'Who's to Blame and Who Pays' – Understanding bathing water management through the integration of river, network & coastal models

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Abstract
The revised Bathing Water Directive (rBWD) will repeal the current Bathing Water Directive (cBWD) in 2015 (European Parliament, 2006). The first round of bathing water studies in the UK has been completed and suggests that the rBWD is a significant change from the cBWD in terms of the interpretation of results and the outcomes. The differences in the methods of classification give rise to a shift in the relative importance of different types of bacterial sources. Whereas previously, high concentration but less frequent events dominated cBWD compliance, this does not appear to always be the case under the rBWD, with more frequent, lower concentration discharges often being highlighted as key contributors. This has required a corresponding response in terms of modelling technologies and scope, with the development of rural runoff quality models and optimisation techniques to accurately represent bacterial inputs and ensure that compliance is achieved efficiently as the optioneering process is now more complex. This greater emphasis of lower level frequent discharges includes inputs beyond the control of water companies and makes "blame" less easy to apportion. In many cases, significantly improved water quality as measured by the rBWD can only be achieved by using a partnership approach involving regulators, water companies, local councils and land owners.

Some quirks of the rBWD related to the assumed log normal distribution used in sampling analysis and the application of 95% confidence levels have also been established which must be understood by all stakeholders to ensure that real improvements are made to the water quality of our bathing waters and rBWD targets are met at least financial cost. The knowledge gleaned during the first programmes of bathing water studies suggests that optimised solutions are possible with the latest modelling techniques but that shifts of thinking are also required for successful delivery of an rBWD programme.

Keywords
catchment modelling, coastal modelling, diffuse pollution, revised bathing water directive, sewer modelling

Introduction
The revised Bathing Water Directive (rBWD) will repeal the current directive in 2015. The approach for measuring performance in the rBWD is a shift from the threshold type approach used for the current directive. This represents an accompanying shift in the types of most influential sources and also the logic for developing optimal solutions. The first round of rBWD studies in the UK has revealed some subtle differences in the results provided and some counter-intuitive solutions.

BW bacteria sources
Inputs of bacteria that can impact at bathing waters can derive from a number of sources. These are most notably:

- Combined Sewer Overflow (CSO) spills during heavy rainfall
- Storm Water Outfall (SWO) discharges during any rainfall
- Continuous discharges from Wastewater Treatment Works (WwTWs)

- Private and trade foul discharges
- Cross connections
- Washoff from rural areas directed via watercourses
- Urban runoff directly to beaches
- Bird colonies and wildlife
- Dog walking on beaches

The influence that any of these sources will have on any particular bathing water will vary greatly on a case by case basis. This will depend on the scale and concentration of the discharge and the degree to which the discharge is dispersed and decayed prior to reaching the bathing water. Although this is a relatively complex process with many local features and influences involved, fortunately, most of these sources and influences can be reasonably accurately modelled.

Modelling approaches

The vast majority of the Urban Drainage Group will be familiar with the provision of sewer network and Wastewater Treatment Works (WwTW) model data for bathing water assessment. A long time-series of rainfall (typically 10 years) is run through the models, and hydrographs for both intermittent and continuous discharges are used as marine modelling inputs along with details of each discharge so that an appropriate mean concentration can be applied for the required bacteria types. This is an established process, and the use of well verified models and accurate measured data result in a good degree of confidence in these bacterial inputs.

Previous studies have shown that bacterial loadings washed off rural catchment areas at times of rainfall can be significant enough to warrant their inclusion in bathing water studies (Crowther et al, 2002). For previous cBWD investigations, an approach using a hydrological model and mean bacterial concentrations for low and high flows was used. However, this approach misses some of the variability in river loads that is important in defining the full distribution of concentrations under the rBWD. Consequently Intertek developed a rural bacterial washoff model that represents bacterial inputs derived from agricultural areas in response to rainfall and the bacterial loading of the land.

