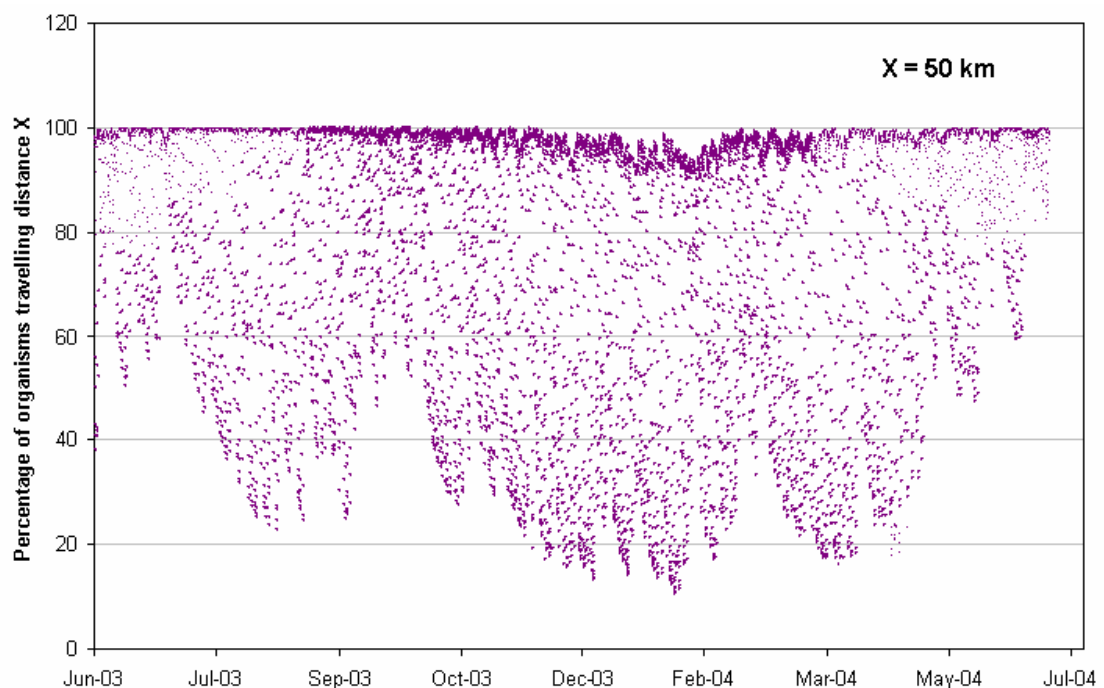
**Figure 17.** Time series of  $X_{90}$  for the Motueka River at Woodman's Bend.

**Table 4.** Summary statistics for  $X_{90}$  in the Motueka River.

Die-off measure	Max	Min	Mean	5 <sup>th</sup> %ile	95 <sup>th</sup> %ile	Median	SD
$X_{90}$ (kms)	1405.0	3.8	218.1	10.7	618.1	166.4	198.9
$T_{90}$ (hours)	217.8	2.8	66.7	5.9	160.7	65.5	48.9
$k_{tot}$ (per hour)	0.005	0.354	0.015	0.169	0.006	0.015	0.056

Figure 19 demonstrates that under darkness conditions throughout the year  $X_{90}$  is greater than 100 km, and storm flows can increase this to distances significantly greater *i.e.* there is little likelihood of significant die-off during storm events. The model suggests that summer daytime low flows are capable of reducing  $X_{90}$  to less than 10 km, but this only under the brightest conditions.

In addition it is possible to estimate the percentage of organisms that travel a chosen distance from  $(1 - 10^{-X_{90}/X}) \times 100\%$  (Wilkinson 2001). This may be useful to provide an indication of whether organisms from a given distance are likely to reach a point of interest. Figure 20 demonstrates that for the relatively fast flowing Motueka River, the majority of organisms are capable of travelling 50 km at night. Daytime and especially summer brightness may reduce the proportion travelling that distance. Over greater distances or in slower flowing rivers the proportion of organisms that make the distance will be lower.



**Figure 18.** Percentage of organisms that can travel 50 km under modelled die-off conditions in the Motueka River.

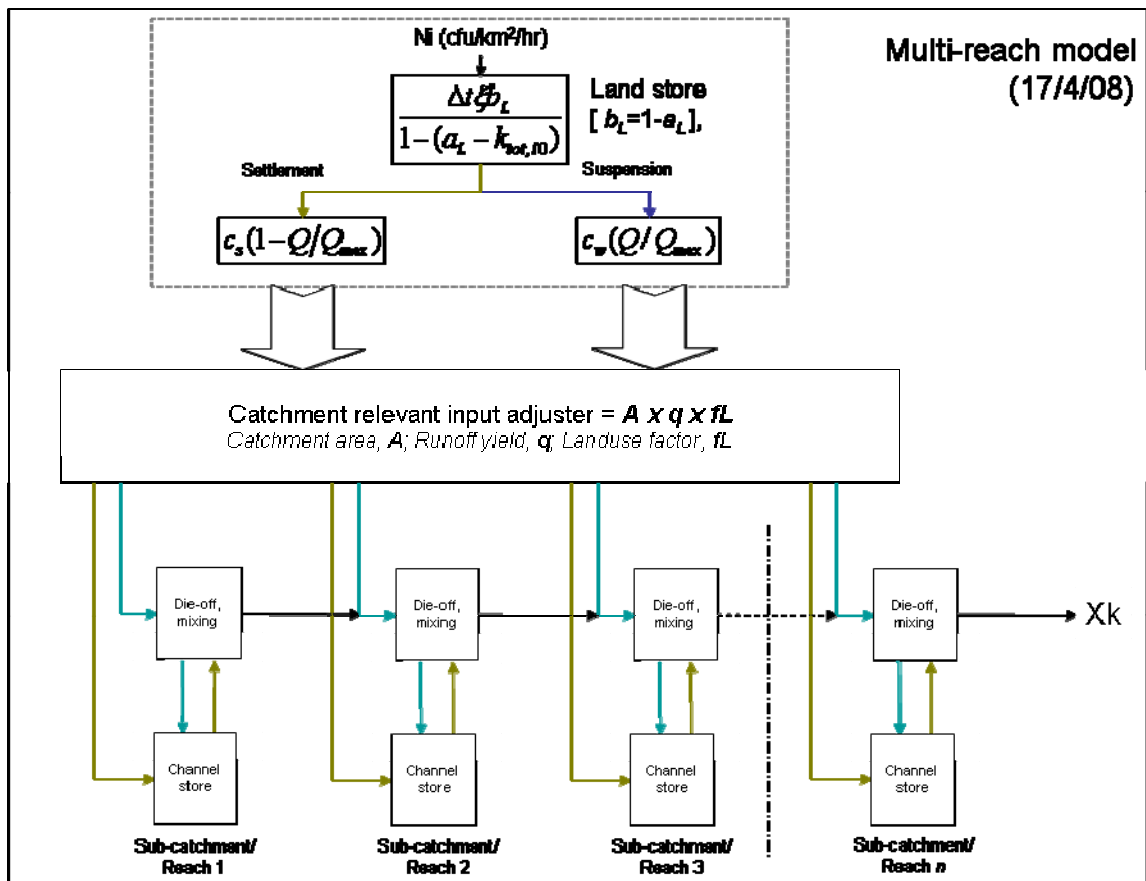
This measure can be useful to estimate, or indicate, what distance upstream of a point of interest can be considered to be having an impact downstream. On this basis it should be possible to produce a map of isohyets of maximum travel distance for a catchment for a range of seasons and fluvial/lighting conditions. This would bring the modelling a stage closer to being able to estimate the impact of upstream land-use changes on catchment outlet FIO concentrations and hence possible fisheries and bathing waters impacts. This is a proposed development and suggestion for further work.

## 2.3. Multi-reach model version

A multiple reach version of the FIO flushing model has been built for the Motueka River (Figure 21). This version is an extension of the single reach version and incorporates:

- Settlement from the water column (this allows the downstream accumulation of organisms in the channel store);
- Simple travel time delay (A fixed time delay between reaches which is based on the travel velocities at high flows when accurate description of travel velocities is most important);
- Per unit area, land-use and flow yield weighted input which is split between channel storage (river bed) and remaining in suspension;

- Catchment switches. To switch certain areas on or off;
- All working done in total *E. coli* numbers (rather than concentrations). This reduces the number of computational steps and the need to estimate reach flow volumes;
- Erosion and settlement are treated as a time varying parameter and define the relative magnitude and timing of erosion and suspension at each site.



**Figure 19.** Schematic representation of the multiple sub-catchment/reach version of the FIO flushing model applied to the Motueka River (note: the backwards shift operator  $Z^{-1}$  has been left out of the input transfer function, to simplify the diagram).

### 2.3.1. Sub-catchment input adjuster

The most useful part of the multiple reach model is the catchment input adjuster. This represents a first step towards the ability to investigate the impact of changes in land-use or stocking density on FIO concentrations downstream. Currently the land-use adjustment factors are purely qualitative, based on a visual assessment of the land-use maps for each sub-catchment. Table 5 details the land-use adjustment factors for each sub-catchment, as well as the contributing catchment area and the mean annual flow yield in litres per second per square kilometre. These combined factors combine to provide an area, flow yield and land-use weighted input to each sub-catchment in the model. The model reaches are set at 10 km each

and this provides an approximate distance between nodes. This will require refinement at a later date, but serves for an adequate initial evaluation. Travel time delays are based on high flow river velocities only, this is when channel flush and faecal load delivery to the coastal zone is most significant. Since the model is coded in Microsoft Excel it is not straight forward to “wire-in” a variable time delay, use of an alternative programming language would enable this facility.

