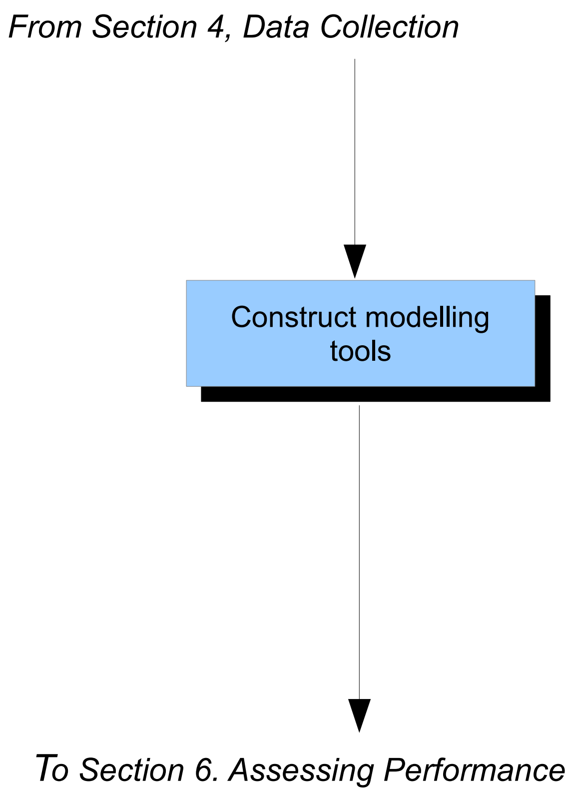
Section 5
Construct UPM Tools



5. SECTION 5: CONSTRUCT UPM TOOLS

This section is divided into a series of subsections that provide information on the types of modelling tools that may be required to carry out a UPM study. Each subsection covers a particular generic type of modelling and identifies typical applications, model requirements and modelling approaches based on the use of a range of simple to complex modelling tools for each type of application. Specific modelling tools and software products are not identified. The issues related to the building, calibration, verification of each type of modelling tool are considered in relation to 'fitness for purpose'. Generic limitations of current modelling tools are considered.

The section is subdivided as follows.

5.1 Introduction to modelling tools

The application of simulation modelling tools forms the key activity in a UPM study. Modelling tools are used to predict the performance of urban wastewater systems and to assess the resulting environmental impact of wet weather discharges against appropriate environmental criteria, as discussed in Section 2. A range of modelling tools are available to cover all components of a UPM study. Section 3 considered the selection of appropriate models and modelling capability, in relation to the requirements of a site specific application. This section considers the generic types of modelling tools that may be required to implement a study in terms of wastewater transport and treatment, and environmental impact assessment in river, estuary and marine receiving waters. These types are:

- rainfall modelling;
- sewer flow and quality modelling;
- sewage treatment works modelling;
- river impact modelling;
- estuary impact modelling;
- marine impact modelling; and,
- integrated urban pollution modelling.

Each subsection considers the issues related to the type of modelling and models available in relation to the requirements for typical applications of the UPM Procedure. Attention is focused on making models 'fit for purpose' for subsequent modelling scenarios, through the stages of model building, calibration and verification. Applications issues are also considered in relation to the selection of boundary conditions and initialisation, where appropriate.

Specific modelling tools and software products are not identified, therefore, data requirements and modelling approaches are illustrated in general terms. Users should select a particular model in the light of the requirements of the study identified in Section 3.

5.2 Rainfall modelling

By definition, rainfall is the main driving force affecting the wet-weather performance of the receiving environment, as the rainfall impacts on the discharges from sewers and STWs diffuse inputs and also run-off from natural surfaces. The way in that rainfall is represented, or modelled, is of crucial importance to the correct understanding of wet-weather problems and the subsequent development and testing of solutions.

This section describes the selection and use of long rainfall time series in UPM studies, and is subdivided as follows:

5.2.1 Rainfall requirements for UPM modelling;

5.2.2 Historical rainfall time series;

5.2.3 Synthetic rainfall time series;

5.2.4 Event definition.

5.2.1 Rainfall requirements for UPM modelling

Rainfall data are required to drive the models that simulate the wet weather performance of urban wastewater systems and depending on the amount of detail in the study, the receiving environment. Long time series of rainfall events are needed to provide as full an account as possible of the variability in rainfall over the chosen time period and to give the user maximum flexibility in making different event selections depending on the question being addressed. The effort required in ensuring the reliable representation of rainfall inputs is likely to be considerably less than that required for other modelling components. However, rainfall modelling cannot be overlooked given the crucial importance of rainfall in gaining an accurate understanding of wet-weather performance.

Variability of rainfall

UPM modelling studies require long series of rainfall data that are representative of the local catchment. The length of the record is important to ensure that the data do not represent a particularly wet or dry period. A record of at least ten years duration is recommended for UPM applications and consideration should be made to extending this to 25 years if there is evidence of significant variation across a 25 year period. The various elements of UPM modelling require rainfall in a range of formats. The main requirements are for values in hourly or shorter timesteps for deterministic modelling. Five minute timestep data has been used, and some studies have used 15 minute timestep data with no significant detriment to modelled sewer system performance. Long time series of hourly rainfall data are generally needed as input for simplified sewer flow models.

Spatial variability of rainfall may require the application of more than one time series for large catchments. This is more easily made use of by using historic rainfall data if sufficient long term gauges are available.

5.2.2 Historical rainfall time series

Historic rainfall data can be obtained from either raingauge data or more recently rainfall radar data.

Digitised rainfall data from raingauges are available for a number of UK sites and may be obtained from the Meteorological (Met.) Office or some other local sources, such as the municipality, environmental regulator or local water company. The timestep of the datasets will range between short timestep data of minutes up to daily rainfall records. For direct use in deterministic models generally, the timestep of data will need to be short, and not more than 15 minutes. Some simpler lumped catchment models can use data at hourly timesteps.

There are a number of factors that need to be taken into account when using raingauge data, including:

- Selection of the most appropriate gauges;
- Spatial variability of rainfall across large catchments;
- Length of rainfall record; and,
- Amount of missing data and accuracy of gauge data.

Selection of the most appropriate raingauge site to represent the catchment to be modelled needs to take account of the distance of the gauge from the catchment in question, the elevation of the gauge compared to the catchment, and the appropriateness of the annual rainfall at the site compared to the catchment.

It may be possible to use more than one raingauge to provide rainfall inputs, dependant of course on the availability of data when undertaking analysis of large catchments,. This will allow an element of spatial variation of rainfall across the catchment. The advantage of using historical data for this purpose is that the data is dated.

The rainfall record length needs to be long enough to take account of the inherent variability of rainfall. Both intensity and duration of rainfall is important in assessing discharges from CSOs and storm tanks and some discharges are very sensitive to changes in antecedent conditions. Hence, a wide a range of rainfall data is required over as long a time period as possible. If a relatively short rainfall record is available, reviews should be undertaken of long rainfall records to ensure the period of the rainfall record is consistent with long term trends. This could be done, for example, by comparison with a daily rainfall record. Historical data should always be screened and checked for missing data values.

Although use of radar rainfall data resolves any spatial variation issues, the duration of the available rainfall radar data set usually precludes its use in UPM analysis. If there is a suitable duration of data available, checks should be made on the perceived accuracy of the data when compared to raingauge data and missing data.

5.2.3 Synthetic rainfall time series

Although there are more short timestep raingauge data available for longer periods, there are occasions when either there are insufficient suitable rainfall sites, or suitably long records available for the use of historic data. The use of synthetic rainfall data will be required in these instances.

There are various stochastic rainfall generators available that will generate long rainfall times series containing all of the characteristics of historical data for any location in the UK. Input data requirements to generate a series are generally related to parameters such as average annual rainfall, grid reference, altitude and distance from the coast. Generally, these can also be seeded with measured rainfall data.

The stochastic rainfall series generated needs to be at a suitable timestep, generally hourly or 5 minute dependent on the type of modelling tool being used.

Any synthetic rainfall series generated should be checked against local rainfall records. A comparison between the synthetic series and the local rainfall record should be undertaken for:

- average and range of annual rainfall totals;
- average and range of monthly rainfall totals; and,
- frequency distribution of daily rainfall totals.

It is important that any future changes are evaluated and incorporated as part of the agreed environmental planning framework (Section 3), particularly in relation to climate change.

5.2.4 Event definition

Unless continuous simulation modelling is to be employed, a major first step before using any long rainfall series (either historical or synthetic) is to produce an event file that contains only the events required for subsequent analyses. The definition of an 'event' is based on the dry period separating events. This is open to some interpretation and should be considered on a catchment by catchment basis. The definition of a suitable inter-event dry period should be related to the time taken for the sewerage system to return to baseflow conditions, including the emptying of any upstream storage. However, event definition will also be driven by how calculation of failures against standards is analysed. If this is carried out on an individual event basis, the use of very long rainfall events may cause under-prediction of failures.

During the event definition process, the antecedent conditions and catchment wetness parameters can be calculated for each discrete rainfall event. These parameters are required for estimating run-off in sewer flow models.

5.3 Sewer flow and quality modelling

This section addresses the role played by sewer flow and sewer quality models (SFMs and SQMs) in UPM studies. The main processes that affect the pollutant loads discharged from sewer systems during wet weather periods are considered. Also, the principles of sewer quality modelling are discussed.

This section is subdivided as follows.

5.3.1 Hydraulic performance of sewer systems

An adequate understanding of how a sewer system performs hydraulically is a basic requirement for any UPM study. It is essential to be able to predict how flows will vary during storms before any sensible predictions of the spatial and temporal variations in pollutant concentrations and loads can be made.

a) Detailed sewer flow models (SFMs)

Detailed SFMs require a representation of the pipe network together with information about areas and populations connected to each pipe. The level of detail of this representation can be varied depending on the accuracy required. Typically, surface run-off hydrographs can be produced from a specified rainfall profile and then routed through the modelled pipe network. Flows and depths are calculated throughout the network at each timestep and surcharge and flooding at manholes are predicted. In addition, ancillaries such as CSOs, tanks and pumping stations can also be represented.

The simulated performance must be checked to ensure that it accords with actual performance before a SFM is used to design major upgrading measures on an existing system. This checking process is called verification and may involve evaluation against historical data (for example, flooding records) and against specific field measurements of flows and depths during storms (CIWEM Urban Drainage Group, (2002)).

A SFM should be used to represent not only the combined sewer system but also any major separate storm water systems in the urban catchment area. This will help to ensure that all flows and loads discharged to the river during a storm event are accounted for.

b) Simple sewer flow models (SFMs)

In certain situations it may be beneficial to represent CSO performance in a sewer system without building a detailed SFM. Such situations will be characterised by relatively small catchments where there is little interaction between CSOs and where the magnitude of the continuation flow can be estimated with some confidence (e.g. where it is controlled by a pump). Reasonable estimates of spill volumes can be made using simple tank models, that do not specifically model pipe hydraulics. Instead, the flow processes are represented by a number of tanks in series and in parallel. Each tank receives foul flows and run-off from a different subcatchment. One of the principal characteristics of this type of model is the ability to simulate multiple events or long chronological rainfall sequences rapidly. The same effect could be achieved by heavily simplifying a detailed SFM.

5.3.2 Sewer quality performance under wet weather conditions

The pollutant loads carried by (and discharged from) sewer systems during wet weather vary in a complex manner. This is illustrated by the pollutographs shown in Figures 5.1 and 5.2. Many different processes are involved. These processes need to be understood qualitatively, and the most significant processes must be quantified to enable reliable predictions to be made. Equally important are the dry weather processes that contribute to the build-up of sediments on catchment surfaces and in sewers. Such deposits form an important source of pollutants that can be deposited in dry weather and subsequently mobilised by the rainfall and the resulting higher flows that occur during wet weather.

The main processes influencing the quality of flows in combined sewers are:

- foul inputs;
- build-up and wash-off of sediments from the catchment surface;
- deposition and erosion of sewer sediments;
- sediment transport in sewers;
- sediment partitioning in tanks;

- advection and dispersion of pollutants; and,
- biochemical reactions.