The CATCHMENT IMPACT modelling approach utilises catchment hydrology, land use and livestock data to derive dynamic flows and 'pollutographs' of bacterial load in response to rainfall. CATCHMENT IMPACT has an integral hydrological model which is used to develop upstream boundaries for a river model based on the Revitalised FEH Method (CEH, 2005). The water quality model enables the inclusion of dispersion, attenuation and the simulation of conservative and first order decay parameters. Interactions between water quality parameters are also modelled such as:

- The oxygen cycle including decay of carbonaceous BOD, oxidation of ammonia to nitrate and reaeration
- Nutrient modelling
- Priority substances
- Bacterial decay

Point sources are input into CATCHMENT-IMPACT and routed through the river system. These inputs are subject to the dispersion, attenuation and decay processes described above. In long river systems with areas of storage and dams, these can be important effects to represent.

For diffuse pollutants, in addition to the transport processes, the land use and livestock data are used in conjunction with both rainfall and runoff data to determine the build up, decay, washoff, infiltration and transport of livestock waste on the catchment surface. The impact of this washed off load can then be assessed either in the watercourse or the receiving marine environment.

Urban diffuse inputs can be estimated using simplified runoff modelling methods if they are not included in drainage network models and then applying default surface water concentrations. A site visit should be made for these cases so that surface flow paths can be determined.

Direct inputs from dog and wildlife inputs are more difficult to model due to their less predictable nature and a dearth of modelling data. Bacterial loading estimates can however be determined for these inputs based on local population information (Alderisio & DeLuca, 1999) and can be included in the analysis, albeit with a lower level of confidence.

Figure 1 – CATCHMENT-IMPACT Modelling Schematic

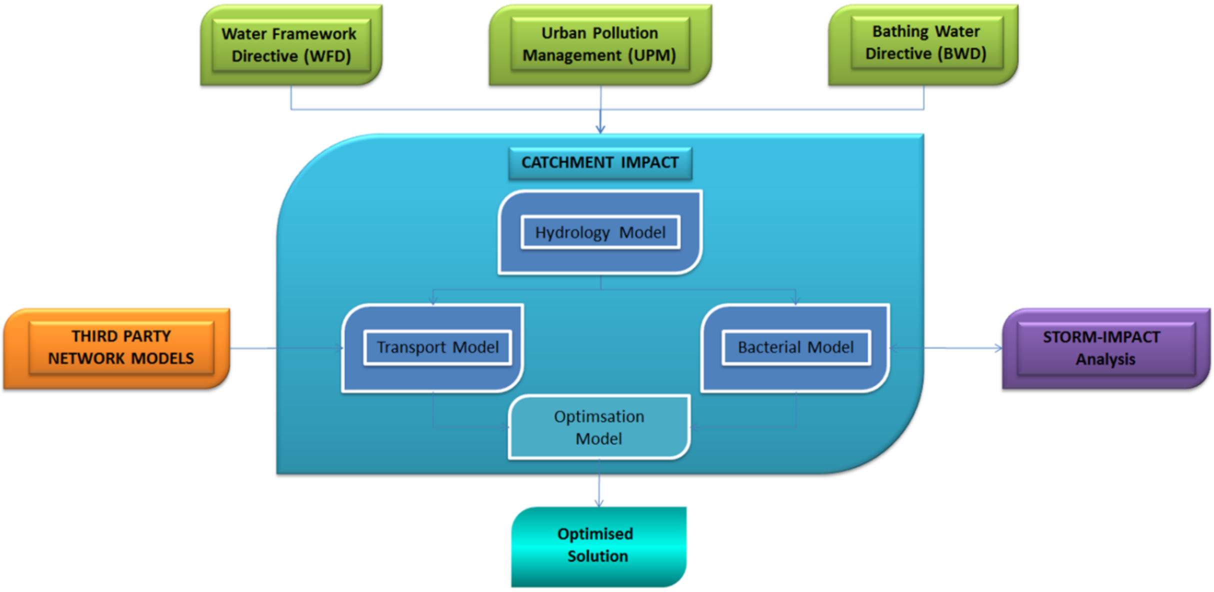


Figure 1 shows a schematic of how the main modelling elements link together with CATCHMENT-IMPACT in the analysis. All of the inputs are combined with detailed marine modelling and statistical approaches to determine a compliance record or classification for the bathing water being studied. No matter whether the current or revised directive is being considered, the compliance record is constructed in the same way by combining detailed marine model outputs with statistical analysis in what is called the STORM-IMPACT model. STORM-IMPACT has been as the key tool for compliance assessment and source apportionment in the UK for more than 10 years.