**Table 5.** Sub-catchment characteristics and loading factors for the multiple-reach FIO flushing model.

Zone	Main Creeks	Time delay, hr	Catchment Area, $A$ km <sup>2</sup>	Runoff yield, $q$ , L/s/km <sup>2</sup>	Land-use loading factor, $fL$	Input adjustment factor	General land-use
Zone 1	Woodmans Bend	1	181.4	40.1	0.9	6547	Scrub, pastoral
Zone 2	Pearce, Graham, Orinoco, Pokororo	2	145.7	52.8	0.3	2310	Native bush, scrub, pastoral
Zone 3	Baton, Stanley, Dove	3	395.9	27.3	0.3	3239	Native bush, scrub, pastoral
Zone 4	Wangapeka (- Dart, Sherry)	4	320.0	58.4	0.2	3739	Native bush, scrub, pastoral
Zone 4	Sherry	5	78.4	24.9	1	1952	Pastoral and exotic forest
Zone 5	Dart	6	80.6	34.1	0.2	550	Native bush
Zone 4	Tadmor	5	118.5	26.7	0.8	2528	Pastoral, exotic forest
Zone 5	Upper Motueka	7	382.5	13.2	0.2	1008	Exotic and native bush
Zone 5	Motupiko	7	344.0	15.0	0.7	3612	Pastoral, exotic and native bush

In the working version of the multiple reach model the impact of changing the inputs from each sub-catchment can be investigated by varying  $fL$  the land-use loading factor. This factor has no constraint, so can be varied to any extent, so the extreme impact, for example, of a 100-fold increase in faecal input to the Motupiko catchment can be demonstrated.

### 2.3.2. Heterogeneity of rainfall distribution

An, as yet, unpublished NIWA report by McKergow and Davies-Colley investigates the influence of rainfall distribution and the variable response of different catchments through time. It will be possible to incorporate this kind of information into the model as a switch *i.e.* flushing of certain catchments can be turned off according to which events rained on each

catchment. An additional refinement of the model will be to make full use of the existing flow time series for each sub-catchment, this has been beyond the scope of the current time scale for reporting.

### 2.3.3. Model parameterisation

Table 6 represents a set of basic parameters that fit the model with  $R^2=0.996$  (RMAE = 0.129). By reducing the channel store die-off it is possible to reduce the input and redirect fewer input bugs to channel storage. This may be appropriate if it is assumed that a greater proportion of the channel store bugs accumulate in deeper pools. Alternatively, the interstices of cobbles in riffles may harbour significant numbers of bugs. A fuller review of the literature will help to improve confidence in these aspects of the model. It is worth noting that from a perspective of viewing the fluvial bacterial response at a point of interest, mixing and dispersion processes, integrate out the small scale variations that might be produced by the many and varied in-channel sources.

**Table 6.** Model parameter values for calibration of *E. coli* flushing events in the Motueka River between March 2001 and July 2004.

<i>N<sub>i</sub></i> , input numbers (organisms per km <sup>2</sup> per second)	<i>c<sub>s</sub></i> , input settlement coefficient (0 to 1)	<i>v<sub>s</sub></i> , settlement velocity (m/d)	<i>ε</i> , erosion coefficient	<i>a</i> , dispersion coefficient (0 to 1)
10,000,000	0.99	5.0	0.5	0.77

It is therefore possible to fit the model with a range of parameter values, however, as a modeller and water quality scientist, it is prudent to choose combinations of parameters that best reflect our conceptual understanding of the system operation, and adjust these in the light of new information.

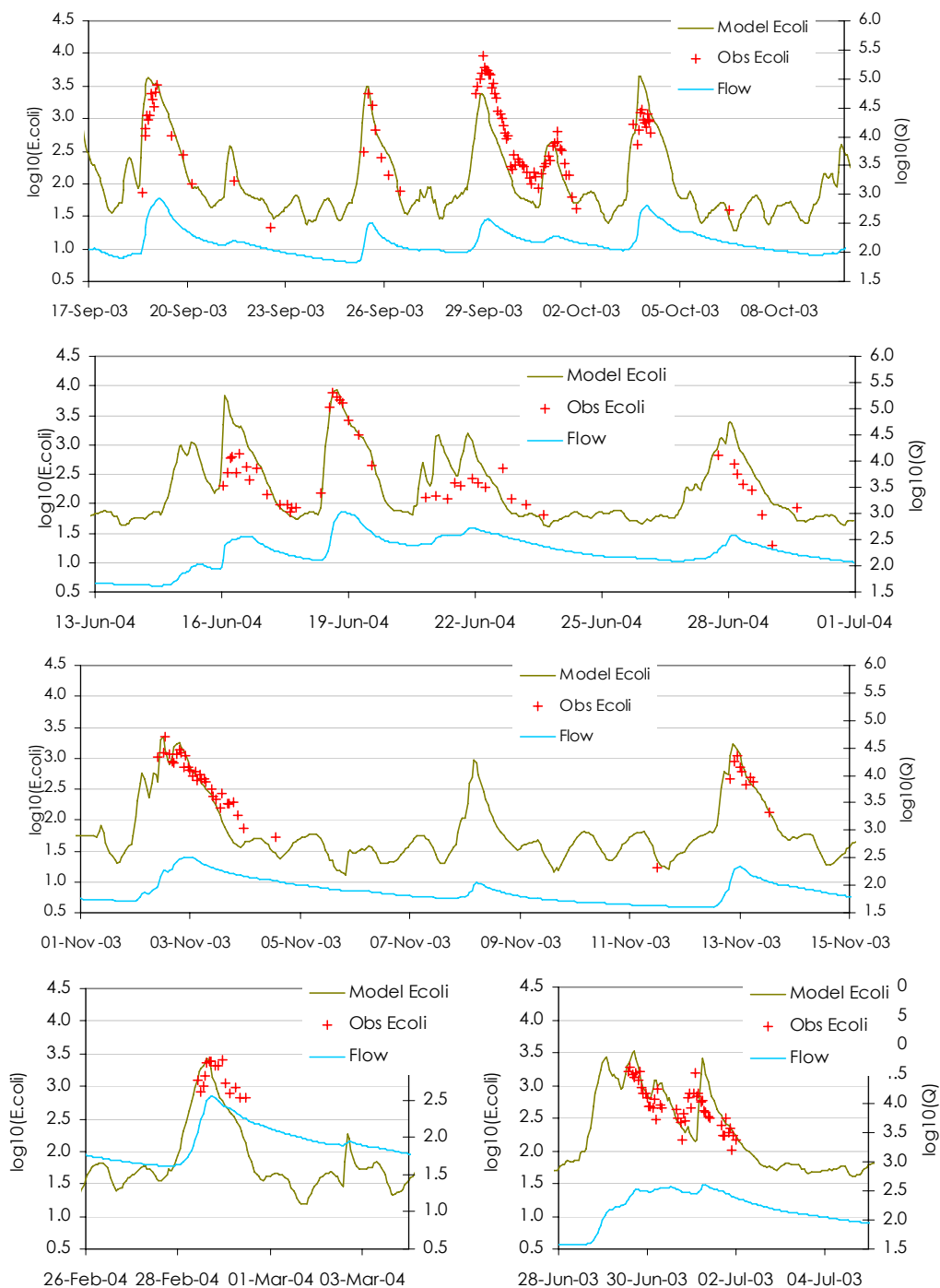
Figure 20 provides an example of the multiple reach model fit without additional refinement of the sub-catchment land-use loading. The model fit to the observed data is certainly an improvement on the single box model, however the differences in fit of each event response can be attributed to rainfall distribution (McKergow, NIWA, pers.comm.).

Table 7 summarises basic statistics for the FIO flushing model paired results. The model statistics fit the observed data relatively closely. Additional statistics can be taken from the full time series to assess the models predicted high- and low-flow *E. coli* concentrations. This might be of use for assessing bathing waters issues.



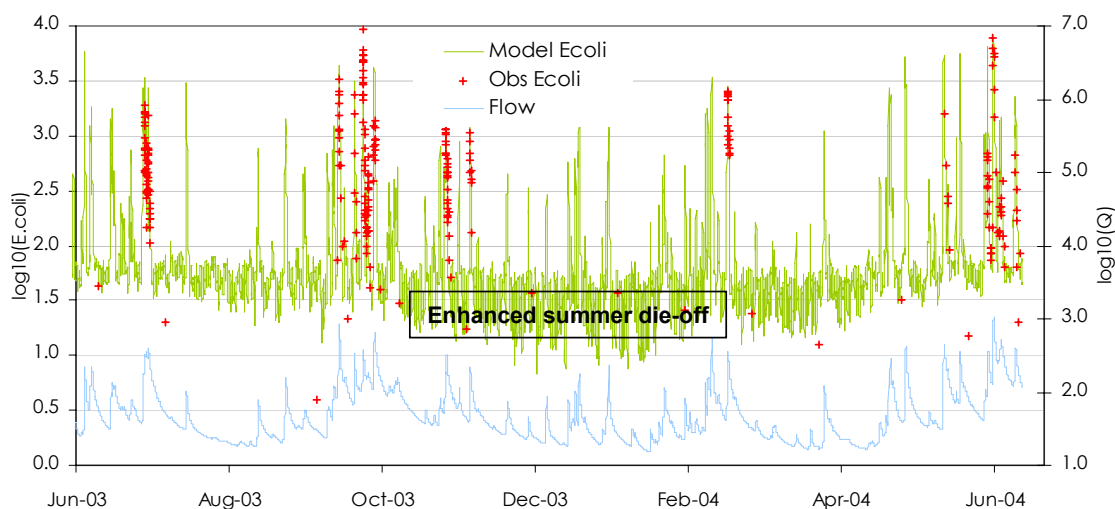
**Table 7.** Comparison of observed and model *E. coli* values at Woodman's Bend on the Motueka River.

<i>E. coli</i> (cfu/100ml)	Median	95 <sup>th</sup> percentile	5 <sup>th</sup> percentile
Observed data	456	4117	37
Paired model data	482	3526	42



**Figure 20.** Event by event modelling result for the multiple reach version of the FIO flushing model.

Figure 21 demonstrates the influence of greater summer die-off on the modelled *E. coli* concentrations. The model variation encompasses the few low flow samples that were available, however, these data are clearly too sparse to adequately assess the validity of the low flow predictions of the model. Further low flow validation sampling is required over a number of diurnal cycles during winter and summer and with varying light conditions.