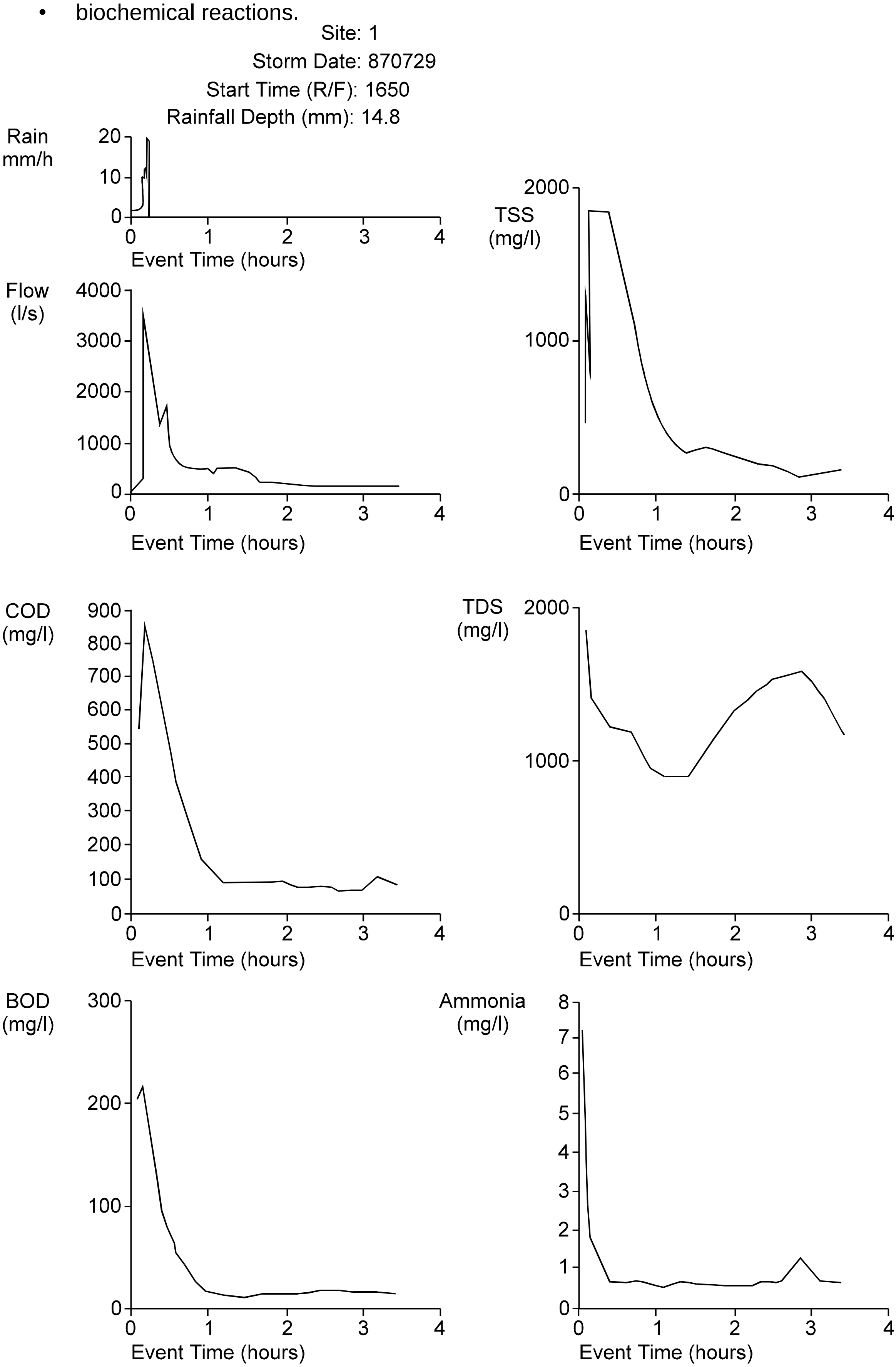


Figure 5.1 Example of observed pollutographs in sewer flow - Site 1

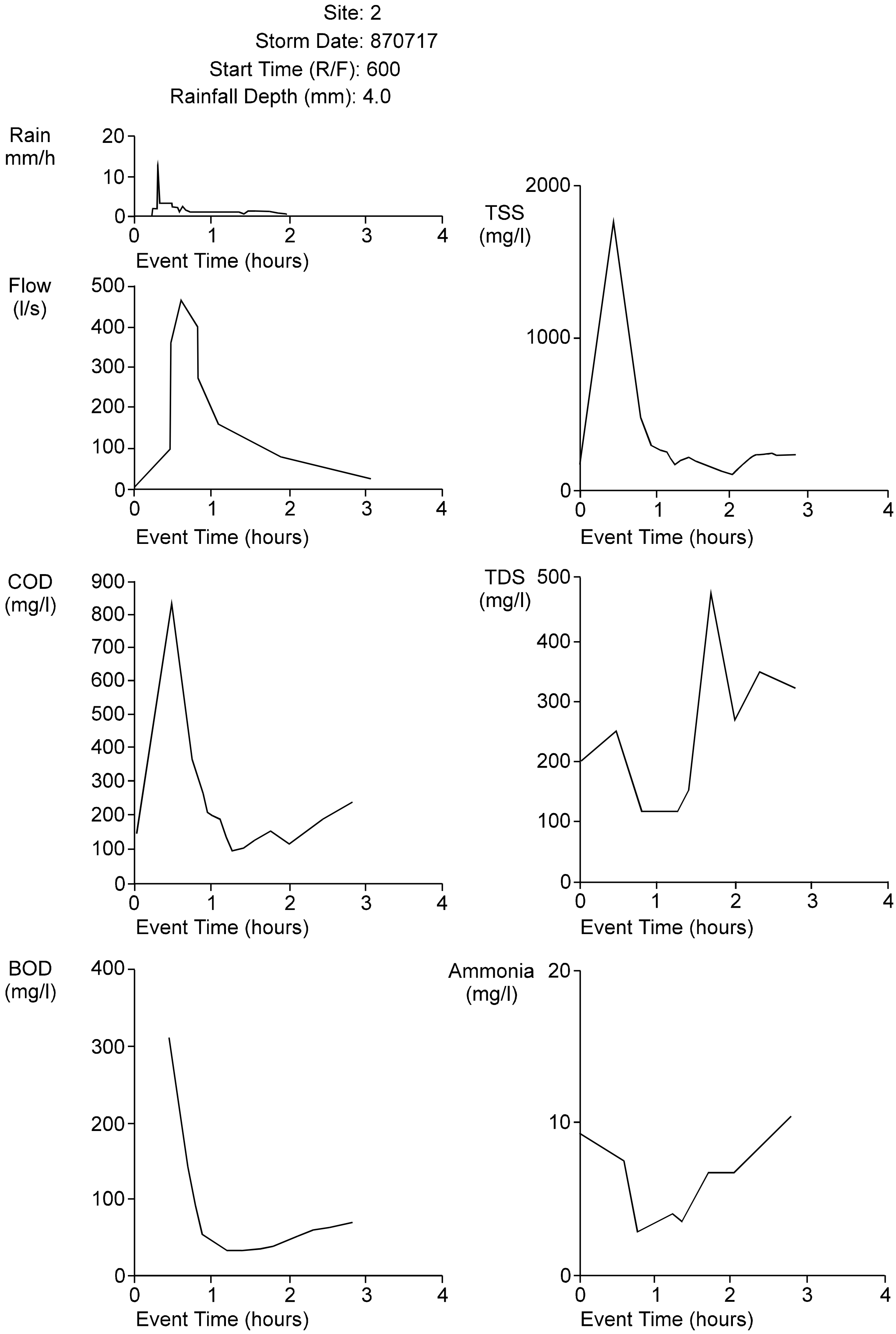


Figure 5.2 Example of observed pollutographs in sewer flow - Site 2

a) Foul inputs

Foul inputs include the domestic, commercial and industrial inputs to sewers. The flows and pollutant loads vary both spatially across a catchment and temporally (diurnal variation and variation from day to day). Typical diurnal variation is illustrated in Figure 5.3.

b) Build-up and wash-off of sediments from the catchment surface

Sediments build up on roads, roofs, pavements and in gully pots during dry weather periods. The quantity and characteristics of these sediments will depend on many factors, including the different land uses and on length of the dry weather period. Various pollutants will be associated with these sediments. These sediments and attached pollutants are washed off the catchment surfaces during storm events and enter combined sewer systems and separate storm water systems. The quantities washed off will depend on the intensity of the rainfall and the erosion capability of the overland flow, as well as the actual quantities available on the surfaces. The quantity available on the surfaces will be dependent on the length of time between rainfall events.

c) Deposition and erosion of sewer sediments

Suspended sediments tend to settle out of the sewer flow and deposit on the bed of the sewer during periods of low flow and at locations where flow velocities are low. The deposition process depends on many factors, including the size and density of the sediment particles and the flow regime in different parts of the sewer system. Coarser, denser sediments, that may be derived from catchment surfaces, tend to deposit more readily than the finer organic sediments that are derived from foul inputs. The deposited sediments can have attached pollutants (e.g. BOD) and act as a “store” of pollutants within a sewer system.

As flows increase, either as part of the normal diurnal dry weather cycle, or during storm periods, deposited sediments may be eroded. The point at which erosion begins and the subsequent erosion rate depends, among other things, on the flow velocity, the width of the sediment bed and the characteristics (in particular the shear strength) of the sediment deposits. As sediment is eroded, interstitial water and associated pollutants are released into the sewer flow. Sediments and associated pollutants may be deposited and re-suspended many times during their passage through a sewer system.

d) Sediment transport in sewers

Once sediment has been eroded and entrained in the flow, it may be transported down a sewer either in suspension or as bed load. Lighter material tends to travel in suspension and the heavier material as bed load. The mode of transport is strongly influenced by the flow velocity and the degree of turbulence, that will vary in space and time in the sewer system.

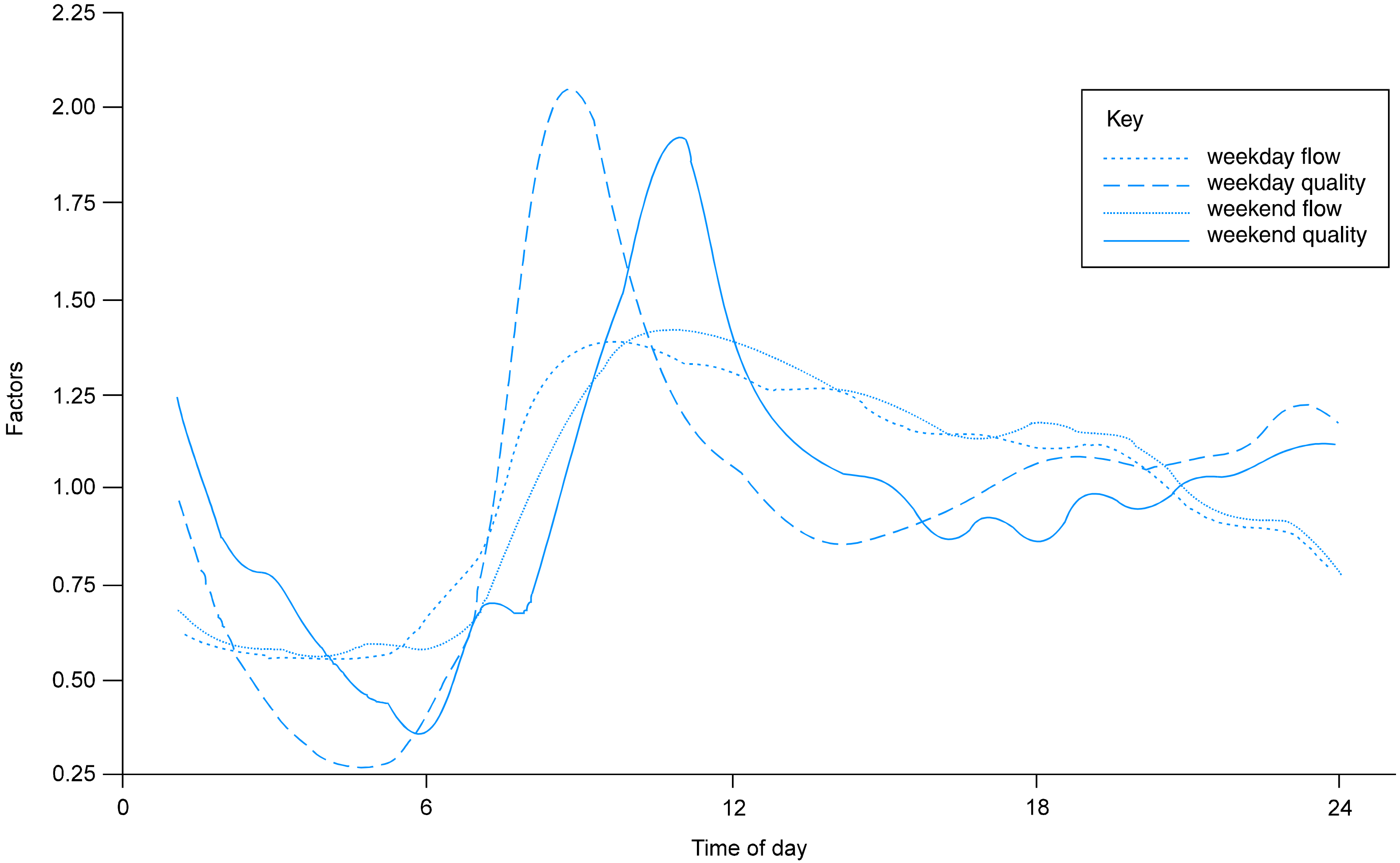


Figure 5.3 **Typical diurnal profiles for dry weather flow and quality in foul sewers**

e) Sediment partitioning in tanks

One of the effects of detention tanks in sewer systems is to reduce the local flow velocity. This in turn encourages suspended sediment to settle and, as a consequence, the concentration of suspended sediment tends to be less in the spill flow from a tank than in the tank itself. This partitioning effect helps to reduce the polluting impact of spills. The deposited sediments may eventually be re-entrained from the base of the tank and conveyed to the STW or a downstream tank. This will create a short term peak loading rate at the end of a storm when storage tanks are full and may result in a high pollutant concentration spill to the environment.

f) Advection and dispersion

Dissolved pollutants and fine suspended sediments are moved through a sewer system by the process of advection and dispersion. Advection is the movement of pollutants and suspended sediments in the same direction and at the same speed as the flow. This is by far the main pollutant transport process.

Dispersion, on the other hand, is a more random process whereby pollutants and suspended sediments move in such a way as to reduce concentration gradients along a sewer. During a storm, and to a lesser extent during dry weather flow, concentration gradients will build up due to the varying input loads and the resuspension of sediment deposits. Turbulence within the flow regime will tend to disperse the pollutants so as to reduce these concentration gradients.

g) Biochemical reactions

Finally, some pollutants can undergo biochemical reactions (for example, the decay of BOD and ammonia), during their transit through a sewer system. Hence, the load of some non-conservative pollutants can be reduced within the sewer system prior to discharge. However, this will reduce the dissolved oxygen level in the sewage that may increase the environmental impact of a spill.