The ten year stochastic drainage system, river and continuous outputs are combined with tidal and wind conditions to predict the total bacterial concentrations at the bathing water. The modelling is done in such a way that the individual contribution of each modelled source is determined independently (source apportionment). However, the interpretation of these results within the context of the two bathing water directives has been found to vary depending upon the directive considered.

The information gleaned from the STORM-IMPACT assessment is fed back into the optioneering process using CATCHMENT-IMPACT, engineering knowledge, stakeholder engagement and where appropriate optimising tools to develop the best option for any given bathing water.

cBWD and rBWD differences

The classification systems for the current and revised directives are set out in Table 1 and
Table 2 respectively. The most significant difference between the two schemes is that the cBWD is defined as the number of times that an upper limit can be breached, whereas the rBWD classifications are defined by the concentration at a given percentile for the distribution of concentrations experienced at the bathing water. The effect of this is to essentially make every contributing load count towards the rBWD classification at all times throughout the bathing season, whereas the cBWD focuses on peak concentration events that may cause breaches. This

fundamental shift can also move some of the focus away from the "peaky" intermittent type sources for the rBWD that were targeted in the cBWD (e.g. CSOs).

Table 1- cBWD Classification Limits

Parameter	Faecal coliforms (FC/dl)	Faecal streptococci (FS/dl)	Total coliforms (TC/d)
Guideline	100 (16/20 must be below)	100 (18/20 must be below)	500
Mandatory	2000 (19/20must be below)	n/a	10,000
Fail	>2,000	n/a	>10,000

Table 2- rBWD Classification Definitions

Parameter	Escherichia coli (EC/dl)	Intestinal enterococci (IE/dl)
Excellent	250 (95%ile)	100 (95%ile)
Good	500 (95%ile)	200 (95%ile)
Sufficient	500 (90%ile)	185 (90%ile)
Poor	>500 (90%ile)	>185 (90%ile)

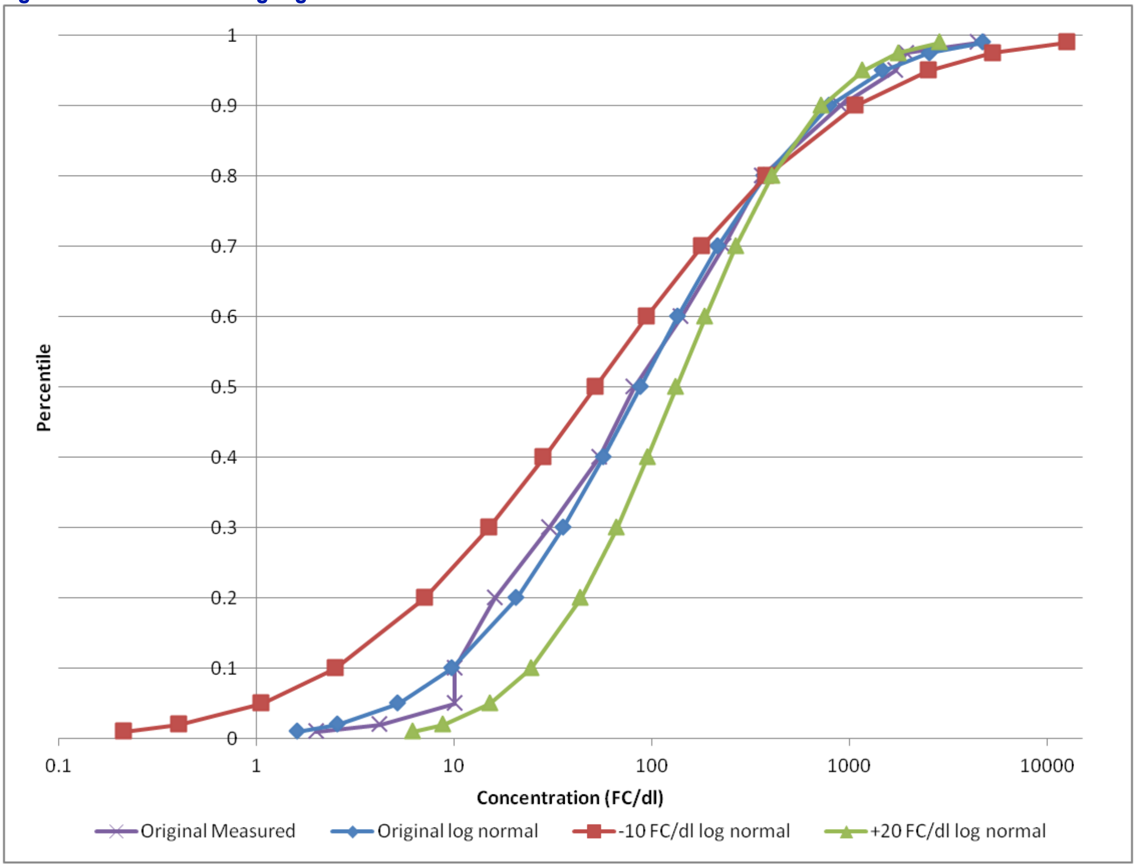
The rBWD requires that a minimum of 20 samples are taken at a representative sampling point within each designated bathing water. The bathing water classifications are determined on a four year rolling basis using 80 samples. Crucially, as this sample size is small in terms of determining representative percentiles, a log-normal distribution is fitted to the data and used to extract the required percentile. This in itself can have significant implications, as the studied bathing water may not display a log normal distribution and an accurate model should replicate the actual distribution of the bathing water quality rather than the assumed log-normal fit.