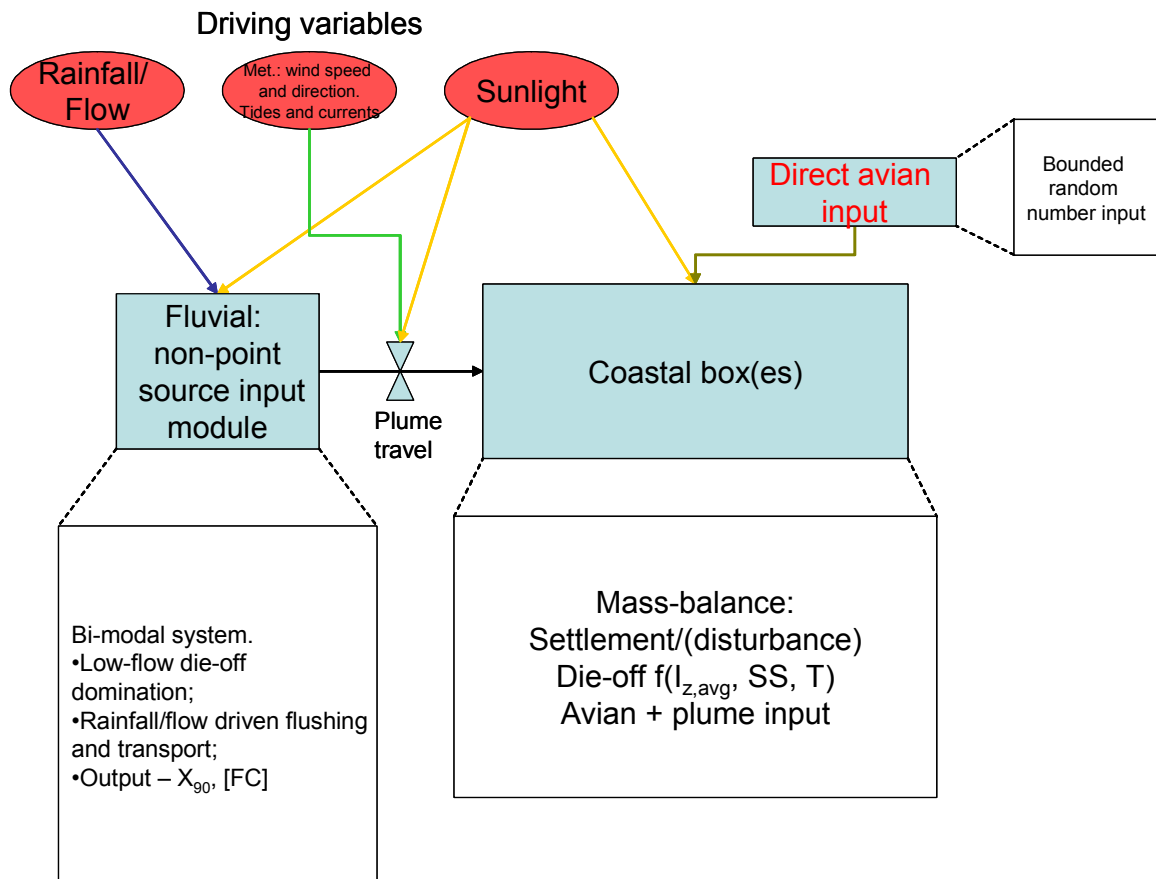
**Figure 21.** The multi-reach model full 2003-04 time series demonstrating the influence of die-off over the summer period.

## 2.4. Plume tracking off the Motueka River

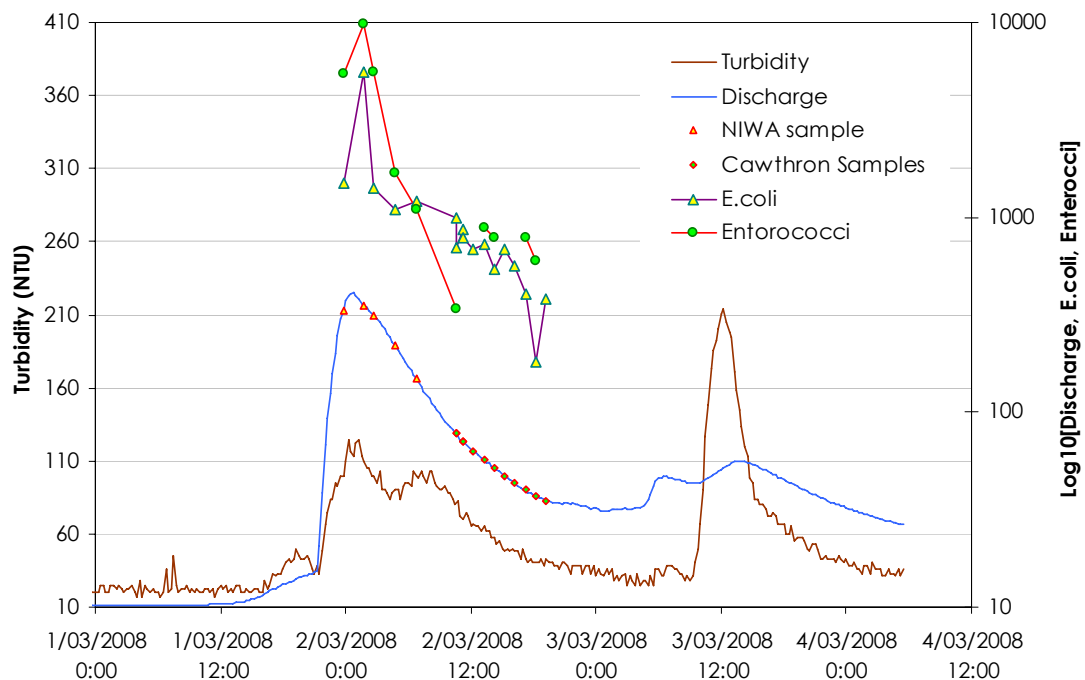
The work reported here contributes to an investigation of FIO plume movement and dispersal out into Tasman Bay. The Tasman Bay plume modelling is reported elsewhere. The aim of the study was to link the FIO die-off model to a transport model for Tasman Bay, with input from the Motueka River FIO model (Figure 24). In the Tasman Bay model, indicator numbers are assigned to particles in a particle tracking model which moves the organisms out into the bay according to prevailing transport conditions. Die-off is applied to reduce the bug numbers on each particle. From the particle tracking results an equivalent grid-square FIO concentration is estimated to indicate the concentration of plume water as it passes through shellfish harvesting waters.

In order to ground-truth the model, a series of plume tracking exercises were planned. The results of the first event monitoring are presented (Figure 25). The event of 2 to 4 March proved to be insufficient to cause a significantly contaminated plume, and monitoring of transects in Tasman Bay returned very low FIO counts. The river sampling was also hampered by problems with auto-samplers. The use of auto-samplers for bacteriological sampling is often frowned upon because of the potential for sample cross-contamination from organisms remaining in tubing. In this case, double flushing helps to reduce residual contamination

between samples, and perhaps more importantly in a river the size of the Motueka River the storm associated temporal dynamics are long-scale, measured in hours. This is unlike small streams where the mixing and transport dynamics are short time scale relative to the sampling frequency. In other words the river is so large and well mixed that short-term variations are minimal *e.g.* Figure 14.



**Figure 22.** Schematic representation (conceptual model) of river load and plume tracking model.



**Figure 23.** *E. coli*, Enterococci, turbidity and discharge in the Motueka River at Woodman's Bend on 2 March 2008.

As part of the plume tracking exercise it was necessary monitor the discharge from the Motueka River. Figure 25 presents the monitoring results. Samples were collected from the Cawthron and NIWA auto-samplers, and a few grab samples were also collected. The Cawthron auto-sampler was triggered late on 1 March 2008 prior to the main hydrograph rise. Unfortunately the sampler jammed and did not collect any samples. NIWA samples were available from later on 1 March at a lesser sampling frequency. The Cawthron sampler was re-triggered around midday on 2 March and collected a further ten samples until it ceased operating due to battery depletion.

Samples were enumerated for *E. coli* and enterococci and additional continuous turbidity data from the NIWA logger was available. As shown in Figure 25, enterococci numbers were greater than *E. coli* numbers at the peak of the main flow event. The highest enterococci level was around 10,000 cfu per 100 ml, whereas *E. coli* was 560 cfu per 100 ml. No hydrograph rise samples were collected. During the recession enterococci numbers appear to have declined more rapidly than *E. coli* numbers. The variation in FIOs was similar to the changes in turbidity. The second storm flow of 3 March was much smaller than the 2 March flow, but resulted in a far greater turbidity spike. This may have been due in part to resuspension of settled material and catchment runoff. No FIO samples were taken during this second event.

The river event appears not to have been of sufficient magnitude in volume or concentration to have a significant impact on the bay. Sampling of the storm plume across a number of transects and at various depths recovered few samples with FIO numbers above the limit of

detection. However, shellfish can filter large volumes of water -- oysters for example can pump 1,500 times their own body volume in one hour (see Lee *et al.* 2002). So, despite the lack of a measurable FIO plume it is possible that shellfish still ingest indicator organisms and pathogens and, as certain pathogenic organisms migrate beyond the stomach sac into body tissues, any tentative conclusion that no measurable FIO increase in the bay is a good result should be treated with caution.

This leads to further work. If an objective of this work is to enhance the protection of shellfish harvesting areas, then funding for further storm event sampling is desirable.

## 2.5. Conclusions and further work

This report introduces faecal indicator modelling work in the Motueka River. The model is described and its application and calibration is explained. The model is capable of reproducing observed *E. coli* event responses over a one year period using the same calibration coefficients throughout the year.

The dry-weather dynamics of the model are determined by die-off forcing variables; sunlight and secondary variables. During wet weather, die-off forcing is minimal and fluvial mechanisms determine the model response. The least well-defined “parameter” is the input number of bugs, but this can be treated as a single number that is constant over time. The input bugs are applied to a catchment store and then routed into channel storage and suspension in the water column according to river flow.