5.3.3 Modelling sewer quality performance

Various approaches can be used to model the performance of sewer systems in terms of the pollutant load discharged during wet weather. These approaches can be divided into three groups:

- simple tank simulation models;
- detailed SFMs plus event mean spill concentrations; and,
- detailed dynamic SQMs.

These are discussed in more detail in the CIWEM Urban Drainage Group publication "GUIDE TO THE QUALITY MODELLING OF SEWER SYSTEMS", Version1 November 2006.

5.3.4 Determinands Modelled

The determinands that are usually modelled by SQMs are BOD, COD, ammonia, and suspended solids (metals and bacteria can also be modelled). Suspended solids are more generally referred to as sediment, and include the solids that are deposited in the sewer system. Sediment is divided into sediment fractions (normally two: fine and coarse) each characterised by particle size, density and settling velocity. Different sediment stores (e.g. road, roof, pipe) are described by the sediment types they contain. A sediment type is a particular mixture of fine and coarse sediment fractions with other characteristics, such as shear strength, wet bulk density and moisture content. Each sediment fraction in a sediment type is also given a potency factor that expresses the amount of pollutant attached to a mass or volume of deposited sediment. When the sediment moves, the pollutant attached to the sediment also moves.

The pollutants modelled can be assumed to be conservative in that no biochemical changes will occur during their transport through the system. Alternatively, more complex biochemical interactions can be modelled to represent the decay of pollutants as they are advected downstream over long distances.

b) Processes modelled

SQMs typically contain submodules to represent:

- foul inputs;
- surface washoff;
- pollutant and sediment behaviour in pipes; and,
- pollutant and sediment behaviour in tanks.

Most dynamic SQMs estimate foul inputs from domestic sources based on the different land uses assigned to subcatchment areas in the model. Diurnal variations in flow and quality are included.

The representation of surface wash-off varies between dynamic models. Generally, however, the wash-off of sediments and pollutants is a function of rainfall intensity either for unlimited amounts of sediment or for a set depth of sediment on the surface of the transport source. Gully pot processes may also be included.

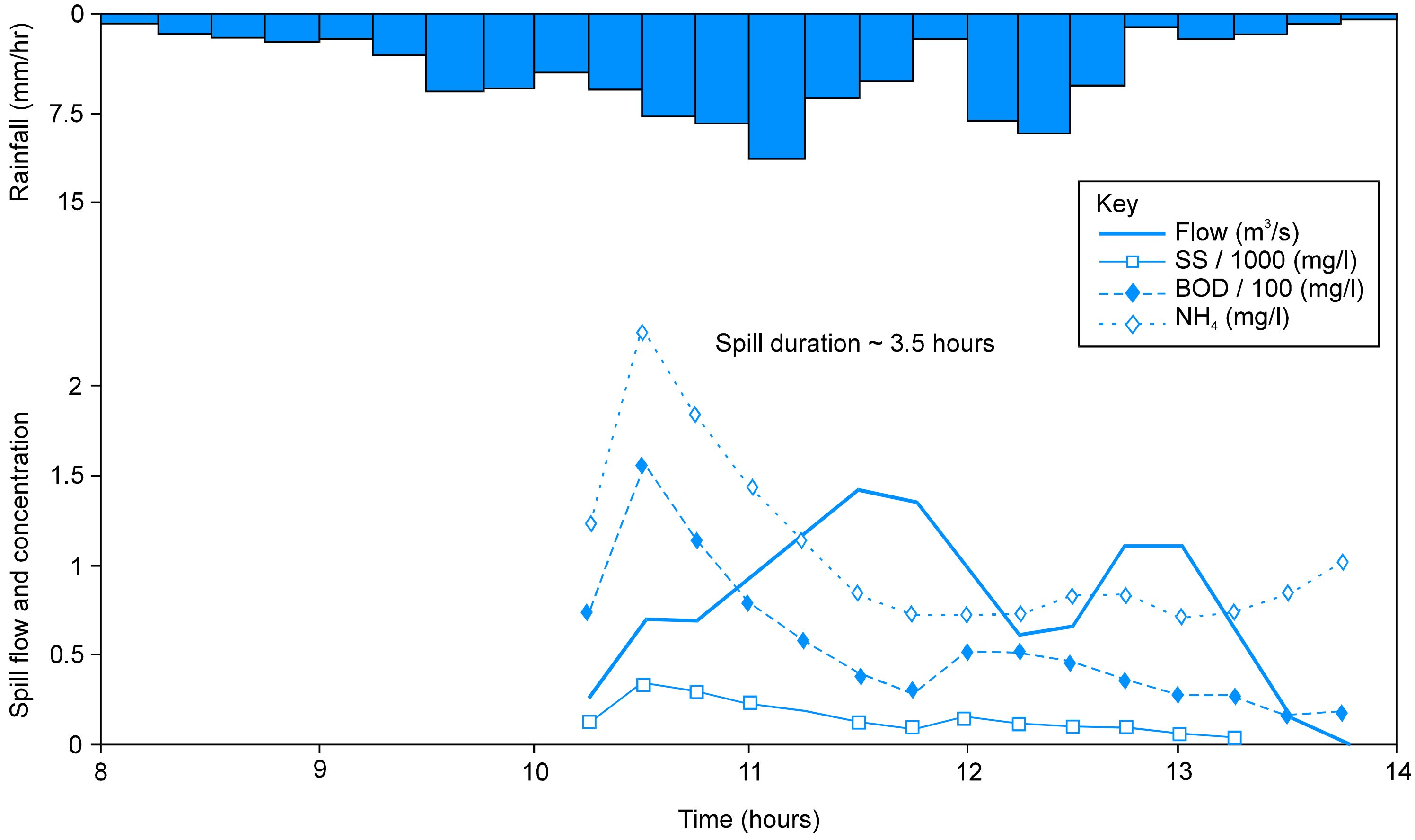


Figure 5.4 Example of spill pollutographs produced by dynamic sewer quality model - Site 1

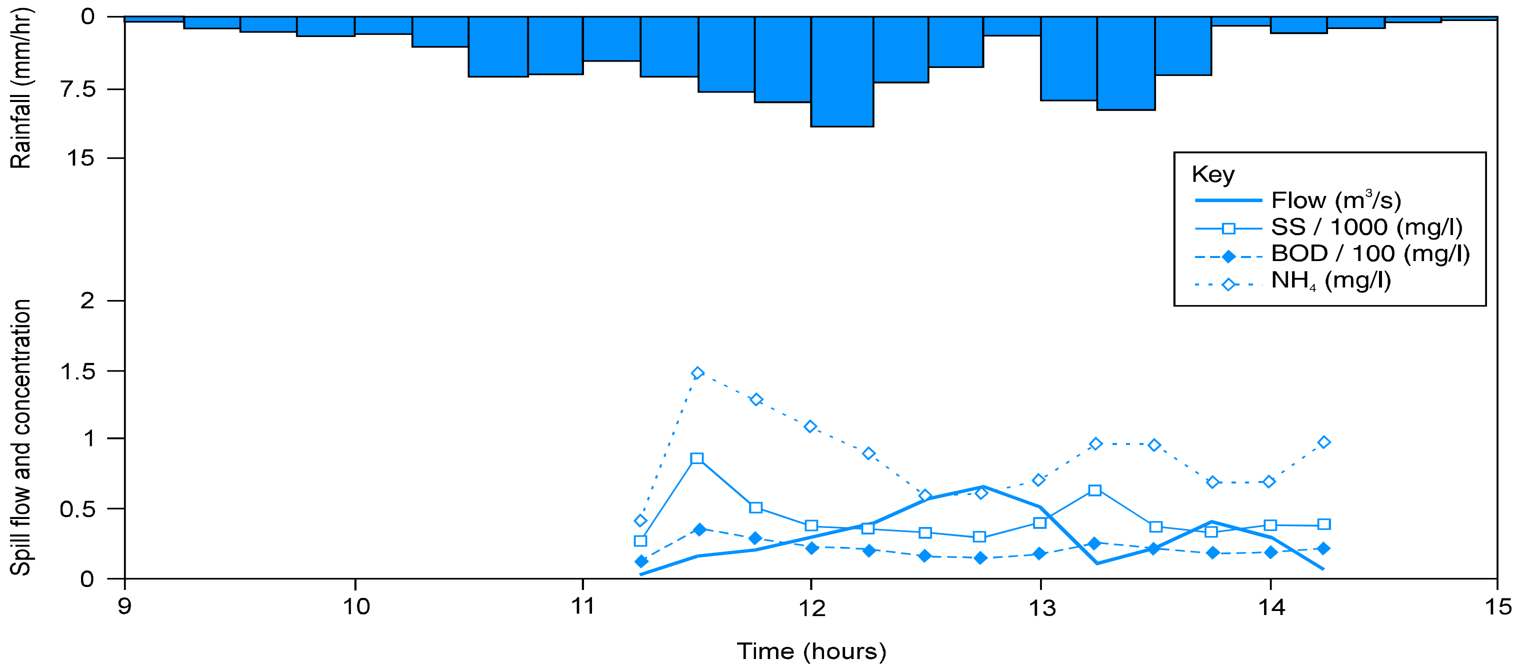


Figure 5.5 Example of spill pollutographs produced by dynamic sewer quality model - Site 2

Various formulations are available for modelling transport processes (in particular, sediment transport) in sewers. Generally, it is assumed that pollutants and suspended sediments are moved in the flow by advection. Dispersion may be ignored, as it is considered to be of little effect, although some SQMs do include the effect of dispersion. A sediment transport model is used to determine how much sediment is eroded from a defined bed of sediment and transported by advection through the system. Settling velocities for the sediment particles are used to define how much deposition takes place. Pollutants attached to the sediment are transported at the same rate as the sediment. Dissolved pollutants are advected through the system at the flow velocity rate.

Sewer tanks can be modelled as a simple mixing chamber where no deposition occurs. An efficiency factor can be used to enable different pass forward and spill concentrations from the tank to be modelled as would occur if settling had taken place. Alternatively, the process of sediment and pollutant settling can be represented in the tank to give different sediment and pollutant concentrations in the spill and pass forward flows.

5.3.5 Using SQMs

The verified SFM should be used to demonstrate the hydraulic performance of the system before starting to plan the use of the SQM. This can provide several types of information:

- the likely extent of the pollution problems;
- where the data collection for model calibration/verification is most required, particularly in relation to sediment depth for model initialisation; and,
- the likely solutions to the existing problems.

The way in which the model is initialised should be considered when applying a SQM. System initialisation involves defining the initial conditions of all pollutants and sediments within the system, both above and below ground. It also involves defining the sediment depths on the catchment surface and in the pipes. The build up of sediments in the pipes is related to the antecedent dry weather period. It is suggested that the system is initialised by running a few days of dry weather flow through the system, so that indications are made as to where sediment is likely to deposit. These indications can then be tied to any available site measurements or local information.

A suitable run time must be defined once the model is initialised. This will depend upon the use to be made of the model results. If only the period when the CSOs operate is of interest, a run time can be investigated using the SFM. If the results from the model are being passed onto a STW model, the SQM needs to be run so that the effects of the storm pass right through the system until the STW returns to dry weather flow conditions. Care must be taken to ensure that any storage facilities have been fully emptied.

As well as using the correct run time, it is necessary to optimise the timestep used in the model. Using a timestep that is too small can sometimes generate instabilities in a model, especially at structures. A small timestep will also increase the time taken to run the model such that it may become impracticable. However, using a timestep that is too large can cause the model to be inaccurate.

The start time of the storm relative to the diurnally varying dry weather flow pattern must also be considered. There are favourable times and unfavourable times in terms of pollutant discharge. The model may be used to present an average situation or a worst case situation. The storm should be timed to coincide with the highest level of dry weather flow to represent a worst case situation.

Related to this is the use of point source inputs to the system, such as, industrial discharges. These inputs may be of a continuous nature or they may be intermittent discharges to the system. The intermittent discharges may occur at set times during the day or may happen at any time. When the latter is true it is difficult to model the input of these discharges. The same approach, as described, for dry weather flow, can be used where either a worst case situation or an average case situation can be represented.

5.4 Sewage treatment works quality modelling

This section discusses how to represent sewage treatment works flow and quality in UPM studies.

This section is subdivided as follows.

5.4.1 Sewage treatment works performance under wet weather conditions

Typically, sewage works design is based on the treatment of multiples of DWF to achieve defined standards of effluent quality. The environmental regulator will specify requirements for treatment capacity in relation to flows arriving at the STW and the definition of DWF. In England and Wales the treatment standard required usually varies for flows up to a nominal 3 DWF; for flows between 3 and 6 DWF; and, for flows in excess of 6 DWF. All flows up to 3 DWF pass through the full sequence of treatment processes installed on any particular site. Beyond this there are several possibilities for how and when flows in excess of 3 DWF are dealt with. Thus, at sites where primary and secondary treatment processes are installed, flows in excess of 3 DWF (but less than 6 DWF) may be diverted to storm tanks either before or after primary treatment (usually the former). At some sites, flows in excess of 6 DWF are diverted directly to the receiving water following preliminary treatment while at others all flows above 3 DWF pass through storm tanks before discharge. At small works, storm tanks are not always installed and full treatment is provided for up to 6 DWF, even at sites where primary sedimentation is not installed.

Sewage treatment works design has traditionally been based on empirical rules that, by experience, have produced good results in the past. There is the potential for WwTW to perform differently in wet weather conditions. While well designed STWs operating within capacity (e.g. up to 3DWF) may not show a significant response to increased storm flows, it is implicitly understood that performance, as measured by percentage removal of pollutants, may deteriorate during storm conditions. Such deterioration is often of a short duration, since prolonged wet weather flow results in a dilution of the influent sewage arriving at the treatment plant. There is a risk of decreased effluent quality in storm conditions at treatment plants. Storm flows are usually associated with a ‘first flush’ of foul sewage containing high concentration of pollutants. Treatment works can be adversely affected by high flows and pollutant loads in a variety of ways. The effects are a function of the nature of the catchment, the size of the catchment, the size of the works, the treatment processes installed at the works and the effluent quality required. Some treatment processes, particularly those with long retention times, are relatively insensitive to flow and load increases, others such as ‘small footprint’ processes with short hydraulic retention times are easily upset by sudden increases in load.