Additionally, the use of the log-normal distribution can result in some statistical anomalies during optioneering. Figure 2 shows the effects that fitting an assumed distribution can have. The purple line marked with crosses is the distribution of measured concentrations at a bathing water on the east coast of Scotland. The blue line marked with diamonds is the fitted log-normal distribution. As can be seen, the log-normal distribution fits the data very well. The other two lines represent cases where the base data was modified before fitting the log-normal distribution. For the red line, all recorded concentrations were reduced by 10 FC/dl and for the green line an increase of 20 FC/dl was applied. Although these are arbitrary simplified modifications to the sampling record, the presence of a nearby, large continuous WwTW discharge means that base changes in loading would be possible.

The impacts of these changes are counter-intuitive as a result of the distribution being defined by the log-normal shape, the mean and the standard deviation of the data. In the case of all data being reduced by 10 FC/dl, at lower concentrations, the change in the base data results in a consequent reduction in the percentile values determined by the distribution as one would expect. However, as the log-normal scale magnifies the influence of the lower end of the distribution, the standard deviation is essentially widened. This makes the distribution shape much flatter and actually results in an increase in the higher percentile values derived using the distribution. At the critical 90 and 95 percentiles for the rBWD classifications, the reduction of 10 FC/dl from all base data results in an increase of 25% and 40% respectively. The effect is even greater at higher percentiles.

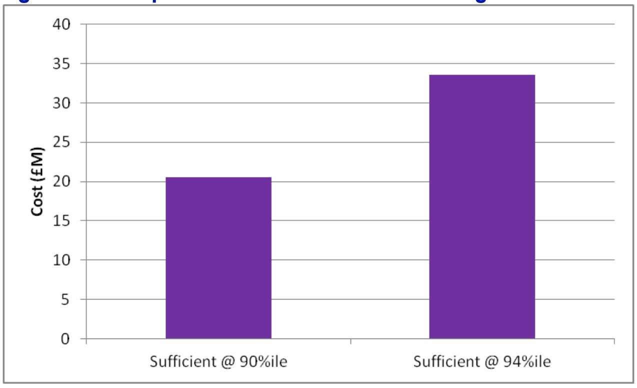
Conversely, in the case of the green line marked with triangles, the base data were modified by adding 20 FC/dl to all values. This results in a narrowing of the standard deviation as the magnified smallest data values are eliminated from the distribution. As would be expected the lower percentile values are all increased by a small amount. However, at high percentiles, despite adding 20 FC/dl to every value, the distribution derived values actually reduce. At the 90 percentile this reduction is 10% and at the 95 percentile the reduction is 25%.