The flushing of bugs in the river channel during storm flow events is well known as a cause of elevated FIO concentrations (*e.g.* Wilkinson 2001). This appears to be the case for the Motueka River. Settlement into channel storage is modelled as a flow-dependent process, with the majority of organisms settling into channel storage during dry weather and input organisms predominantly staying in suspension during high river flows.

Die-off emerges as a variable which has a number of uses. It can be used;

- To indicate how long organisms can be expected to survive under a range of environmental conditions;
- When combined with mean travel velocity, to indicate how far the majority of organisms might be expected to travel before major die-off has occurred; and
- For a given distance, the percentage of upstream organisms reaching a certain point downstream may be estimated.

In slower flowing rivers, die-off and travel time can be considered in the context of connectivity. If the velocity is sufficiently slow and the die-off suitably high, organisms from one point may die-off before they have a chance to be transported a significant distance downstream. In such a case it might be reasonable to argue that two locations are microbially disconnected, and that FIO numbers at the downstream point are more likely to emanate from

a nearby site. As river flow increases during a storm and die-off mechanisms are minimised the extent of “active catchment” contributing to a given point of interest must expand.

The multi-reach version of the model allows connectivity to be investigated, and although low flow bacterial inputs have minimal impact downstream, when the flow rises, inputs from high up the river network can have a major influence downstream.

The modelling of faecal contamination does not conform to simple behaviours. The simplest case in water quality modelling is a conservative solute. It is conservative meaning that it does not undergo any transformations, other than dispersion as it mixes downstream. Faecal contaminants suffer die-off, attach to particles and settle and can be resuspended. Their river transport is as complicated as any other biological vector.

The models presented are simple “at a point” and “multiple reach” simulation tools and achieved a very good fit to observed data.

Further developments of this work could include:

1. Refinements of the model:
  - Sensitivity/redundancy analysis;
  - Build-in faecal indicator results for other Motueka River locations;
  - Refine land-use and associated FIO loadings (review United Kingdom loading studies);
  - Optimise reach lengths for each site;
  - Build model in a more versatile platform (to allow variation of time delays with flow/velocity);
  - Undertake additional low flow monitoring over diurnal cycles to confirm die-off influence under clear skies, and to improve low flow validation;
  - Incorporate reach by reach existing river flows to correct individual reach event responses;
  - Examine river channel-bed microbial concentrations immediately after a storm in order to assess rapid post-storm deposition and decline thereafter;
  - A summary of die-off values typical on a seasonal basis and relative to wet/dry weather conditions.
2. Other applications of the model:
  - Apply to other sites;
  - Apply to other contaminants;
  - Collaborate and compare with AgResearch (Richard Muirhead), NIWA (Graham McBride, Rob Davies-Colley, Lucy McKergow).

The FIO flushing model presented in this report is a useful tool for simulating *E. coli* peaks in the Motueka River. It offers potential application to many sites and other variables, and for feeding into a coastal model for faecal contamination. There is the potential to investigate bathing water quality, and the multiple reach version of the model will be of use to examine the impact of land-use changes and issues regarding stocking intensities.

### 3. ACKNOWLEDGEMENTS

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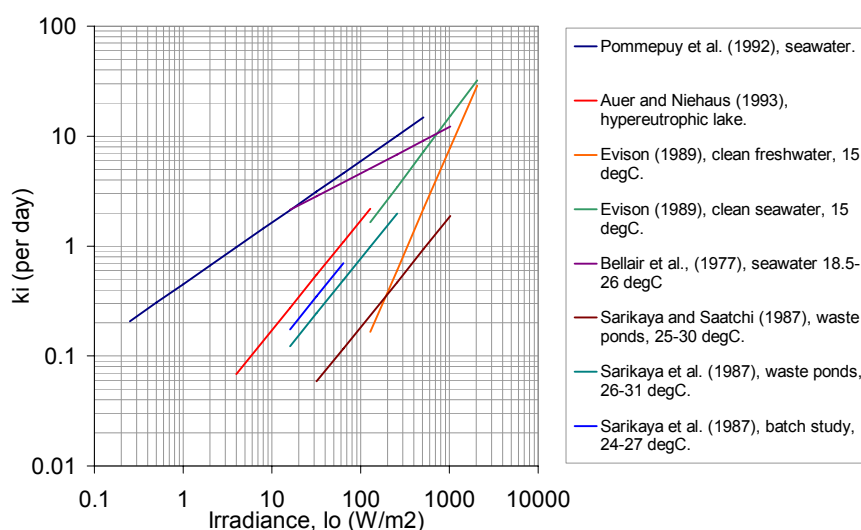


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## 5. APPENDICES

### Appendix 1. FIO model die-off component (from Wilkinson 2001)

Three sets of equations are presented below relating faecal coliform (FC) die-off to solar irradiance, water temperature and pH. The equations relating FC die-off to irradiance and suspended solids, are taken from the literature (*e.g.* Pommepuy *et al.* 1992; Auer & Niehaus 1993). The equations for temperature and pH are derived from common observations in the literature. These are the only environmental variables used in the model for the purposes of calculating die-off rate. Microbial interactions in the self-purification of natural waters are of great importance (*e.g.* Verstraete & Voets 1972; Enzinger & Cooper 1973; Roper & Marshall 1978; Flint 1987; Singleton & Sainsbury 1981; Brettar & Höfle 1992), but these interactions are many and complicated and it would be difficult to measure their effects readily for the purposes of driving a water quality model. A model incorporating the microbial functions would have many additional parameters that would be difficult to assign a meaningful value. The model would have more parameters than variables and would be considered over-parameterised. In the current model the effects of microbial interactions on FC die-off are considered to be an internal process of the aquatic system. Their effects are assumed to be implicit within the functions driven by the readily measured, or derived, physical and chemical variables *i.e.* irradiance and suspended particles, temperature and pH. This assumption is supported by the use of parameter assignments associated with observations of FC die-off behaviour in natural river or lake-water in the presence of the natural micro-flora and fauna. Thus the functions incorporate not only the direct influence of the driving variable on the coliform organisms but also the hidden effect on the natural microbes and the changes, or otherwise, in their influence on the coliforms. The different equations are presented below.



**Figure 1.** Modelled relationships between solar radiation and faecal coliform die-off (after Wilkinson 2001).

Faecal coliform die-off is related to the strength of sunlight (solar irradiance) by the power law

$$k_i = \alpha I^\beta$$

where,  $\alpha$  = die-off,  $k_i$ , at  $I=1$  and  $\beta$  = the slope of  $\log_{10}(k_i)$  against  $\log_{10}(I)$ . Figure 1 presents power law models fit to the results of five studies examining the relationship between light intensity and FC die-off in fresh and salt-waters and in waste treatment lagoons, under field and laboratory conditions. Light intensity values have been converted where necessary to  $\text{W.m}^{-2}$  and die-off rates are expressed as per day. The data of Auer & Niehaus (1993), Evison (1989) and Sarikaya *et al.* (1987) produce a value of  $\beta$  of around 1, and similar values for  $\alpha$  and have been used in later calculations of die-off.

For sunlight penetration in water irradiance is usually denoted  $I_{z,avg}$ , the depth averaged irradiance (see Sarikaya *et al.* 1987; Qin *et al.* 1991; Auer & Niehaus 1993);

$$I_{z,avg} = \frac{I_0}{\eta z} \{1 - e^{-\eta z}\}$$

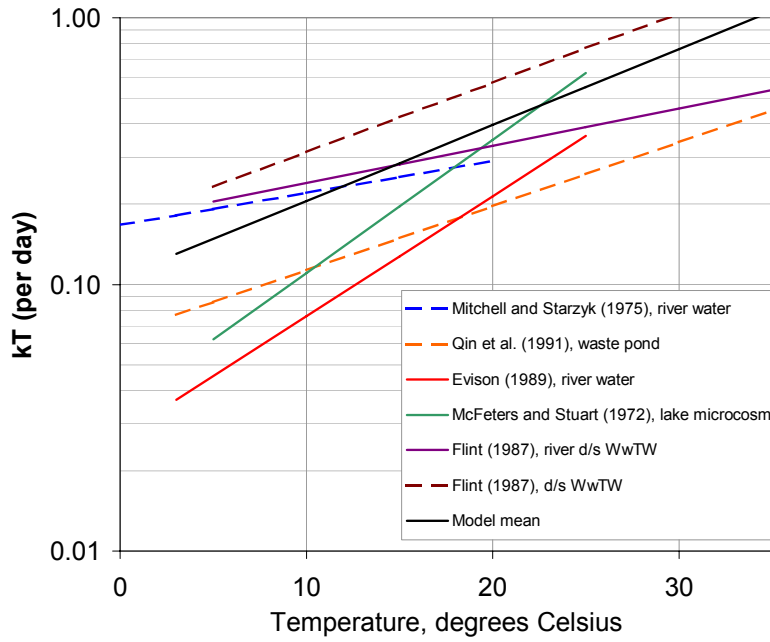
where,  $I_0$  is the irradiance at the water surface,  $\eta$  is the light attenuation coefficient and  $z$  is average water depth (Sarikaya *et al.* 1987; Qin *et al.* 1991; Auer & Niehaus 1993). Pommeupuy *et al.* (1992) found 73% of observed light attenuation in fresh river water to be due to suspended and dissolved organic matter, such that

$$\eta = 0.22 C_{sm}^{0.78}$$

where  $C_{sm}$  was the concentration of suspended matter in  $\text{mg.l}^{-1}$ .