The effects of storm flows on a STW can be generally sub-divided into the following categories.

- Reduction in settling performance - treatment processes that rely on physical separation of suspended solids by gravity are adversely affected by high flows.
- Reduction in biological removal efficiency - the removal of organic pollutants (measured as BOD) and specific nutrients such as ammonia, depends on the contact time between the micro-organisms and the pollutants. Removal efficiency is reduced during periods of high flow.
- Washout of solids - this is a particular problem with activated sludge processes that rely on the efficient separation of suspended biomass from treated effluent in a final settling tank. The effects are similar to those in the first category with the additional problem that excessive solids loss can result in reduction in biological treatment efficiency.
- Mechanical problems - high flows during storm periods, particularly ‘first flush’ events can inundate preliminary processes such as screens and grit removal plant with sewer debris etc. that may cause operational difficulties.

5.4.2 Modelling sewage treatment works (STW)

It may be necessary to represent STW flow and quality in the UPM process.

The first UPM studies used calibrated sewage treatment works models to derive water quality data for the STW. This was not particularly successful due to the following reasons:

- the matches to observed performance were not particularly good;
- they were expensive to build and calibrate;
- they did not give the observed variability in water quality in dry weather conditions; and,
- the variation in wet weather and dry weather performance was generally within the accuracy bounds of the modelling process.

The preferred approach is to use a log-normal distribution for water quality, based on an analysis of STW data either collected for operational purposes, or if this is insufficient, enhanced by additional specific sampling.

As previously stated there is the potential for STWs to perform differently in wet weather periods. It is therefore necessary to split the datasets between dry and wet conditions, and check that there is no significant difference in performance between the two sets. If there is, there will be a need to produce two distributions, one for wet weather and one for dry periods.

There are other mathematical models that are available to describe STW performance, that fall into several categories. These are:

- detailed mechanistic;
- reduced-order;
- statistical correlation;
- time series; and,
- improved statistical distributions.

a) Detailed mechanistic models

A considerable amount of work has been carried out in the development of mechanistic models. These are derived from theoretical equations to describe the biological and physical processes taking place. Although there are recommended default values for the calibration parameter values required, most of these models require site-specific calibration. The theoretical basis to dynamic mechanistic models makes them the most appropriate models to investigate plant operation outside the normal operating conditions.. If there is the potential for significant changes in STW performance due to changes in either operating regime or Capital Improvements it may be worthwhile to develop a detailed mechanistic model.

b) Reduced-order models

Reduced-order or simplified models are developed by simplification of mechanistic models. For example, nitrification terms are dropped where the sewage works does not nitrify, or the solids hydrolysis terms are left out where the sewage retention time is long (e.g. extended aeration). The resulting models are as valid as the full model, as long as the processes omitted are known to be insignificant. Reduced-order models have been developed with STW control as the main aim, the model outputs being used as part of a decision support system.

c) Time series

Time series models are intended for short-term application, backed up with long-term data. By keeping the time series up-to-date, predictions for the near future can be made of expected sewage flows and strengths, and expected STW performance. The more time series data there are to predict seasonal, diurnal, or other variations, the better (in theory) is the predictive capability of the model. These models have the built-in assumption that the future will reflect the past. They are most likely to fail in predicting effluent quality when there are gross changes in the way the plant is configured or is being operated, or (to a lesser extent) when there are gross changes to the influent sewage flow or quality.

d) Enhanced Statistical distributions

These models are based on the assumption that the effluent quality behaves as a random variable. The underlying random distribution has to be selected, but is commonly assumed to be based on the log-normal distribution. The choice of probability model may need to account for cross-correlation between determinands. For example, high solids are usually associated with high BOD, and high flow is also usually associated with high solids. The probability model may also need to be modified to include historical effects that account for the fact that the probability of a determinand taking on a value is not equally likely at a given time, but is affected by past values, hence low values are more likely if previous values have also been low. Enhanced statistical distribution models have not been widely used because of the difficulty of creating an appropriate probability distribution function that includes the features of known effluent quality.

It is recommended that the STW is included in the sewer hydraulic model and flow data is taken as an output from the model.

5.5 River impact modelling

This section describes the general effects of intermittent wastewater discharges on river water quality and the role of river quality impact models (RQIMs) for quantifying these effects. The stages required to make a RQIM ‘fit for purpose’ are identified.

The section is subdivided as follows:

5.5.1 Effects of intermittent discharges on river quality;

5.5.2 Types of modelling approach;

5.5.3	Description of dynamic RQIMs;
5.5.4	Building, calibrating and verifying RQIMs;
5.5.5	Using dynamic RQIMs.

5.5.1 Effects of intermittent discharges on river quality

Wet weather, resulting in intermittent discharges from CSOs, SWOs and storm tanks, plus poor quality effluents from STWs, commonly affects water quality in many urban rivers. On occasions this can result in gross pollution of large stretches of urban watercourses. The main changes in river water quality due to wet weather discharges can include:

- a reduction of DO as a result of:
  - degradation of dissolved BOD;
  - degradation of BOD attached to sediments;
  - resuspension of polluted bed sediments exerting an additional oxygen demand; and,
  - low DO levels in spilled storm sewage; and,
- a rapid increase in river concentrations of ammonia, bacteria, COD, and suspended sediments, as well as heavy metals and other toxic substances where trade effluent discharges are present in sewage.

Processes affecting DO levels are illustrated in Figures 5.6 and 5.7.

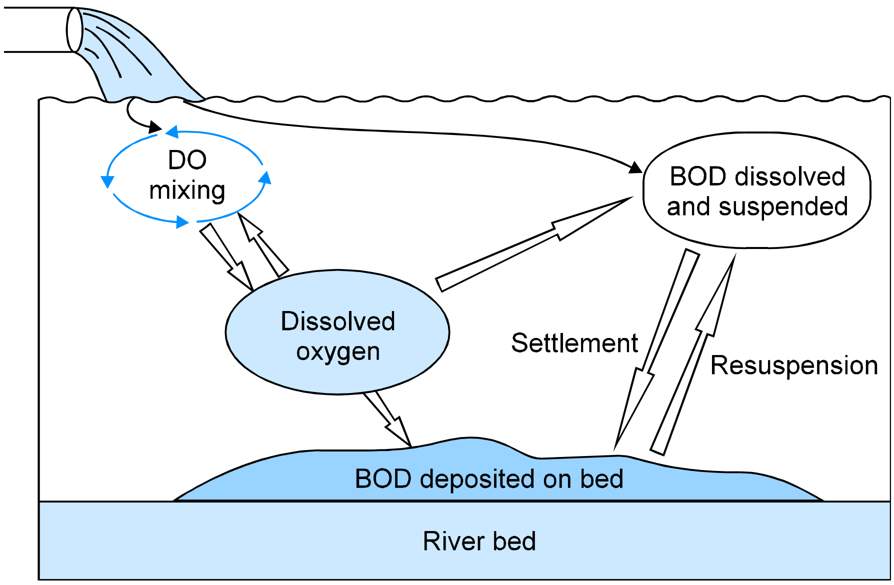


Figure 5.6 Processes by which organic loads in CSO spills can affect DO levels in rivers

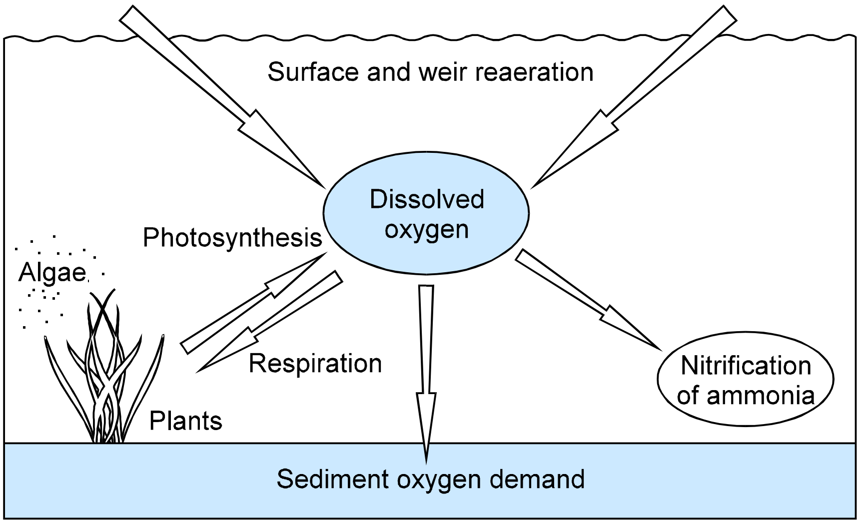


Figure 5.7 Additional processes affecting DO levels in rivers

The magnitude of the impact will vary from site to site and will be affected by river characteristics, such as:

- upstream quality and flow;
- channel slope;
- channel geometry and roughness;
- in-river structures;
- pH;
- temperature;
- ecology (macrophytes, algae, fish and invertebrates); and,
- nature of effluent discharged.

Each of these is considered below.

a) Upstream quality and flow

High DO levels and low concentrations of pollutants in upstream river flows will increase the capacity of a river to assimilate wet weather discharges. Upstream quality will vary due to:

- natural diurnal and seasonal variations of flow, DO and temperature;
- channel geometry; and,
- the impact of discharges and abstractions.

b) Channel slope

In general, steeper channels create more turbulence, thereby increasing the rate of oxygen transfer (re-aeration) across the air/water interface. This leads to higher re-aeration rates and may, depending on the rate of decay of any BOD, result in increased levels of DO in the river.

Sediments and any BOD attached to these sediments will tend to settle out on the bed in rivers with flat sections. This will affect the severity and location of the DO sag, as illustrated in Figures 5.8 and 5.9.

c) Channel geometry and roughness

The channel cross-section and roughness affect the water depth. In general terms, a shallower water depth will induce more turbulence by increasing the effect of bottom roughness and, therefore, allowing greater opportunity for re-aeration to occur.

d) Structures

Structures (weirs, bridges, culverts etc.) will generally reduce flow velocities and consequently increase water depth directly upstream. The net effect is to reduce the amount of re-aeration (by increasing depth and reducing the water surface gradient) occurring immediately upstream of the structure, that may create critical quality conditions. Conversely, velocities are increased downstream, that will increase re-aeration. Re-aeration at weirs can cause a significant downstream increase in DO levels.

e) pH

The pH of a river is critical in determining both the amounts of un-ionised ammonia present and hence, the toxicity to fish. Higher pH levels will increase the proportion of un-ionised ammonia for a given concentration of total ammonia. However, the toxicity of un-ionised ammonia is reduced at high pH levels. Further information is presented in Appendix A.

f) Temperature

Higher temperatures will also increase the levels of un-ionised ammonia for a given concentration of total ammonia. In addition, higher river temperatures will result in lower DO saturation concentrations. Also, BOD decay processes will proceed more rapidly at higher temperatures.

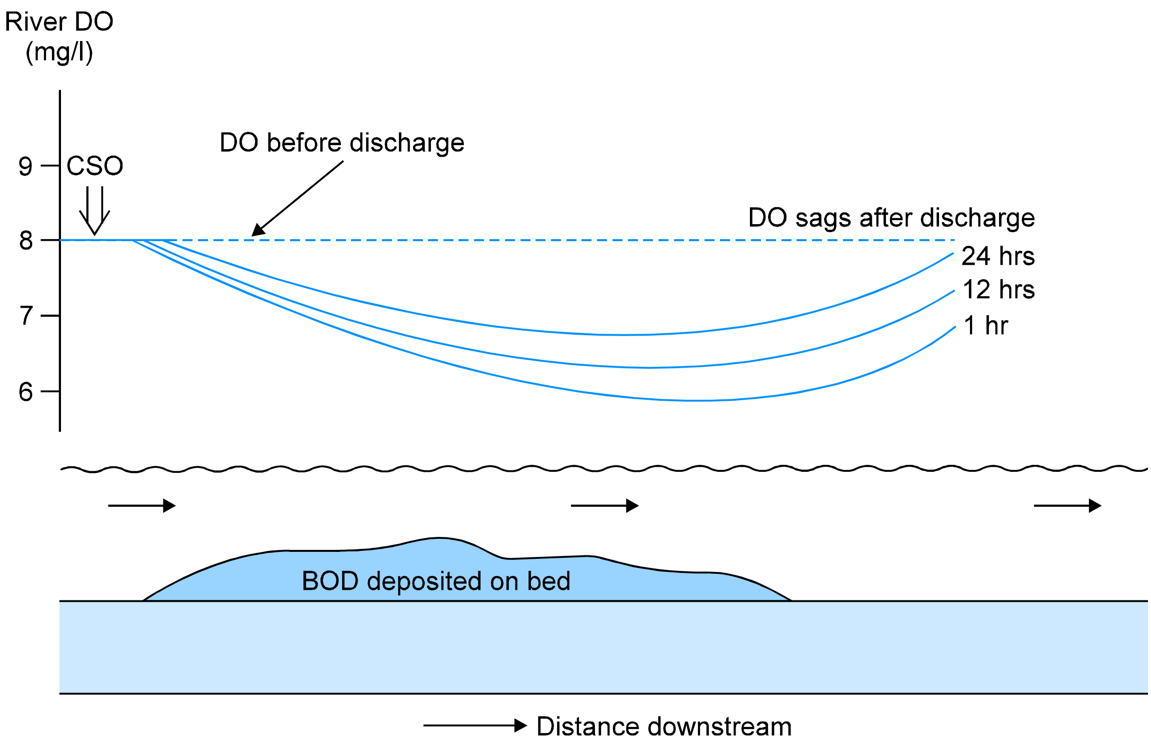


Figure 5.8 Potential DO sag after a CSO discharge where most of the BOD settles to the river bed