Figure 2 - Effects of fitting log normal distributions



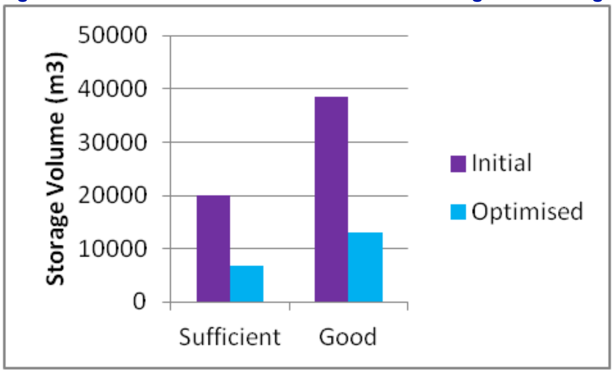
Under the cBWD, options were developed to meet the required standards with 95% confidence – referred to as the design standard. This means that natural variability of the sampling should be taken into account and that you must design any improvements to a higher standard than the face value requirements of the legislation. A similar approach was adopted for the rBWD studies. To achieve the standards with 95% confidence, the required Sufficient concentrations should be met at the 94th percentile rather than the 90th and the Good and Excellent concentrations at the 97.5 percentile instead of the 95th. Several of the recent rBWD studies carried out to date have shown that the effect of this can be to bring several new sources into the mix of contributing discharges that are not required to be considered at the face value percentiles. These sources can include low concentration, continuous discharges such as watercourses and more remote private discharges. This means that options developed at the design standards by considering only water company discharges can very easily become enormously impractical with high associated costs. The cost difference in an ongoing Scottish Water study was found to be 65% more when considering the design standard percentiles rather than the face value percentiles (Figure 3).

Figure 3 - Comparison of Face Value and Design Standard Percentile Solutions



At these higher design standard percentiles, optimising solutions becomes even more important. Initially high cost solutions developed at a Northumbrian Water bathing water were optimised looking at improved operation of the network to maximise in-pipe storage in addition to targeting new storage at key locations. This additional optimisation work resulted in a reduction of the additional storage volumes by approximately 66% for achieving both the Sufficient and Good standards (Figure 4).

Figure 4 - Northumbrian Water Solution Savings Following Optimisation



Key Findings of the rBWD Programme

The first round of bathing water studies has been completed. This has allowed some conclusions to be made and some "lessons learnt" regarding the rBWD. These can be summarised as:

- All inputs contribute to the distribution of concentrations and therefore the rBWD percentiles. This makes "blame" more difficult to apportion.
- There is a dearth of data for some sources that play a significant role in the classifications achieved under the rBWD.
- Initial thoughts that remote discharges are not likely to impact have often been shown to be wrong. The area of influence for the rBWD would appear to be wider than the cBWD particularly for low level background inputs.
- Accurate river modelling is more important under the rBWD. This should distinguish between agricultural, private, water company and illicit discharges. CATCHMENT-IMPACT modelling has allowed cross connections and individual farm areas to be identified and their impacts assessed.
- Continuous discharges such as WwTW, septic tanks and cross connections require careful management and should be included in the blend of solutions.
- Design assumptions of 95% confidence can lead to massive storage requirements and can include other discharges not previously implicated at face value 90 and 95th percentiles.

- The use of the log-normal assumed distribution can lead to strange anomalies and inappropriate theoretical solutions that should be understood by designers, water companies and regulators.
- True optimisation looking at storage, WwTW performance, network operation, livestock practices and catchment management is required to meet the directive at least cost.

Recommendations

As a result of the findings detailed above, the following recommendations are made to improve future rBWD, shellfish, virus and water quality investigations.

- More data are required for a range of inputs. Spatial and temporal sampling on significant watercourses, bacterial sampling of effluent and event monitoring at the most significant (in terms of impact) discharges are all required. This is a subtle shift in sampling strategies and leans more towards management rather than purely compliance monitoring.
- Proactive management of cross connections is likely to be cost effective and have a greater impact than constructing more storage in a number of cases.
- Detailed modelling of river systems will allow more accurate representations of bacterial inputs and allow previously untested interventions to be quantified for improvements (e.g. changes in livestock management).
- Stakeholders should approach the optioneering phase collaboratively to ensure that the most cost effective solutions are considered.
- Optimisation tools should be used in conjunction with the stakeholder approach to ensure compliance at least total expenditure. This requires thinking beyond storage.
- The effects of the log-normal distribution require to be considered during the optioneering stage.

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