Gameson & Gould (1975) reported that the effect of sunlight on coliform die-off was additive and independent to that of temperature such that

$$k = k_T + \alpha I_{z,avg}^\beta$$



**Figure 2.** Modelled relationships between water temperature and faecal coliform die-off (after Wilkinson 2001).

Die-off in natural waters increases exponentially with temperature such that at higher temperatures, a small change in temperature will cause a greater change in die-off rate than at lower temperatures. This relationship can be represented by the simple equation

$$k_{T_2} = k_{T_1} 10^{\theta \Delta T}$$

where  $\Delta T$  is the change in temperature ( $^{\circ}\text{C}$ ) and,  $k_{T_1}$  and  $k_{T_2}$ , are the previous and current die-off rate, respectively.  $\theta$  is the slope of  $\log_{10} k_T$  against temperature;

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

An alternative formulation for  $k_T$  is

$$k_T = 10^{K_0 + \theta T}$$

This gives the die-off for any temperature, relative to the reference die-off at  $T = 0^{\circ}\text{C}$ ,  $k_0$ , and

$$K_0 = \log_{10} k_0$$

The mean values of  $\theta$  and  $k_0$  are 0.0285 and 0.1067, respectively. These have been used below to illustrate the effect of the various die-off functions with environmental data.

For the third physical variable, it has been possible to derive a relationship with pH. Observations of the effect of different pH conditions on FC die-off have shown that die-off is increased at non-neutral pH *i.e.* die-off is accelerated, as conditions become increasingly acid or alkaline. This approximates a hyperbolic cosine function,

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

where,  $k_{pH}$  is die-off due to pH,  $pH_{kmin}$  is pH at minimum die-off ( $k_{min}$ ), and  $a$  is a constant of proportionality. Fitting this equation to the data of McFeters & Stuart (1972),

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

The parameters used to fit the data of McFeters & Stuart (1972) have been used in the illustrations below. The application of the equation is such that the value  $k_{min}$  is replaced by the prevailing die-off value at pH 6.5.

The three die-off components are combined to give a value of total, or overall, die-off per day,

$$k_{tot} = (k_d 10^{\theta(T_2 - T_1)} + \alpha I_{z,avg}^\beta) \cosh(a(pH_{min} - pH)). \quad \text{Equation 1}$$

The die-off effects are treated in the following progression

$$k_{tot} = (k_d f(T) + f(I_0, SS)) f(pH)$$

*i.e.* in words, the effect of changing temperature on the darkness die-off rate is calculated, to this are added the effect of irradiance and particulates ( $SS$ ). The result is multiplied by the pH effect. Since pH is an internal quality of the water or aquatic system this seems reasonable. For example at higher temperatures or irradiance rates, when the contaminant microbial population is already heavily stressed, if the pH deviates from around neutral, the increase in die-off would be far more dramatic than during dull or cool conditions. If pH were treated as an additive effect then the increase in die-off would be the same under high or low stress conditions.

The die-off component of the model has a maximum requirement of five variables and six parameters. The assignment of parameters is considerably simplified by the use of rate constants derived from literature data. Only one parameter has to be assigned, this is the slope of the temperature response,  $\theta$ , which can be chosen according to the degree of sewage contamination of the water being studied. The other parameters are assumed to be valid for a wide variety of applications, while accepting that further investigations of die-off are necessary.

Water depths vary significantly in the direction of flow and across a channel with the variety of channel forms in any river reach (see Bathurst 1993). The depth value in the model is used as a general indication and the effects of spatial variations in depth on die-off rate are assumed to be averaged out by mixing within the channel.

The die-off model is initialised with the assignment of a value of darkness die-off rate. This may be chosen from Figure 2. Once this initial value has been chosen the model is run and die-off is automatically adjusted by the driving variables. Finally, in cases where pH does not vary significantly the effect might be ignored. In other cases such as in rivers affected by acid mine drainage or in upland streams affected by acidic episodes and generally low pH (*e.g.* Neal *et al.* 1999) the incorporation of the pH function would be appropriate.

### **Settlement: the apparent die-off**

Faecal coliform settlement is of importance to the examination of die-off in the River Dee with the ponded lower reaches from Farndon to Chester weir. During summer low flows settlement may be an important contributor to the overall loss of organisms from the water column.

Reynolds (1979) proposed a model for the deposition of particles from turbulent flow based simply upon mean water depth,  $z$ , and stokian (still-water) settling velocity,  $v_s$ , of the particles. The settlement rate,  $k_s = v_s / z$ . This represents a gross settlement within the water column and does not account for the actual deposition of particles. The net rate of deposition will decrease with increasing flow because increased turbulence will maintain particles and associated organisms in suspension to a greater extent than at lower flow (e.g. Graham 1990). In the mass-balance model presented by Wilkinson (2001) the reduction in deposition is accounted for automatically within the model. For the purpose of the illustration presented below it was necessary to use a simple function  $k_s = v_s / z (1 - Q/Q_{95\%ile})$  to reduce loss by settlement with increasing discharge. The settlement velocity used in the estimates presented below was  $1.2 \text{ m.d}^{-1}$  after Auer & Niehaus (1993) and is consistent with the findings of Milne *et al.* (1986) studying coliform settlement with estuarine mud and silts.

## Appendix 2. FIO entrainment model (from Wilkinson 2001)

Entrainment mechanisms result from the disturbance of organisms, attached to or stored within the wetted perimeter of a stream or river channel, by the passage of a propagating flow-wave and expansion of the field of the main turbulent flow body. This section gives some background to the steps taken in model development, describes a new model devised and developed with the event data to reproduce the observed behaviour, and presents the modelling results.

The early development of the model for faecal coliform fluvial dynamics followed attempts to apply the model of Jenkins (1984), as well as simple threshold entrainment models. The model of Jenkins (1984) used equations for entrainment and settlement from a single channel store of organisms with non-linear functions and threshold values to set "knee-points" in response curves to limit the rate of entrainment. Since the Jenkins model could not reproduce the behaviour observed in the field experiments and threshold approaches were over-simplistic an alternative model was devised.

The basic form of the new model follows the mass-balance plan used by Jenkins (1984). The river reach is treated as a two-box system; an upper box for the changes in concentration in the water,

$$X = I + E - S - D$$

and a lower box for the organisms stored in the associated part of the channel

$$N = S - E - D$$

where,  $X$  is the mass balance of inputs,  $I$ , entrainment,  $E$ , settlement,  $S$ , and die-off,  $D$ . The number of organisms in channel storage,  $N$ , is the sum of settlement and losses due to entrainment and die-off.

The key process of the model for this application is entrainment. The model has two components for entrainment, a slow entrainment mechanism associated with the expansion of the turbulent body of the flow with associated processes of bursting and of cobble disturbance. The second part of the entrainment system relates to rapid entrainment associated with wavefront processes.

An essential part of the proposed model is the integration of entrained organisms with the passage of the flow events. Active-mixing-volume theory was used as the basis for this. Investigation of active mixing volume modelling in rivers has shown that only a proportion of the water volume in the river reach is effectively involved in mixing and dispersing a tracer or contaminant pulse (e.g. Wallis *et al.* 1989); hence the term active mixing volume. The proportion of the river reach volume,  $V$ , termed the "effective volume",  $V_e$ , is said to be the "dispersive fraction" ( $Df$ ) of  $V$ , and has been shown by experimentation at different discharge rates in a number of United Kingdom streams and rivers to be weakly dependent on discharge, decreasing slightly at high flows (e.g. Wallis *et al.* 1989)

$$\text{As } Df = \frac{Ve}{V} \text{ and } b_0 = \Delta t \frac{Q_k}{V_k}, \frac{b_0}{Df} = \Delta t \frac{Q_k}{Ve_k},$$

and since the variation in  $Df$  is small with respect to discharge, it has been assumed to be constant for the purposes of this application. The mixing volume residence time,  $T = Ve/Q$ , gives a measure of the time a conservative tracer would reside in the study reach (Price *et al.* 2000). On the assumption that hydrograph propagation is analogous to the dispersion of a conservative pollutant (Whitehead & Hornberger 1984) the routing parameter,  $b_0$ , was used in the water quality model to determine mixing characteristics through the artificial hydrograph events. In the one dimensional approach used here, an important simplifying assumption made is that the water is completely mixed across the channel cross-section (see Henderson-Sellers *et al.* 1988).

Price *et al.* also present the full AMV equation for time varying  $Q$  and  $V$  written in the mass-balance arrangement, which for a conservative solute is

$$\frac{dM_o}{dt} = \frac{Q_t}{Ve_t} M_{i-t} - \left\{ \frac{Q_t}{Ve_t} + \frac{1}{Q_t} \frac{dQ_t}{dt} + \frac{1}{Ve_t} \frac{dVe_t}{dt} \right\} M_o.$$

where  $M_i$  is the mass entering the reach and  $M_o$  is the mass leaving the reach outlet. Since

$$\frac{dM_o}{dt} = \frac{dVe_t X_t}{dt} = Ve_t \frac{dX_t}{dt} + X_t \frac{dVe_t}{dt}$$

substituting and rearranging for the change in concentration of solute with time gives

$$\frac{dX_t}{dt} = \frac{Q_t}{Ve_t} U_{t-\tau} - \left\{ \frac{Q_t}{Ve_t} + \frac{1}{Q_t} \frac{dQ_t}{dt} + \frac{2}{Ve_t} \frac{dVe_t}{dt} \right\} X_t.$$

This is the basic AMV for a conservative solute. For modelling faecal coliforms the additional terms for die-off, settlement, rapid and slow entrainment, and lateral inflows have to be added to the mass-balance, which becomes;

$$\frac{dX_t}{dt} = \frac{Q_t}{Ve_t} U_{t-\tau} - \left( \frac{Q_t}{Ve_t} + \frac{1}{Q_t} \frac{dQ_t}{dt} + \frac{2}{Ve_t} \frac{dVe_t}{dt} + k_s + ktot_t \right) X_t + \frac{Ne_t + Ni}{V_t}$$

where  $V$  is the volume of water in the reach and  $k_s$  is the settling rate,  $k_s = v_s / z_t$ , where  $v_s$  is the stokian settling velocity and  $z_t$  is the water depth,  $ktot_t$  is the die-off rate. The inputs are  $Ne_t$ , which is the sum of the entrainment inputs, and  $Ni$ , catchment derived inputs.