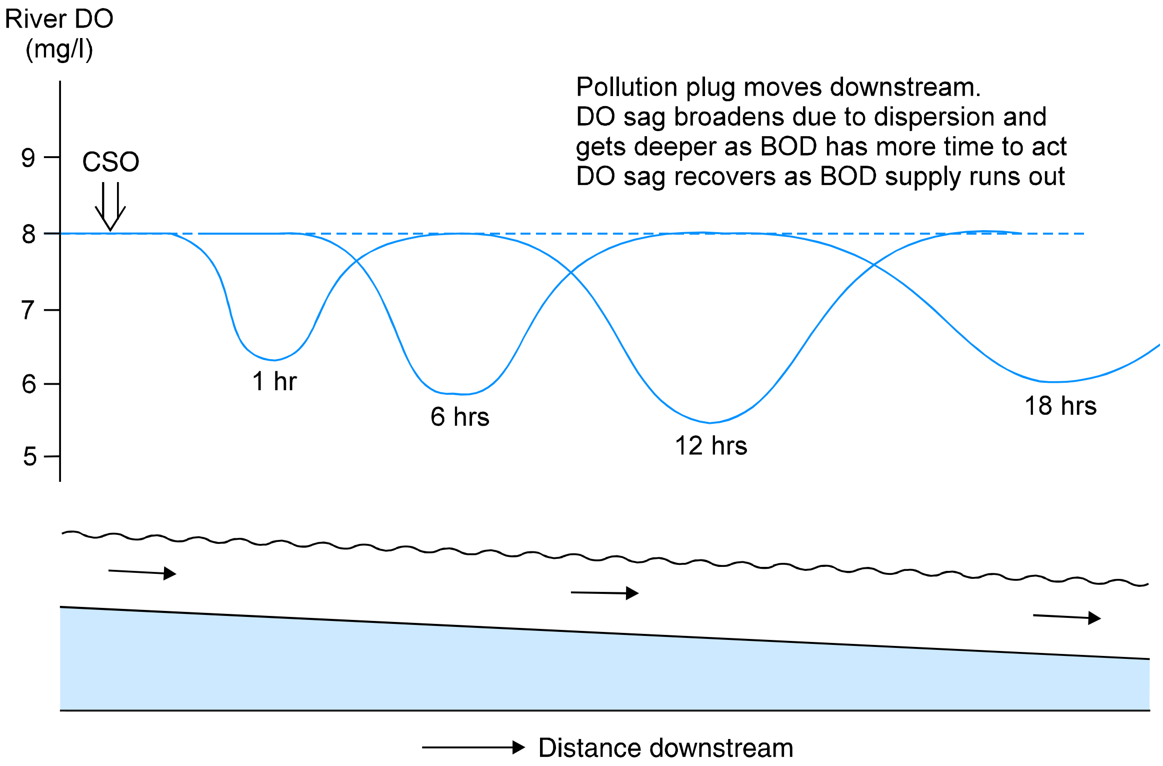


Figure 5.9 Potential DO sag after a CSO discharge where most of the BOD remains in suspension

g) Ecology

Aquatic vegetation affects DO levels in water by the action of two processes. Photosynthesis adds oxygen to the water column during daylight hours. During the night, as oxygen production declines, plant respiration depletes oxygen levels in the water resulting in diurnal variation in DO levels that may become extreme in eutrophic conditions.

h) Nature of effluent discharged

The nature of the wastewater discharge is also important.

- BOD decay rates can differ widely depending upon the nature and origin of the pollutants present causing the BOD and the degree of treatment given to the discharge. A wastewater with a higher BOD decay rate will remove oxygen from the river more rapidly than the same load with lower BOD decay rates and result in larger DO sags for the same BOD load discharged.
- Spills with a high sediment BOD load attached will produce a different impact on DO levels in the river, compared to those spills that contain predominantly dissolved BOD, due to sediment deposition and erosion, (Figures 4.18 and 4.19).
- DO levels in the discharge affect the oxygen levels in the river at the point of initial mixing. DO in spills is often higher in steeper sewer catchments. re-aeration at aprons of spill, and other structures can result in short-term increased DO levels in the river after the initial mixing.

Figure 5.10 illustrates the typical impact of a CSO discharge on a small receiving watercourse. It shows that the impact on DO is severe: DO levels fall to almost 1 mg/l. Initially, during the spill event, re-aeration is greater than oxygen consumption. Once the spill ceases, oxygen consumption from the breakdown of deposited organic material dominates below the CSO.

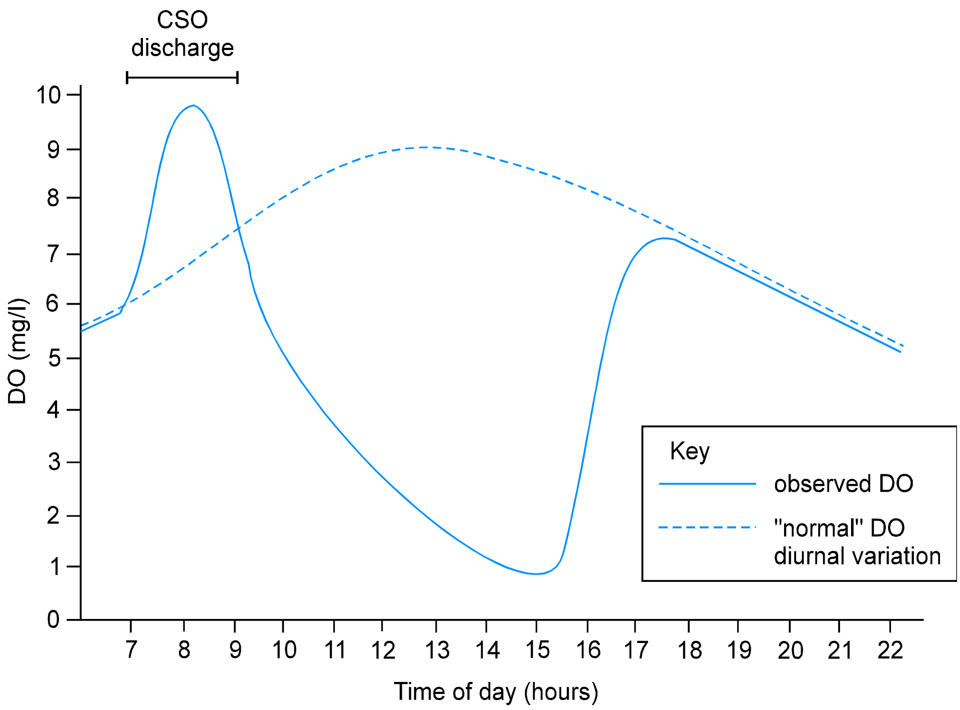


Figure 5.10 Example of measured DO concentrations in a river just downstream of a CSO following a storm discharge

5.5.2 Types of modelling approach

a) Generic types

There is a wide range of RQIMs available to assess the impact of wet weather discharges on receiving waters. They range from simple equations to complex, stochastic or deterministic, steady state or dynamic models. This section describes the types of model that are available to demonstrate compliance with long term (percentile standards) and wet weather (Fundamental Intermittent standards) water quality criteria, as described in Section 2.3, to support a UPM study. The choice is largely determined by the type of problem to be analysed and the accuracy required. The best guidance is to select the simplest model to address the problem under consideration.

Two generic types are considered.

- **Stochastic models**
  In general, stochastic models assume steady state hydraulic conditions and aim to represent the behaviour of river quality and flow over a long term period. Stochastic models are designed to be used to estimate percentile criteria to assess the impact on long term water quality expressed as, for example, 90, 95 or 99 percentiles.

- **Deterministic models**
  Deterministic models are designed to be used to estimate the concentration and duration of water quality resulting from specific event based inputs of flow and quality to the modelled river. Such models can have a steady state or dynamic representation of river and discharge flow and can be used to provide predictions for comparison with, for example, the Fundamental Intermittent standards. Event based results can also be converted into statistical criteria for comparison with percentile standards.

  Both model types may be used with differing levels of complexity in representing the effects of in-river chemical and biochemical processes to predict the impact of intermittent and continuous discharges on receiving water quality. Three levels of model complexity can be considered.

- **Simple mass balance**
  This involves mixing the predicted wastewater discharge with an appropriate quantity of river water to give an estimate of the resulting downstream river quality. Mass balance analysis does not take into account any in-river processes or dispersion of the input. As such it is most useful for determinands such as ammonia where the worst impact is most likely to be at the point of initial mixing.

- **Mass balance including simple BOD/DO processes**
  This method allows both the BOD and DO levels to be calculated through time (and thus at distances downstream) from the point of mixing of the river and effluent flows. This involves the use of equations to calculate the DO balance resulting from the decay of BOD and surface re-aeration within the river, using typical BOD decay rates and re-aeration rates. As these equations are relatively simple, it is possible to derive analytical solutions or to employ numerical procedures on a spreadsheet to arrive at a solution. However, many simplifying assumptions have to be made, including:

  - an assumption that the flow is steady and the channel is uniform;
  - ignoring one of the main oxygen demand processes - either that in the water column or at the bed; and,
  - ignoring nitrogen transformations.

- **Water quality process interactions**
  This level of complexity may be required when in-river processes exert a significant influence on the impact of an intermittent discharge following initial mixing. These processes can include pollutant dispersion, as well as advection; biochemical processing involving BOD and nitrogen transformation; photosynthesis and respiration; and, sediment attached pollutant interactions. The choice of process interactions will depend on the nature of the problems being considered and the characteristics of the river.

  UPM studies are likely to require the application of the following river impact modelling approaches. The specific type of model and degree of complexity required will reflect the requirements of the environmental regulator, in terms of the type of results and the degree of confidence required. Simplified, integrated modelling tools, discussed in Section 4.8 represent a combination of some aspects of these two approaches, with different levels of complexity in representing water quality processes as required for a particular application.

b) Steady state, stochastic models

Models based on the mass balance approach are only valid if the data used in the mass balance equation relate to the same instantaneous period of time. If the instantaneous values are replaced with distributions of flow and quality observed above a discharge and for the discharge itself, the mass balance equation can be repeated many times with different values taken from the specified distributions. This is known as the Monte Carlo Simulation, that makes it possible to apply correlation between the determinands and specify more representative distribution shapes. The Monte Carlo Simulation method can be applied to a single discharge or on a catchment wide basis.

Typically, Monte Carlo Simulation models are calibrated and verified using data for river and effluent flow and quality collected as part of routine environmental monitoring programmes to assess river quality and effluent compliance against national and EC criteria. Several years of this type of infrequent data can be combined to produce model input distribution statistics. However, these data may not fully represent the effects of infrequent intermittent discharges, particularly from CSOs. Wastewater system simulation models (Sections 5.3 and 5.4) can be used to generate distribution statistics to represent future wastewater system performance scenarios. Upstream boundary conditions for modelling scenarios can be estimated from observed flow and quality distributions that can be amended to reflect the environmental criteria specified for the scenario; for example, improved upstream quality or reduced upstream flow.

c) Deterministic, dynamic river quality impact models (RQIMs)

Complex, dynamic, deterministic RQIMs explicitly simulate the varying flow and quality in a river in response to storm events. These models can be used to produce results for comparison with river water quality standards that are concentration, duration and frequency based (for example, the Fundamental Intermittent standards presented in Tables 2.2 and 2.3). Numerical techniques are employed to solve the differential equations describing the hydrodynamic and water quality processes. Dynamic RQIMs are considered in detail in Sections 4.5.3 to 4.5.5. Dynamic models will include, to varying degrees of sophistication, the following main processes.

- **Hydrodynamics**
  Routing of variable flow through river channels with geometry varying over distance.

- **Pollutant routing**
  Advection, dispersion and mixing of pollutants within the water body.

- **Biochemical processes**
  Significant biochemical degradation processes affecting determinands such as BOD and ammonia.

- **Sediment interactions**
  Settlement, resuspension and transport of river and sewer-derived sediments, plus storage and release of associated pollutants.

5.5.3 Description of dynamic RQIMs

a) General description

Several dynamic RQIMs are currently available, all of that have the ability to model a wide range of varying flow conditions and polluting impacts in complex channel networks. Most of these models are one dimensional; that is, flows and concentrations are assumed to be uniform both laterally and vertically within the water column. Two and three dimensional models are available, but these tend to be used in more specialised cases to model estuaries/tidal rivers and stratified rivers and lakes. This section discusses one-dimensional models only. RQIMs generally have more detailed process representations than current SQMs.

Complex dynamic modelling packages are often expensive and have specific hardware requirements. Additionally, users need to be trained in using and applying a particular model. The models also require an extensive amount of data for model building, calibration and verification for wet weather impact assessment. However, the verified dynamic model solutions are the most reliable method for making predictions for a wide range of potential environmental scenarios. The best guidance is to select the simplest model to address the problem under consideration.

Dynamic RQIMs typically have three main modules. These are:

- **Hydrodynamic module**
  The results from a hydrodynamic module must be available before any advection dispersion or water quality simulations can be started. RQIMs typically use an implicit finite difference scheme using the Saint Venant equations to calculate varying flow conditions. These equations can be solved in a number of ways, depending on the accuracy and complexity of the model.