Taking the finite difference backwards approximation of the one-dimensional differential equation and rearranging for  $X_k$  gives

$$X_k = \Delta t \frac{Q_k}{Ve_k} U_{k-\delta-1} + \left\{ 1 - \left( \Delta t \left( \frac{Q_k}{Ve_k} + k_s + k_{tot} \right) + \frac{2(Ve_k - Ve_{k-1})}{Ve_k} + \frac{Q_k - Q_{k-1}}{Q_k} \right) \right\} X_{k-1} + \dots$$

$$\dots \frac{\Delta t (Ne_k + Ni)}{V_k}.$$

In this fully dynamic formulation the possibility of instability becomes an issue. In situations where there is a smooth transition from low to high flow and *vice versa* it is easy to choose a temporal discretisation interval,  $\Delta t$ , that keeps the changes in  $Q$  and  $Ve$  small with respect to



the absolute values of  $Q$  and  $Ve$  from one time step to the next. For greater  $\Delta t$ , where there changes in  $Q$  and  $Ve$  may be large with respect to  $Q$  and  $Ve$ , there is the potential that the model will be unstable *i.e.*

$$\Delta t \left( \frac{Q_k}{Ve_k} + k_s + k_{tot} \right) + \frac{2(Ve_k - Ve_{k-1})}{Ve_k} + \frac{Q_k - Q_{k-1}}{Q_k} \geq 1$$

which results in instability. This was a concern with the experiments carried out in the current study, where rapid changes in discharge and volume resulted from the stepped reservoir releases. Instability was also an issue when modelling the River Dee, the high values of summer die-off requiring a short time-step.

In the more common steady-state applications for solute dispersion (*e.g.* Green *et al.* 1994) both  $Q$  and  $V$  are constant and the additional terms are not necessary. The effect of the additional terms are as follows. On rising flows, discharge and volume increase and hence the influence of  $X_{k-1}$  on the current value of  $X$  is reduced, this is sensible given that more water of different concentration is flowing into the system. On a hydrograph recession the effect is reversed and the influence of  $X_{k-1}$  on the next value is enhanced. Put another way, the influence of the input concentration  $U$  is reduced, which is logical since less water with concentration  $U$  is entering the hypothetical river reach.

The bed-store component of the model was tested in two formulations, one used a single channel store, the other used a distributed channel store, the two alternatives produced very similar results. It was also found that the model could be greatly simplified by reducing the mixing volume part of the model to a single constant parameter, and ignoring die-off which was very slow compared to the other processes operating, such that

$$X_k = bU_{k-\delta-1} + \{1 - (b + \Delta tk_s)\}X_{k-1} + \frac{\Delta t(Ne_k + Ni)}{V_k}$$

The model was tested on the Afon Clywedog data with and without the additional terms and found to produce very similar results in the different cases.

The mass-balance equation for the single bed-store component was as follows

$$\frac{dNT_t}{dt} = k_s V_t X_t - k_b NT_t - Ne_t$$

and

$$Ne_t = \left( \varepsilon_w \frac{d^2 Q_t}{dt^2} + \varepsilon_r \frac{dQ_{t-\tau_e}}{dt} + \varepsilon_s Q_t^2 \right) NT_t = New_t + Ner_t + Nes_t$$

where  $NT_t$  is the total number of organisms,  $\varepsilon_w$  is wavefront entrainment constant,  $\varepsilon_r$  is the wave response mean velocity entrainment constant and  $\varepsilon_s$  is the slow entrainment constant. The second derivative of discharge is used for wavefront entrainment, where the entrained organisms enter the wave and travel with it in the overrun. It was necessary to make this conditional so that it only worked on rising discharge. The mean velocity wave entrainment is time lagged by  $\tau_e$ . The organisms disturbed by the wave are lifted into suspension but are not trapped in the wave overrun and the resultant bacterial peaks propagate at the mean flow

velocity. On the Clywedog since the main phase of entrainment was associated with post-wave mean velocity transport the time delay was calculated from the relationships between wave speed and discharge and flow velocity and discharge.

The other alternative and equivalent version of the model uses a distributed channel store with multiple sub-stores for the rapid entrainment component. This version was found to produce almost identical results to the preceding single store version, but with the added utility of accounting for the discrete changes in storage throughout an event. In effect it maintains a flow memory of the history of entrainment in the channel. This allows for the hypothetical possibility that part of the channel can become depleted of organisms, but storage remains in regions flushed at higher flows.

The channel store is constructed by dividing the range of observed discharges for the study period,  $Q_0$  to  $Q_m$ , into a number of sub-stores,  $n$ , this gives what is in effect a "bed-store access interval",

$$\Delta Q = \frac{Q_m - Q_0}{n}$$

The number of sub-stores undergoing entrainment is

$$n_e = \frac{Q_t - Q_0}{\Delta Q}$$

where  $Q_t$  is the discharge at time  $t$ . At constant discharge  $n_e$  sub-stores are empty and further entrainment can only occur if the discharge increases. The initial rapid-supply number of organisms for the study reach,  $NR_0$ , is then divided into  $n - n_e$  sub-stores resulting in  $N_{t,j}$  organisms in each store;

$$N_{t,j} = \frac{NR_0}{n - n_e}.$$

The stores assigned between  $Q_t$  and  $Q_m$  at any time  $t$ , are free from entrainment and able to accumulate organisms settled from the flowing water. This allows for re-accumulation of entrained organisms into other areas where accumulation is still possible. In the short-term the areas of storage in the channel may maintain a "flow-memory". This would depend on the time elapsed between events, the water depth, settling velocity of the particles and the concentration of organisms in the water. A first flow event might deplete an area of storage and further entrainment would only be possible with a second higher discharge which would flush organisms from previously undisturbed areas. If the organisms can be redistributed on rising flows, then they can also re-accumulate in previously depleted areas once the conditions no longer cause entrainment.

The net change in rapid channel supply numbers of organisms  $NR_t$  is

$$\frac{dNR_t}{dt} = k_s V_t X_t - \epsilon_r \sum_{n=0}^{n_e} N_t = k_s V_t X_t - N e r_t$$

where all variables and constants are as previously defined. The reduction in numbers of organisms in each of  $n_e$  sub-stores undergoing entrainment is the sum of settlement minus entrainment and die-off,

$$\frac{dN_{t,j}}{dt} = \frac{k_s V}{n} X_t - (\varepsilon_r + k_b) N_{t,j}$$

where,  $k_b$  is the net bacterial die-off rate on the river bed. The change in storage in the stores not subject to entrainment is settlement minus die-off,

$$\frac{dN_{t,j}}{dt} = \frac{k_s V}{n} X_t - k_b N_{t,j}.$$

Settlement is treated as occurring in all parts of the channel not just in quiescent zones; accumulation will only occur if settlement exceeds entrainment and die-off (see Graham 1990).

The slow entrainment component of the channel supply is identical to the slow component of the single store formulation, such that

$$\frac{dNS_t}{dt} = k_s V_t X_t - \varepsilon_s Q_t^2 NS_t$$

where,  $NS_t$  is the size of the slow release channel store.

### **The full four input model**

The full water quality model is presented in this application with dynamic mixing and transfer to and from channel storage. In order to run the fully dynamic model it was necessary to produce time series of the driving variables with a 20 minute time interval. In the daily time-step model it was necessary to use effective parameters *i.e.* values that have been adjusted to compensate for instability problems. In this case it was possible to use more empirically determined parameter values based on the relationships presented elsewhere in the thesis. The basic structure of the model was similar to that presented above, but because of the specific circumstances relating to the characteristics of the Lower Dee reach approaching the Huntington Intake, special consideration had to be made to the connection of the various inputs to the model. The main issue was regarding whether the component inputs should be treated as an upstream input, which would route them through the mixing model of the reach, or whether they should be treated as direct inputs at the sampling location. The inputs to the model were:

- A point-source input influenced by die-off and diluted into the flow at the upstream end of the reach.
- A "slow" entrainment component ( $Q^2 N_s$ ) as presented above, but additionally influenced by die-off and then diluted into the flow upstream.
- Rainfall-runoff driven by the quickflow component of the estimated response of the remaining 800 km<sup>2</sup> of unregulated catchment downstream of the Manley Hall gauge.
- The rapid entrainment component of the fluvial dynamics model, simplified for improved computational efficiency.

A range of variations in the inputs and their interactions were investigated and found to give minor changes to the response of the model. An example of this involved splitting the rainfall

runoff and rapid entrainment inputs between the upstream and downstream ends of the model reach. This arrangement acknowledges that there will be rainfall-runoff and entrainment throughout the channel network up to the modelled reach input, and also inputs to the modelled reach itself. The reservoir release experiments and modelling exercise indicated that entrainment in the immediate vicinity of the sampling location was important in determining the dynamic behaviour observed *i.e.* the peaks and troughs of concentration would be more pronounced since the organisms had not undergone mixing and had not been dispersed and integrated along the study reach.

Die-off determines both the long-term and shorter term variations in concentration, so these experiments became an exercise in attempting to apply a more complex model to see whether the results could be improved-upon. The main aim was to attempt to model the peaks of concentration perceived to be the result of hydro-meteorological stimuli. One important aspect that was not built into the model or investigated in depth was the effect of tidal reversals, this should be built into any further investigations on the Lower Dee.

The details of the model application and the equations used are described below. The discrete time version of the model used to simulate the behaviour observed in the reservoir release models was

$$X_k = \Delta t \frac{Q_k}{Ve_k} U_{k-\delta-1} + \left\{ 1 - \left( \Delta t \left( \frac{Q_k}{Ve_k} + \frac{v_s}{z_k} + k_{tot} \right) + \frac{2(Ve_k - Ve_{k-1})}{Ve_k} + \frac{Q_k - Q_{k-1}}{Q_k} \right) \right\} X_{k-1} + \dots$$

$$\dots \frac{\Delta t (\varepsilon N r_{k-\delta e} + Q_k^2 N s + N i)}{V_k},$$

where, since the upstream inputs were negligible the entrainment inputs were added at the reach outlet. In the application to the Dee, although a variety of arrangements were investigated, the following arrangement will be described since it was deemed to be the most physically reasonable and produced justifiable results:

$$X_k = \Delta t \frac{Q_k}{Ve_k} (Ues_k + US_k + Urq_k) + \left\{ 1 - \left( \Delta t \left( \frac{Q_k}{Ve_k} + \frac{v_s}{z_k} + k_{tot} \right) + \frac{2(Ve_k - Ve_{k-1})}{Ve_k} + \dots \right. \right.$$

$$\left. \dots \frac{Q_k - Q_{k-1}}{Q_k} \right\} X_{k-1} + Xe_k,$$

where, the input concentration from slow entrainment,  $Ues_k = \frac{(1 - k_{surf}) Q_k^2 Nes}{V_k}$ , and  $Nes$  is a constant.

The point-source input

$$US_k = \frac{(1 - k_{surf})(US_{k-1} V_k + Ns)}{V_k},$$

also a constant.

The quickflow runoff input component

$$Ur_q_k = \frac{\varepsilon_r (Qq_k - Qq_{k-1}) Nl_k}{V_k}, \text{ where } Nl_k \text{ is the size of the land-store and}$$

$Nl_k = (1 - (\varepsilon_r (Qq_k - Qq_{k-1}) + k_{surf})) Nl_{k-1} + Nil$ , where  $\varepsilon_r$  is the runoff entrainment rate and  $Nil$ , the number of organisms input to the land store at each time step, is a constant. The quickflow runoff component is conditional on the quickflow discharge,  $Qq_k$ . For  $(Qq_k - Qq_{k-1}) < 0$ ,  $(Qq_k - Qq_{k-1})$  is constrained to zero and hence  $Ur_q_k = 0$ , *i.e.* if the discharge is not increasing there is no entrainment from the catchment surface. In addition the land store bacterial number is prevented from falling below zero.

The die-off rate used is  $k_{surf} = k_T + \alpha I_0$ . The surface irradiance,  $I_0$ , is used rather than the depth averaged irradiance,  $I_{z,avg}$ , since the organisms are derived at or near the surface of the water, or the surface of the land.

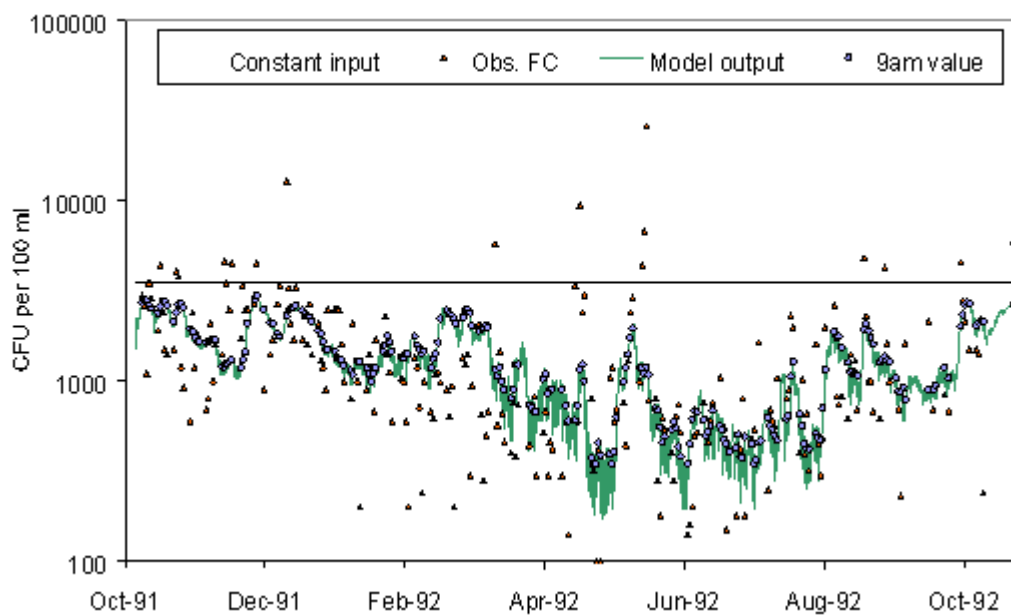
$$\text{The rapid entrainment input } Xe_k = \frac{\varepsilon(v_k - v_{k-1}) Nr_k}{V_k}, \text{ where } Nr_k \text{ is the size of the channel}$$

store and is the sum of additions by settlement and losses due to entrainment and die-off;  $Nr_k = (1 - (\varepsilon(v_k - v_{k-1}) + k_z)) Nr_{k-1} + v_s / z X_k V_k$ , and  $v_k$  is the flow velocity. Reach averaged flow velocity is used to drive the entrainment function in this application since it biases the majority of entrainment towards changes in flow at the lower end of the discharge range. The rapid entrainment component is conditional on increases in velocity in the same way that land-store runoff was conditional on quickflow discharge. For  $(v_k - v_{k-1}) < 0$ ,  $(v_k - v_{k-1})$  is constrained to zero and hence  $Xe_k = 0$ . In addition the land store bacterial number is prevented from falling below zero. The die-off rate used for the channel store was  $k_z = k_T + \alpha I_z$ , where  $I_z = I_0 e^{-\eta z}$ , the light intensity at the base of the water column.

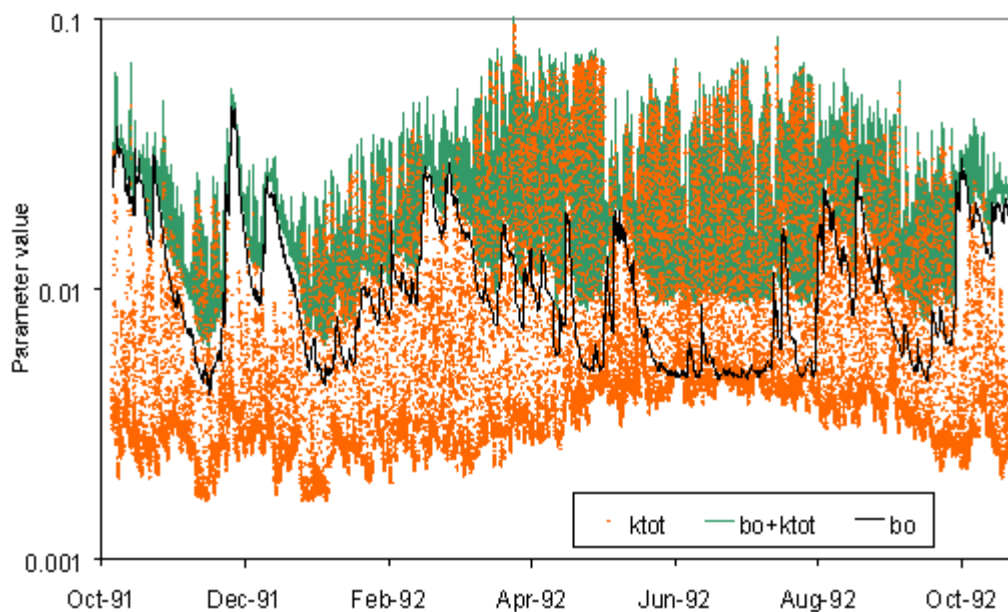
### **A reduced model with constant input**

In this version of the model there is only one input which is a constant value, the time variable parameters of the model produce the dynamics of the resulting output concentration (Figure 1). The model fit was improved by increasing the reach length to reduce the  $b_0$  parameter to the same order of magnitude as the die-off parameter (Figure 2). The model equation was

$$X_k = \Delta t \frac{Q_k}{Ve_k} U_\infty + \left\{ 1 - \left( \Delta t \left( \frac{Q_k}{Ve_k} + k_{tot} \right) \right) \right\} X_{k-1}.$$



**Figure 1.** Model output with constant single input value of 4,000 cfu per 100 ml.



**Figure 2.** Model parameters for the single constant input model of faecal coliform concentrations in the River Dee.

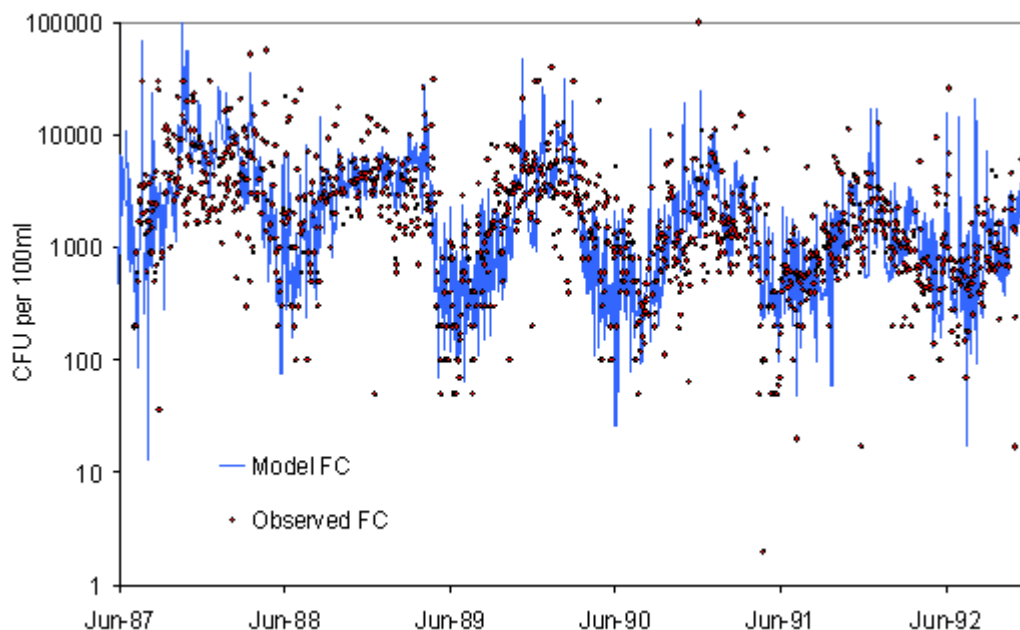
The results with daily data were similar to those presented above, although for the longer period it was necessary to introduce a trend multiplier to fit the data over the seven year record (see below).