- **Advection dispersion module**
  The advection dispersion module is usually based on the one dimensional equation for conservation of mass of a dissolved or suspended substance. An implicit finite difference scheme can be used to solve a Fickian advection dispersion equation. Sediment transport calculations can also be conducted within the advection dispersion model.

- **Water quality module**
  All the water quality calculations are based on empirical equations to calculate the chemical and biochemical process iterations at each time and distance step. The water quality module is normally run simultaneously with the advection-dispersion module.

  Details of the specific biochemical and sediment transport processes included in individual models should be found in the relevant software documentation. Determinands that can be simulated include:

- DO;
- temperature;
- ammonia;
- nitrate;
- BOD;
- conservative substances;
- sediments and BOD attached to sediments;
- bacteria;
- nutrients;
- chlorophyll-a; and,
- toxic pollutants.

Modelling erosion, transport and deposition of non-cohesive and cohesive sediments with attached BOD allows the correct simulation of the delayed oxygen demand exerted by polluted bed sediments. This is particularly important when modelling the impact of intermittent discharges from CSOs.

b) Results output and interfacing with other models

The results from a dynamic RQIM should be easily presented and interpreted with regard to appropriate environmental criteria. Results processing should allow determinand concentrations to be compared with the appropriate duration/threshold criteria. In addition, models should be able to interface with appropriate sewer and sewage treatment works flow and quality models to support the concept of an integrated modelling approach.

5.5.4 Building, calibrating and verifying RQIMs

Considerable effort is involved in the data collection, building and calibration stages of model application. However, once completed, the model can be used as a predictive tool provided that it has been verified over a range of appropriate conditions and is 'fit for purpose'. The following subsections provide a brief review of the type of work that is typically involved.

a) Specifying the modelling requirements

Once the decision has been taken to use a model to address the problem of wet weather discharges, the user has to be sure that the modelling software selected can fully address the problem. At this point, some appraisal of the modelling requirements is needed. This includes defining the approximate number of boundary conditions, inputs, structures, etc. This should help in identifying any potential modelling problems and help decide modelling priorities, boundaries and limitations that may, subsequently, affect the data requirements.

The modelling requirements will also identify the complexity of model needed; whether to use a simple BOD/DO model or more complex water quality simulation including sediment transport. Increasing model complexity increases both the data requirements and model run times, but potentially increases the accuracy of the model.

b) Identifying data requirements

Calibration and verification involves the collection of substantial amounts of data. It is, therefore, vital that the data collection programme is targeted to obtain the most appropriate and reliable data in a cost effective manner. The model builder should play a major role in defining the needs of a data collection programme. A site visit by the modeller prior to the data collection programme should identify any sites that may require additional data and identify any possible modelling problems.

Data requirements can be divided into three types:

- **Input data**
  The data needed to build a model include cross-sections, general channel geometry, roughness estimates, details of bridges, culverts and other in-river structures and any tributaries. Careful specification of location and definition of cross-sections in the initial stages is important to produce a realistic and stable model; considerable problems can occur during model building if this is not done.

- **Calibration data**
  These are either available from historical records or are collected during short term field surveys under dry weather conditions and storm events. The data will include the following:

  - flow and quality for all significant discharges to the modelled river;
  - flow and quality at selected sites along the river. The minimum requirement is three sites; one upstream of all major urban inputs, one as close as possible to where the worst impact is anticipated and one located well downstream of the main impact at a location where the river is recovering;
  - flow measurements at the upstream and downstream boundary for different river flows, that may require a longer period than that covered by the short term survey;
  - time of travel data at different flows; and,
  - dispersion data, at different flows.

  The data must be robust; that is, they should adequately represent the conditions under that the model will be applied; for example, summer low flows.

- **Verification data**
  These data will be similar to the calibration data but are used to evaluate the performance of the calibrated model, under dry and storm conditions.

  In practice, the data collected during a field survey may in some circumstances be a compromise between that that is desirable to allow optimal calibration and verification of the model under the conditions for that it is to be based, and that that is feasible given the time and resource constraints of the data collection programme. When the available data are not fully comprehensive, the idealised model calibration and verification processes described in 5.5.4 (d) and (e) may need to be pragmatically modified to recognise the shortcomings of the dataset. This may dictate that separate calibration and verification exercises are inappropriate and that a model that is “fit-for-purpose” is produced via an integrated calibration and verification process using all of the relevant data.

c) Model building

The first step in model building is conceptualisation, in that the catchment is defined and river channels and junctions are identified. Decisions are taken about what to include within the model and where to draw the boundaries. The upstream boundary of the main river channel should be above all major urban inputs. Tributaries that are largely urban streams can be modelled as inputs to the main channel with loads and flows calculated by the sewer models (see Section 5.3). This is a simpler representation than explicitly including the tributaries within the RQIM, and will improve the model run times. The downstream boundary of the main river channel should be far enough downstream to be sure that the worst impact area is included within the model. Sometimes this downstream boundary will be obvious, for example, the confluence with a major river that provides ample dilution, or a major weir that ensures replenishment of any oxygen deficit. On a flat sluggish river the worst impact area is likely to be quite close to the main discharge point. On a more fast flowing river the worst impact may not be felt until the river flattens out some distance downstream of the discharge point, possibly behind a weir. These factors and any local knowledge, about water quality problems during wet weather periods, should be used to determine the downstream boundary of the model.

The individual channel and reach details are then defined together with details of any structures such as culverts and bridges. Finally, the boundary conditions are specified as described in Section 5.5.5. The model should be checked for stability, to ensure that the simulated flows and quality results produced are sensible for a range of conditions.

d) Calibration

Calibration is the process of adjusting the various process rates and parameters within the hydrodynamic and water quality models, in order to match model predictions with observed data. This process should be completed for both dry weather and storm events. Both hydrodynamic and quality modules will require calibration. There are three basic stages in calibrating a model.

- **Stage 1 - Time of travel**
  The first stage of calibration compares the predicted time of travel with survey data. Adjustments to the channel roughness coefficients will allow the modelled time of travel to broadly match the observed data. This process may be completed over a range of river flows. Roughness values also affect water depth so that, where available, water level data should also be used to check the model's performance.

- **Stage 2 - Advection dispersion**
  Stage two involves checking the dispersion characteristics in the advection dispersion module. The shape of dye tracer “clouds” measured in the survey should be reasonably predicted by the model. This is usually an iterative process of adjusting dispersion coefficients and factors until a reasonable match is achieved at a range of flows.

- **Stage 3 - Water quality**
  In stage three, water quality predictions are compared with observed water quality time series or spot sample data. Calibration can proceed by adjusting the various water quality process rates until a good fit with the observed data is achieved. It is best to calibrate temperature first, followed by BOD, ammonia and nitrate and finally DO. It is important that the process rates are not altered beyond realistic bounds. In general, process rates that are known (from laboratory studies) should be adjusted first. Next the processes with the most physical meaning should be adjusted before adjusting other process rates and values.

  Difficulties in obtaining a good calibration may result from unknown inputs or from un-modelled processes (e.g. if algal growth has not been represented). In this case, an alternative modelling approach may have to be adopted, although this should have been identified in the initial assessment of the problem.

  The acceptability of water quality calibration is very subjective. Calibration objectives are easier to specify for hydraulics models, where volumes, depths and velocities can be used and matched to within set percentages or calibration criteria, i.e. 5, 10 or 20% of the total. However, it is not as easy to do this with quality data, due to its variability. Often, calibration by eye is as good as any statistical test. But there are some factors that must be considered. For example, how good is the peak to peak match for pollutant plug or DO sag? This will give an indication of the load balance. Is the timing of the pollutant plug or DO sag matched adequately? This will confirm that the appropriate river channel roughness is being used.

e) Verification

Verification refers to the process of confirming the reliability or accuracy of the model, for both dry weather and storm events, by comparing model predictions with observed data from events that were not used for calibration, to ensure that the model is ‘fit for purpose’ for the likely range of simulation conditions.

Trying to match the predicted and observed flow and quality data is often very difficult. Numerous sources of errors can occur that could affect the outcome. In an attempt to rationalise these errors, the model builder should try to identify mismatches between observed and predicted data and categorise them as resulting from:

- inappropriate process rates and other model parameters;
- processes not included in the modelling capabilities of the software; or,
- mismatches caused by unmodelled polluting inputs or abstractions.

Inappropriate process rates may be rectified by further justifiable adjustment of the relevant rate coefficients, i.e. recalibration. If this is necessary the model will not be strictly `verified' unless additional independent data are used.

It may be possible to allow for processes not included in the model either by adjusting process rates beyond expected ranges, that will limit the model's usefulness, or by simply allowing for the error in any results, that is normally the more appropriate approach.

In the event of mismatches caused by unknown inputs. Such discharges may be known to occur but are unknown in terms of both flow and quality, or the mismatch between observed and predicted data may be so significant that it indicates that unmodelled discharges are occurring. For either case, the model should not be made to fit the data and the mismatch should be explained in any accompanying documentation. An attempt should be made to quantify the unknown input or abstraction, that may necessitate further data collection, to increase confidence in subsequent decisions based on model results.

Verification is completed when the model has been tested against a minimum of one dry weather ‘event' and one storm event. A satisfactory match between observed and predicted river quality should be achieved. If this is not possible, but adequate explanation can be given for the mismatch, a model can be said to be verified within the limits specified.

5.5.5 Using dynamic RQIMs

Once a model has been built, calibrated and verified it can be used to investigate any number of scenarios. However, the model user should always exercise care and be satisfied with the model’s applicability. This will help in identifying problems in the model and assessing its reliability. The user should ensure that, as a minimum:

- conditions against that the model was verified have not changed; i.e., structures, process rates, etc.;
- all modelling assumptions and accepted margins of error are made known to the decision maker; and,
- process rates have not been altered beyond acceptable margins.

The model can only be fully applied in a predictive mode once the user is satisfied with the basis of the model and the associated data.

a) Boundary conditions for upstream flow and quality

It is important to be able to assign appropriate upstream river and tributary flow and quality conditions to individual model runs, as the boundary conditions will determine the dilution capacity of the river; affect the re-aeration rates; and, at high flows, can increase pollutant load by resuspending sediments.

The river flow and quality conditions at the time of a storm will depend on many factors. These include:

- the nature of the river catchment, its site, land use and geology;
- upstream discharges and abstractions;
- the time of year; and,
- the rainfall pattern over the previous days; or, for catchments that are predominantly groundwater fed, the rainfall pattern over the previous months.

Many of these factors are difficult to quantify without detailed river catchment modelling. As a result, traditional planning approaches have tended to take a conservative assumption that upstream flows will be low at the time of a storm event. A more rational method, recommended for UPM studies, is to use a statistical approach to select upstream flow and quality conditions. This can be further refined depending on the degree of complexity of the river impact problems and processes being addressed and the type of impact modelling being carried out.

Boundary conditions define the hydraulic and water quality volumes and concentrations at the boundaries of the model. These require a time series of data for the duration of the simulation. The initial conditions are those specified for all points within the model at the first timestep. The boundary time series for flow and quality can take the form of either steady state (constant) inputs or a dynamic (changing) input. For most cases, the upstream boundary conditions can be assumed to remain constant over an event. Catchments with a rapid upstream response to events or significant upstream diurnal variations in quality and possibly flow, will require the use of dynamic boundary conditions to initialise the model prior to the input of wet weather discharges. The initial conditions for running a particular event or scenario can be selected by using a Monte Carlo Simulation approach to select values from upstream flow and quality distributions, that can take account of correlations between rainfall, flow and river quality, as described in Section 4.5.2. Multiple simulations using different input values can be used to test for sensitivity to upstream conditions.

The following sections describe how suitable river flow and quality time series can be derived to represent event boundary conditions.

- **Upstream river flow**
  Two approaches are available to identify upstream flow conditions.

  – Flow duration curves

  Flow duration curves can be used to identify steady state river flow conditions for an individual event. River flow frequency distributions are normally expressed by flow duration curves that give the daily mean flows that are exceeded for different proportions of time. In some instances, the environmental regulator may be able to provide suitable data. However, there will be situations where the available data are inadequate and it will be necessary to estimate the flow duration curve by either using regionalised statistical models (Gustard *et al*, 1992) or measured flows over a suitable period of record. If the latter approach is adopted, it is common practice to infer a long period flow duration curve by relating the measured flows to those at a long term gauging station. In both approaches it is important to account for the potential impacts of artificial influences upon the flow regime. Methods for achieving this are described by Watts *et al* (1995) (see UPM2 References, Section 1.5.2).

Normally, summer flow conditions will be the most critical for intermittent pollution events. This is because flows are low (providing less dilution) and temperatures are high (that reduces DO levels, increases BOD decay rates and increases un-ionised ammonia). Thus, it is important that the river flow duration curves developed at this stage are representative of summer conditions.

In many cases it will be possible to characterise a river flow by a shifted log-normal distribution and to define that distribution by summary statistics; for example, the mean, the minimum standard deviation and the minimum flow, based on historical observations.

– Hydrological modelling

Hydrological modelling allows a long dynamic river flow time series to be matched to a rainfall time series. This provides an excellent basis for continuous simulation of the impacts of wet weather discharges over a long period of time. Alternatively, the information provided by the river flow time series can be analysed statistically to estimate the distribution of river flows at the time of summer storms and to examine any correlations between storms and river flow. This information should provide a better basis for river flow estimation for use with dynamic event modelling than the flow duration curves described for steady-state event modelling.

Conceptual rainfall-run-off models can create a time series of river flows based on a long rainfall time series and evaporation data. These models use a simple representation of the main physical processes that govern water flow in a river catchment. These processes include interception, evapotranspiration, transfers between soil, groundwater and channel storages and time of travel. Some existing concurrent rainfall and river flow data are essential to calibrate the models before they can be used in a predictive mode. Such data will not always be available and this will limit the applicability of this approach. In addition, calibration is not straightforward and should only be applied by specialist staff.

Once such a model is calibrated it can be used to generate a realistic dynamic event time series of river flows using rainfall data. Such river flows should properly reflect the antecedent rainfall conditions that precede any particular storm being investigated in the urban catchment.

– Modelling rapid flow response

There will be situations where the upstream river catchment, or a significant part of it, is known to respond rapidly to a rainfall event because a large proportion of the upstream catchment is urbanised. As a result, river flows increase rapidly and can provide extra dilution at the time of storm sewage discharges.

These situations are best handled by including the part of the river catchment that is responsible for the rapid response within either the sewer flow model or the river impact model itself. In particular, it is most important to include any urban surface water catchments that will have rapid run-off characteristics within the sewer flow model.

It should be noted that most hydrological models will only estimate the natural response of the river catchment to rainfall. Hence, any significant upstream abstraction, discharge or reservoir control may need to be accounted for by using an additional method or by assuming flow variations based on local observations.

- **River quality**

  Two choices are possible to identify upstream quality conditions. The upstream boundary quality can remain unchanged during an event (steady state) or quality can change during an event due to upstream catchment responses and/or diurnal variation. Existing upstream quality data can be used to estimate statistical distributions to be sampled by Monte Carlo simulation to identify the initial conditions for a specific modelling scenario. As with river flows, it will be normally more important to concentrate on summer data. Accuracy in estimating upstream quality conditions is usually less significant than in estimating river flows or effluent flow and quality. This is because upstream quality is generally much better than the thresholds used to assess the significance of river impacts (Section 2.3). Guidance on selection of boundary conditions where rivers are polluted and fail to achieve these thresholds above the study area can be found in Section 2.3.

The selected initial quality input conditions can remain constant over the event simulation or can be varied to reflect observed or assumed variations in the course of an event.

b) Initialisation

All RQIMs should be initialised correctly before use. Initialisation ensures that the correct flow and quality values are used at all points in the model and not just at the boundaries. For example, the correct diurnal variation of DO is only established when the model has been run long enough for data describing the quality at the boundaries to reach all points in the model. Figure 5.11 shows a typical observed summer DO timeseries produced by the diurnal effects of photosynthesis and respiration in a river. Generally model results for DO should replicate the observed diurnal pattern before attempting to introduce the effects of intermittent discharges. However, as noted in Section 2.3.2 (d), it may be appropriate in certain circumstances (for example, eutrophic rivers) to remove the effect of diurnal DO fluctuations from modelling runs to assess compliance with environmental standards.

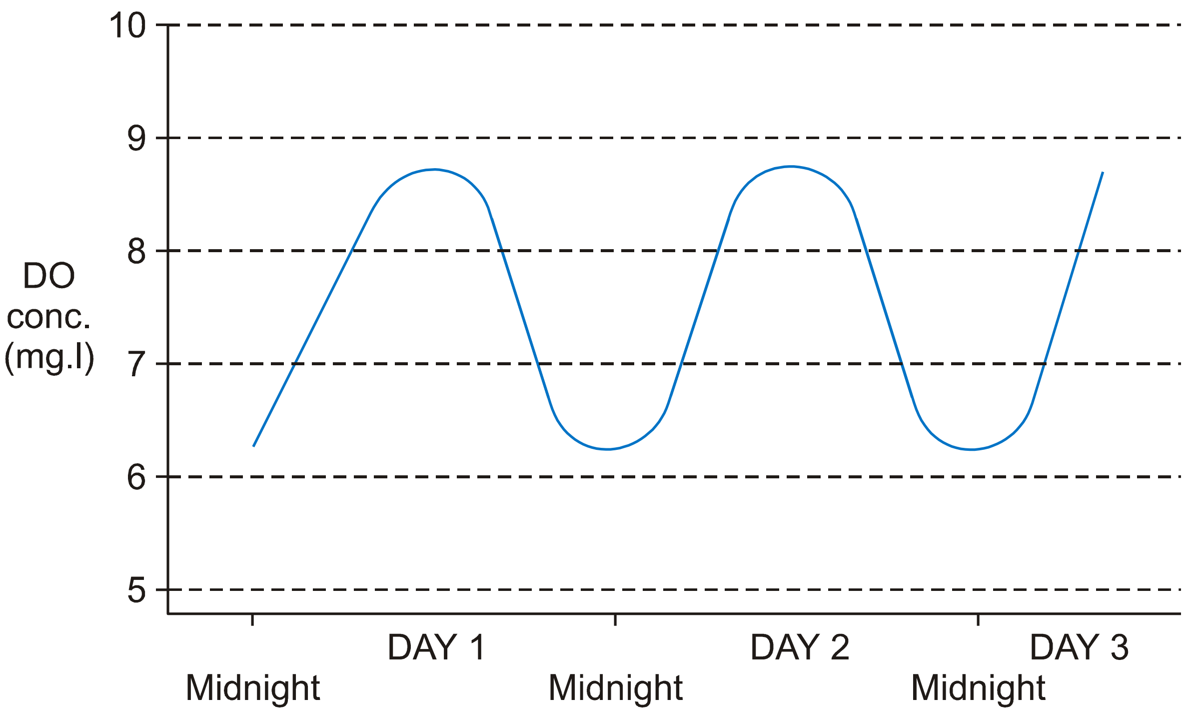


Figure 5.11 Typical diurnal DO concentrations observed in a river

c) Sensitivity checks

It is good practice to complete a few sensitivity checks to assess how the predicted impacts change when sensible changes are made; for example, to process rates, temperatures, the time of day of the wet weather event, etc. This knowledge will help the user to understand the limitations of the model and to interpret the results more reliably.

d) Problems and limitations

Two types of limitation are apparent with dynamic RQIMs. These can be categorised as either limitations associated with a particular piece of licensed software (model specific) or limitations associated with dynamic models in general.

- **Model specific problems and limitations**
  Most models have specific limitations and difficulties peculiar to that piece of software. These limitations may be in the hydrodynamic, advection-dispersion or quality modules. Some may be rectified with future model developments and others may be a fundamental limitation of the model software. Most users who are familiar with a piece of software will be aware of that model's particular limitations, and new users are advised to join the relevant users' group or contact other users to identify these limitations, as frequently they are not identified in the model's User Guide.

- **General problems and limitations**
  – Hydraulics
  The results from a water quality simulation will only be as accurate as the results from the hydrodynamic model. Any errors in the hydraulics will be automatically transferred to the quality model.

  – Initialisation
  All models require correct initialisation and specification of boundary conditions. The model should be in a stable condition prior to any input of pollutants. This ‘stable' condition will, however, allow for diurnal variations, especially in DO and temperature.

  – Stability
  It is often the case that a total eradication of instability in dynamic models is not possible. The model should, however, be free from any numerical instability that affects the accuracy of the results to any significant degree.

  – Modelling low flows
  Problems can occur when modelling water quality in rivers with low flows. The calculations can become unstable and the results erroneous. This can be a particular problem when modelling low flows in steep catchments.

  Modelling tributaries (input as either boundary conditions or via sewer model interfaces) can cause the model to crash if the tributary flow is significantly larger than the flow in the river, such as a final effluent from a large STW discharging into a small river. Normally, this can be partly overcome by reducing either the number of calculation points or the timestep.

  – Calculation points/timesteps
  Some complex models require a small distance and timestep when introducing a large polluting discharge. This is especially the case when modelling steep and complex channel networks, or sediment and BOD interactions. This results in longer run times.

  – Range of applicability
  The model will be most reliable for conditions that are similar to those used for calibration and verification. However, it will normally be necessary to make predictions for more extreme events. Potential inaccuracies as a result of using the model outside its verification range should be quantified.

e) Interpretation of results

All results should be viewed with reference to the model's expected accuracy and limitations of applicability. An associated error band is inevitable within any model. An assessment of this error band, that will be related to the input data and how the model was built, is useful.

The most widely used measure of model adequacy remains direct comparison with observed data. Limitations within the model should also be assessed and quantified where possible.

Suitable safety margins must be used when comparing the model's result against standards. These margins should reflect the accuracy expected from the model, with larger margins for more inaccurate models.

5.6 Tidal waters modelling

Tidal waters modelling covers the use of environmental models that describe waters where tidal movements are significant to pollutant transport. It includes both coastal and estuary environments, and this section of the manual covers both aspects. The complete discipline of tidal waters modelling is very broad and, as with the remainder of this manual, the focus of this section is on the wet weather water quality impacts of discharges from the sewerage network and treatment works.

Depending on the circumstances, the modelling approach adopted can range from simple spreadsheet approaches to complex, dynamic 3D models. It is therefore important to consider what the most appropriate modelling approach is for the specific questions being asked. Where studies involve several organisations, it is important that the overall approach is agreed before detailed work commences.

Modelling of tidal waters generally has two distinct phases:

- Representation of the tidal movement, so that the models accurately describe the transport, advection and dispersion of conservative substances.
- A water quality component that allows for decays, interactions and other transformations of the substances of interest

In some large bacterial modelling studies, where there are complex inputs from a wide variety of sources and the volume of data and associated time taken for processing would be prohibitive, a third post processing unit impact approach may also be adopted.

Accounting for tides can often make the use of simple approaches more challenging, and the outputs less useful, but there will be situations where a simplified approach is reasonable and appropriate. In some circumstances, especially where there is significant uncertainty, even after a study has been completed, it may be considered more appropriate to use surrogate approaches instead of modelling.

Studies will need to be supported by field data sufficient to calibrate, validate and, in some cases verify the models. This data can also assist in an initial identification and characterisation of the sources of pollution, a process known as a source budget study.

In some instances, models of tidal waters may have been previously developed by other organisations, for other purposes, such as navigation. Whilst such models may provide a useful starting point for the hydrodynamic component of the modelling study, it is worth being aware that significant additional work may be required to ensure that they are also fit for purpose for water quality modelling.

The relevant standards that may apply to tidal waters are defined in section 2.3. In estuaries particularly, it should be noted that several different types of standard may be used. As set out in section 3.5, it is key that the use objectives of all receiving waters, and the environmental standards which the environmental regulator requires compliance with, are identified at this stage.

This section is subdivided as follows:

5.6.1 Introduction

Tidal models can be used for a wide range of applications, both in terms of the type of assessment, and in terms of the range of complexity that can be dealt with. They provide a means of understanding environmental conditions (e.g. tides, winds, differing times of discharge) and the potentially complex interaction of discharges (intermittent, continuous and diffuse) which may need to be considered together to develop a useful understanding of impact.

In common with inland modelling, the scoping stage of the study is important, and time should be taken at the start of each investigation to understand the problem, the aims of the study, and the tools and method that will be required. Why a model is needed, and the type of model or the modelling approach needs to be understood. There may be a need to use a combination of approaches to protect receiving waters depending on the importance and vulnerability of the receiving water concerned.

The strategic value of expanding a model, for example, to cover a large stretch of coastline covering several bathing waters, may need to be considered.

5.6.2 Effects of intermittent discharges on environmental quality

The main changes in tidal water quality due to wet weather discharges are:

- A rapid and potentially large increase in the concentration of pathogenic organisms (bacteria and viruses);
- A reduction of Dissolved Oxygen (DO) (this is particularly relevant to estuarine areas);
- A rapid increase in concentrations of ammonia, biochemical oxygen demand (BOD), and suspended sediments.

Bacterial impacts affect bathing waters and shellfish waters. The presence of raised levels of pollution can increase health risks to bathers or to consumers of shellfish, where pathogenic organisms are present. As pathogenic populations are transient, unpredictable, often sporadic and present in relatively low concentrations, indicator species are used which are accepted to identify the potential for exposure to pathogenic organisms. These Faecal

Indicator Organisms (FIOs) are the principal determinands for both bathing waters and shellfish waters assessments. If there is a requirement for direct assessment of pathogens (for example to understand a virus outbreak) then the same modelling approach can be used.

Both FIOs and pathogens are subject to death or a change to non-viable status in the environment (as both are conditioned to survive and flourish in a host, for example a mammalian gut). Temperature, salinity and other factors all play a part in the inactivation of pathogens and FIOs, but the principal factor is exposure to ultra-violet light (sunlight). The impact of UV is modified by factors which protect these organisms from exposure, the most important being turbidity and the potential to adsorb to sediment particles.

Pollutants such as ammonia, BOD and suspended solids will tend to have an environmental impact (often on DO levels) as compared to the potential health impacts of bacteria and viruses. Ecology can be adversely affected in both the long and short term, by restricting or preventing organisms from occupying their normal habitats or providing preferential conditions for other species to colonise a habitat or ecosystem. For example, DO levels have an impact on the ability of migratory fish to pass through estuaries.

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Where there is a complex interaction of sources, impacts may not necessarily always be intuitive, and there is a need to understand how the different sources interact with one another under different discharge scenarios.

The modelling approach selected must have the potential to generate data which can be used to evaluate performance against the relevant standards, over a given period, and provide confidence that all contributing sources are represented adequately.

Source apportionment is the concept of using modelling to identify individual sources and the proportion of impact they contribute under different circumstances, and is vital in understanding how adverse impacts are generated. It is very difficult to apply interventions or mitigations in an effective and efficient manner if the sources of pollution are poorly understood.

Source apportionment is a description often also used for source budget studies. Source budget studies use sampling, monitoring and other field studies to identify key pollutant sources impacting an area, including the scale and range of loads from each source, and are often a pre-cursor to detailed modelling. When a source apportionment project or phase is being referred to it is important to clarify whether this is a modelling approach or a sampling approach.