### A two input model with daily data

Output from this version of the model is presented in Figure 3 below (Table 1 details the parameters used). This version of the model uses the mixing volume formulation with the rainfall runoff delivery model. The slow entrainment/baseflow input component was set to zero and the settling velocity was also set to zero, effectively switching off the channel entrainment function. Point source input was set to be constant, and treated as the upstream input to the mixing model undergoing die-off within the mixing volume component of the model. The quickflow runoff input was added at the model outlet thus retaining its spikiness since it has not been mixed into the reach. The model equation was:

$$X_k = \left( \Delta t \frac{Q_k}{Ve_k} Us_k + \left\{ 1 - \Delta t \left( \frac{Q_k}{Ve_k} + k_{tot} \right) \right\} X_{k-1} + Xrq_k \right) trend$$

Where  $Xrq_k$  is  $Ur q_k$  as described above. The point-source input  $Us_k = Ns/V_k$ ,  $Ns$ , is a constant number of point-source organisms per time step.



**Figure 3.** Model output for the daily ADZ with quick flow input and constant upstream input.

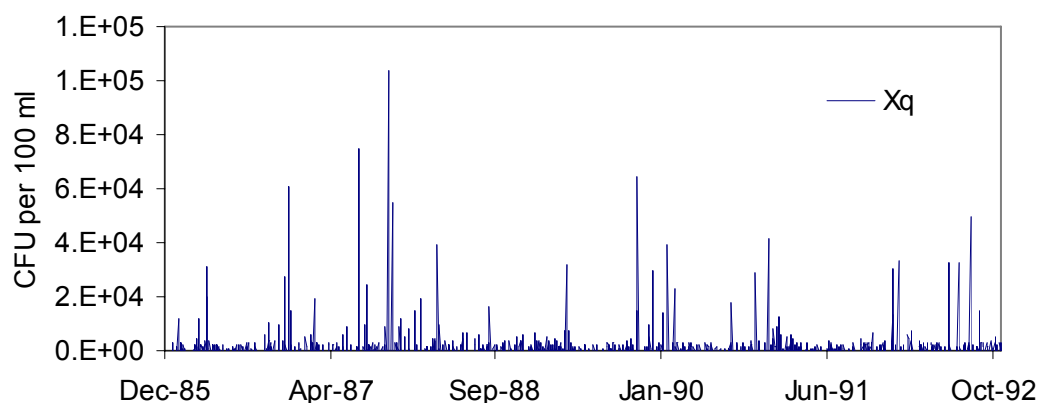
The "*trend*" term was used to adjust the result of the output, the trend indicates a 65% reduction in the long-term faecal coliform concentration over the seven years of observation. The cause of the declining concentrations was not investigated but could be the result of improvements in wastewater treatment or catchment management; the ammoniacal nitrogen concentration shows a similar decline. Alternatively it may have been due to the increase in temperature observed over the period of record. This effect was not detected in the die-off rate, which was probably due to the temperature component forming a small part of the die-off rate. Neither was any account taken of variation in solar flux when estimating irradiance. A

subsequent examination of a 20 year record of satellite measurements of the solar flux (Pap *et al.* 1999) shows an 11 year cycle that has a 0.1% amplitude, consequently the influence of this variation on  $k_i$  is insignificant. The changes in air temperature from 1986 to 1992 do increase and decrease with the variation in solar flux and this apparent relationship might be worthy of further investigation in relation to bacterial die-off. Further examination of solar flux data might however be of value since different wavelengths of light are known to have a stronger die-off effect than others (*e.g.* Gameson & Saxon 1967), and some spectral components of the solar flux vary to a greater extent than the overall variation in the overall flux (*e.g.* Donnelly 1991).

**Table 1.** Parameters and constants for the model presented in Figure 3.

Parameter, constant	Value (units)
Dispersive fraction, $Df$	0.25
Reach length, $L$	6000 (m)
Upstream input, $U_s$	$3.3 \cdot 10^7$ (cfu per 100 ml)
Quickflow input, $N_{iq}$	$1.8 \cdot 10^9$ (cfu)

Figure 4 shows the quickflow runoff component; an interpretation of what it does in the model is that it represents rapid runoff from the catchment surface via non-matrix through-flow and direct runoff. The supply of organisms was assumed to be limited as indicated by Hunter *et al.* (1992). The model die-off and mixing terms accounted for the majority of the variability in concentration at the time scale represented. The die-off term accounted for the summer reductions in concentration and the mixing term accounted for the way increased discharge transported organisms closer to the sampling point and reduced the time in transit and potential for die-off.



**Figure 4.** The quickflow runoff input component for the simulation of the faecal coliform behaviour in the River Dee.



This version of the model reproduced the longer-term changes in faecal coliform concentration and also some of the shorter-term flashiness resulting from rainfall effects. A similar result was achieved without the mixing-zone component, by simply summing the various inputs to the system (see below).

### ***A three input model without mixing-zones***

While investigating building the full mass-balance model it was found that the variations in faecal coliform concentration at Huntington on the River Dee could be reproduced to a reasonable degree using a very simple scheme (Figure 5). This model worked on the assumption that there were three typical inputs contributing to the FC concentration at the point of interest on the river. These were: an upstream input *i.e.* the organisms travelling with the water flowing down the main channel of the river: a local rainfall-runoff input from the catchment immediately feeding the reach approaching the sampling location: a point-source input from piped discharges to the system. The model equations were as follows:

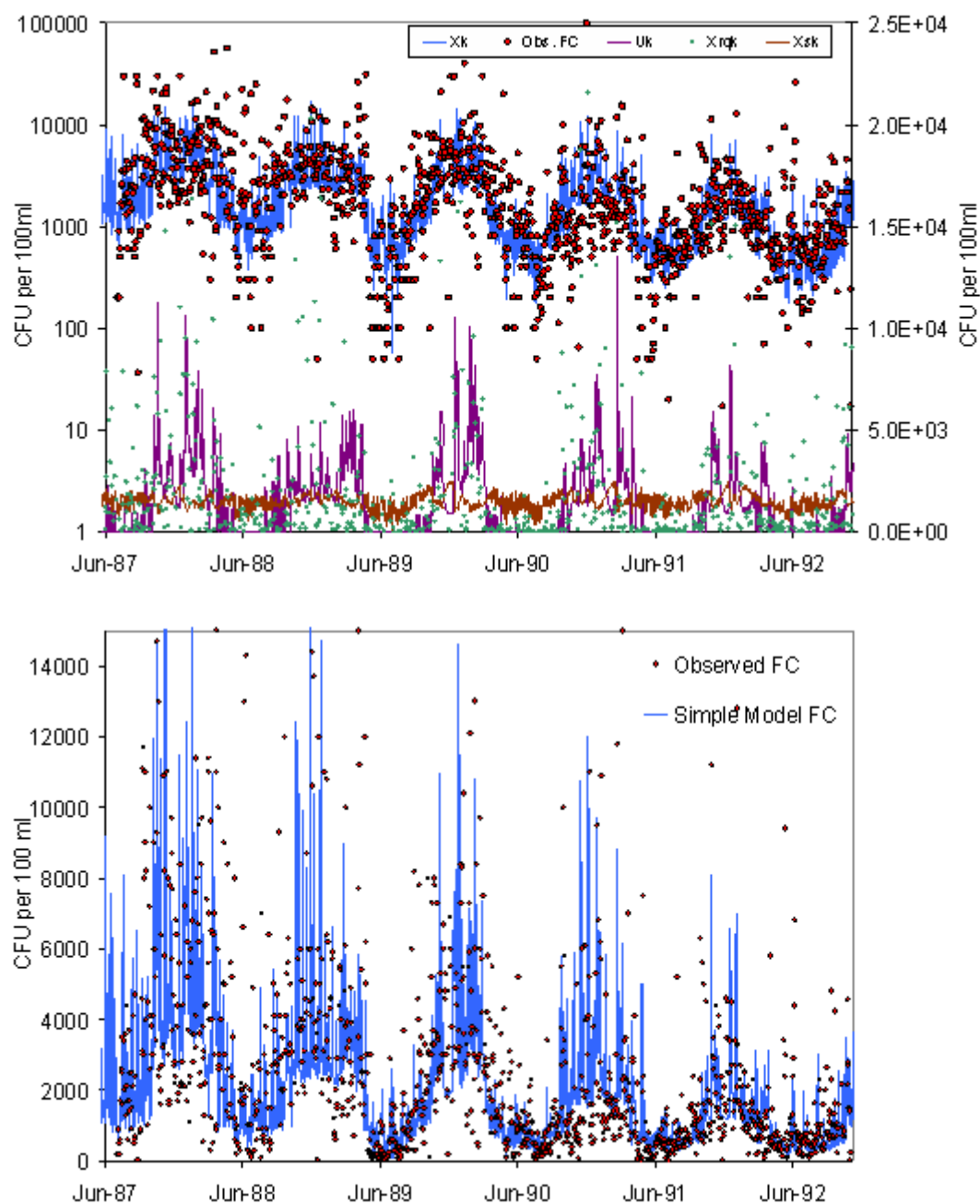
$$X_k = (U_k + Xs_k + Xrq_k).trend$$

where  $Xrq_k$  was  $Urq_k$  as described above, the upstream input concentration,

$$U_k = \frac{(1 - k_{tot})Q_k Nu}{V_k},$$

and  $Nu$  is a constant input per time step. The point-source input,  $Xs_k = (1 - k_{tot})Ns/V_k$ ,  $Ns$ , the number of point-source organisms per time step, which was also a constant.

The only adjustments used to fit the model were made to the input constants (numbers of organisms) of each of the driving variables. This method was repeated with the enhanced frequency data and demonstrated that although the longer-term variations in concentration could be recreated, there was still insufficient information to reproduce the short-term variations in concentration.



**Figure 5.** Variations in model output and observed faecal coliform concentrations for the River Dee at Huntington Intake, plotted on logarithmic and natural scales.