For other impacts, the most commonly required water quality model is for predicting DO. The processes that affect dissolved oxygen in the water column are:

- Temperature
- Salinity
- Sediment Oxygen Demand;
- Decay of BOD;
- Decay of ammonia;
- Reactions with decaying phytoplankton;
- Respiration;

- Photosynthesis;
- Re-aeration.

The process of photosynthesis adds dissolved oxygen to the water during daylight hours. During the night, when photosynthesis does not occur, plant respiration may deplete oxygen levels in the water resulting in diurnal variation in DO levels that may become extreme in eutrophic conditions. The physical process of re-aeration describes the transfer of dissolved oxygen across the water surface. Whilst these reactions directly affect dissolved oxygen concentration in the water, many other reactions may need to be described in the model because they affect the BOD, ammonia or phytoplankton. For example, in an estuary, the settling of particulate BOD on the bed at times of low currents and subsequent resuspension during stronger currents alters the distribution of the BOD. Since ammonia is hydrolysed from organic nitrogen, that too may need to be described in the model. If photosynthesis is considered significant, it may be necessary to model the availability of nutrients and phytoplankton.

In addition, the adsorption and desorption of substances, exchanging between the dissolved and particulate phases, can be important for substances such as heavy metals and pesticides. It is important to take a pragmatic a view as possible regarding complex interactions, without compromising the value of model outputs, to avoid the exercise becoming too complicated. The complexities involved in understanding water quality processes may mean that those processes need to be parameterised or represented more simplistically. In such cases, the reasoning, justification and approach undertaken should be clearly set out and agreed. Sensitivity testing may help to understand the implication of these simplifications. Whatever situation is being studied, use should be made of all information, including verified subjective information.

5.6.3 Types of tidal water quality impact models

The aim of any tidal waters hydrodynamic model is principally to represent the movement of water (i.e. to accurately characterise the movement of the tides and any currents) such that pollutants introduced to the model are correctly transported. A water quality model will then represent the dispersion and decay of those pollutants.

- **Issues specific to Estuaries**

  In general, currents in estuaries oscillate with the tidal period (about 12.5 hours) and can be stronger than in rivers, especially where the estuary is long, shallow at low water, of reasonably uniform width and has a tidal range of more than a metre.

  The two main physical features that distinguish estuaries from rivers are:

  - Tidal oscillating flow; and,
  - Combination of fresh and salt waters.

  Strong currents imply high dispersion, but a tidal oscillation gives a wide variety of impact scenarios, especially for an intermittent discharge:

  - Low water discharge - when the current velocity turns from seawards to landwards, would result in a 'hot spot' because of poor dilution in relatively shallow water. This could give a high impact on water quality in the short term. The incoming tide would then carry the 'hot spot' of pollutant upstream for some six hours, possibly with little dispersion.
  - High water discharge - though again at a time of slack current, would be into deeper water initially and it would be carried seawards and possibly out of the estuary into the sea.

- Intermediate between these two extremes - wet weather spills would be into tidal currents that could give high dilution by spreading a spill over several kilometres in the estuary.

It should be noted that in some locations, local conditions can result in deformed tides, which can result in either unequal flood and ebb tide times, or double peaked tides. These situations should be identified accordingly.

It is important to realise that estuaries can be complex areas to model, and although simple approaches can be used, there is soon a need to understand complexities. The most obvious are tidal oscillation or reciprocation (the tendency for a pollutant to be retained in the estuary due to small residual movement across tides), and the superimposition of pollutant plumes arising from tidal movement. Considering discharges from more than one source, and the need to represent diffuse sources, means that in many cases, simple river-based approaches may not provide the required resolution. Where they do, they will deliver a much quicker and simpler approach for basic assessments.

Tidal oscillation in a long estuary may imply a long residence time for the material discharged, so for an Environmental Quality Standard (EQS) for dissolved oxygen, this may be difficult to meet as pollutants can exert their oxygen demand over many days.

A further complication stemming from an oscillating flow occurs when the estuary is branched. On the ebb tide, water will flow from tributaries into the main channel and out towards the sea. After low water, flow will return up tributaries, although the phasing of outflows and inflows can lead to considerable mixing between tributaries.

In a hypothetical non-tidal estuary, the seawater would tend to form a semi-static wedge as a lower layer with the entering fresh water moving out across the surface as a thin layer (stratification). Vertical mixing between the two layers would be slow as the density difference suppresses vertical movement (and mixing). This may amplify polluting effects or restrict pollutants to one of these layers.

Most estuaries in the UK are of the well-mixed variety, where fresh and seawater mix completely, though some deep estuaries may be stratified, with a gravitational circulation predominantly landwards near the bed and seawards at the surface. The inflow near the bed may vary in strength from a small fraction of the freshwater flow in shallow well-mixed estuaries to some ten times the freshwater flow in deep estuaries. In such cases a 3D modelling approach will be required. It should again be noted that there may be situations where shallow estuaries can exhibit stratification.

An aspect of both estuaries and some coastal areas is the potential for channels and flats to change position and shape. The model should be developed on the best possible information, and consideration should be given to critically reviewing the potential for channel migration to alter results and predictions. It may be that an alternative bathymetry should be tested, although it is recognised that it is not practical to undertake any validation for a situation which cannot be measured in the field. A clear agreement of scope and approach for these issues should be gained at the start of a project.

It should be borne in mind that the resuspension of sediments in estuaries may have water quality implications, including on bacteria, oxygen demand and hazardous substances. A dynamic sediment model that includes the settlement, resuspension and transport of sediments plus the storage and release of associated pollutants may be required if these impacts are significant.

- **Tidal Waters Modelling**

For all the reasons described above, in complex water quality modelling cases as a minimum a 2D dynamic model will be appropriate for estuary modelling. As this is the same approach required for ‘marine’ modelling, it is appropriate to consider both issues together (as ‘tidal waters’ modelling).

The application of models for estuarine and marine situations may differ, but the principles behind model use remain the same. Overall, for tidal waters, the situation may be complex, with both near and far sources interacting (and in different ways at different times of tide, weather, etc.). Impacts also superimpose due to tidal movement taking pollutants away from a receiver on one tide and bringing them back on the return tide. Whilst this leads to challenges in model build and application, models are very important to understanding impacts. It is therefore vital that input data is accurate and calibration and validation is robust and clear.

There are several types of model, and modelling approaches to consider. Figure 5.1 provides a process through which the modelling approach can be determined.

Where the impact is simple (for example from one source, or very close to a sensitive receiver), and volumes or concentrations are likely to be small, then ‘simple’ approaches are likely to be useful. In addition, they offer the benefit of significantly less time and cost to run than dynamic modelling approaches.

Where a standard is required in the initial mixing zone from a discharge (i.e. beyond the ‘boil’ of the discharge reaching the surface - the ‘near-field’), then an initial dilution approach is appropriate. If an understanding is required of further dispersion and dilution (but within one tide), then a model which can incorporate simple secondary dispersion can be employed.

Neither of these approaches will be accurate where the tide takes pollutant away from the area, but then it returns on the next tide (superimposing of pollutants). Current data (speed and direction) and water depth data are required (these might be obtained from data or an existing coastal model), as are details of pollutants and concentrations. Initial dilution may not be appropriate where the discharge is exposed for part of the tide, and initial dilution standards are likely to be failed (as dilution is zero when the discharge is exposed).

Where the transport is complex (returning on subsequent tides) or multiple sources, multiple (and possibly distant) receivers and complex discharge scenarios will be encountered, then dynamic or complex models are likely to be required. These are likely to be mid and far field type situations, as these complex models cannot deliver assessment in the near field. Where an understanding of both near and mid field quality is required (for example to ensure that a discharge meets basic aesthetic and pollution control standards, and that more distant receivers are compliant) then both simple and complex approaches would be employed – that is a near-field study followed by a mid or far field study.

Note that in tidal waters modelling, several definitions of near- and far- field modelling can be used. The one adopted in this document is based on the description provided below.

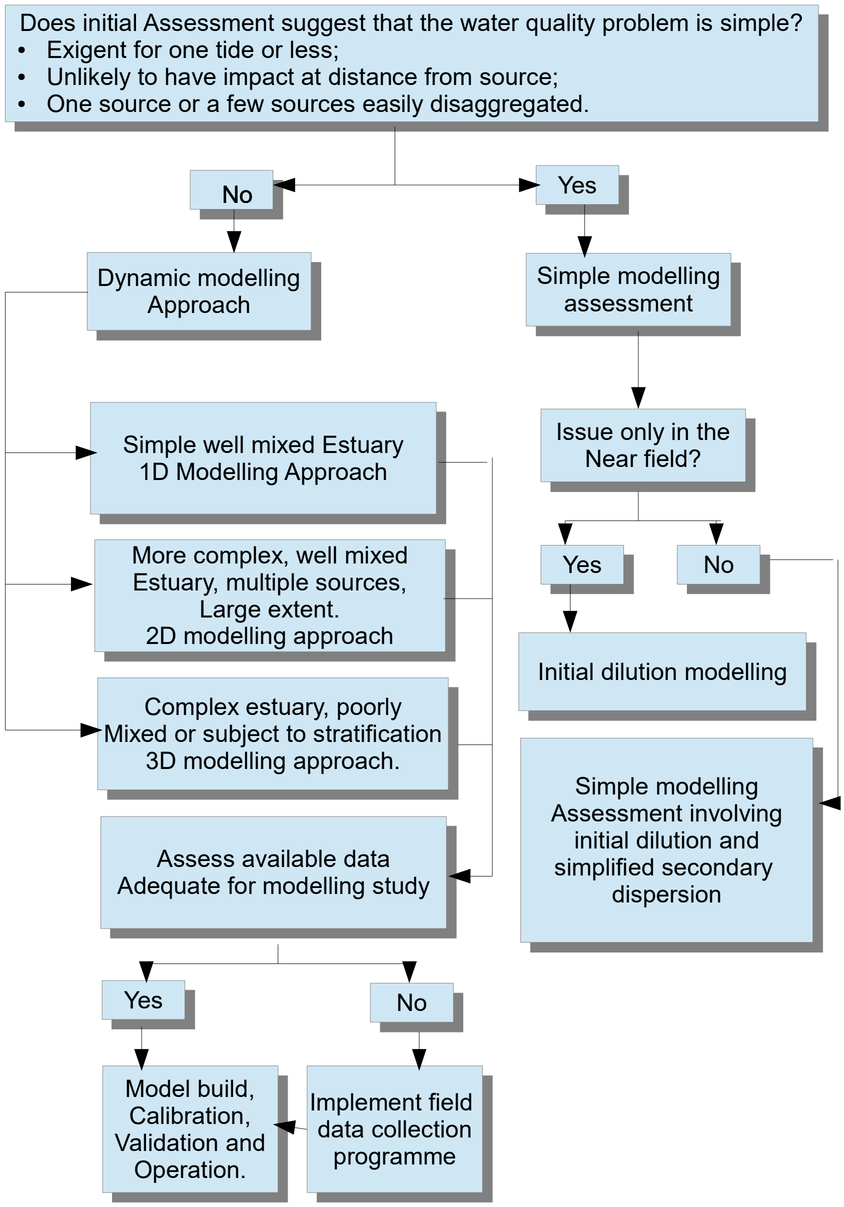

Figure 5.12 Process for detemining type of model to use

- **Near-Field Modelling**

  Near field modelling represents the plume as it is discharged and is moved through the water column. It seeks to describe the initial process of turbulence and entrainment of the discharge in the surrounding water, and the mixing and dilution of the discharge as this occurs. Used largely to determine aesthetic standards for continuous discharges (especially historically when lower levels of wastewater treatment were prevalent) it is an important tool where standards must be met over very short distances or where discharges are made directly into designated receiving waters.

- **Steady-State Near-Field/Simple-Secondary Dispersion Modelling**

  This type of model is generally a spreadsheet-based, steady-state model using current and depth data to deliver near-field output, coupled with basic secondary dispersion. Such models do not generally represent tidal movement well, (for example the superimposition of pollutant due to tidal flux) but can be useful, particularly where there is a requirement to meet a standard over a short distance (for example, mixing zones for a substance with an Environmental Quality Standard). They are therefore most applicable to individual outfalls.

- **Mid-field Modelling**

  This refers to the representation of the transport of pollutants beyond the initial mixing zone, and typically over a timeframe of a few to several days. Transport of pollutant is principally by advection (tidal or current driven movement) and dispersion (mixing at the boundaries of the pollutant plume). Initial dilution (the 'boil' and the initial mixing zone) is effectively ignored.

  Bacterial transport and impact modelling is included in mid-field modelling. This is undertaken with a dynamic model capable of correctly representing tides and current movement, and capable of integrating any wind induced effects. The model used is typically a 2D (depth-averaged) model, but there are situations where a 1D or 3D model approach may be appropriate (more often associated with estuarine areas).

  The mid-field approach can integrate many sources, and can be used to model multiple receivers simultaneously.

  The domain of the model needs to be sufficient to cover all the sources and receivers, and to ensure that pollutant is not lost over the boundary of the model.

  Pollutant, or water quality modelling, can be undertaken either using an advection-dispersion or particle tracking approach. Considerations for selecting the most appropriate approach include:

  - The nature of the environment to be modelled. For example, particle tracking may not be appropriate for estuarine situations with large inter-tidal extents;

  - The resolution of the model (cell size) may have a bearing, when accounting for numerical dispersion in the model;

  - Timescale may be a consideration; particle tracking approaches will generally be much quicker than advection-dispersion (AD) modelling;

  - Ability to represent the associated water quality processes.

If used incorrectly, both approaches can lead to misrepresentation of impacts. For example, particle tracking in inter-tidal areas can lead to particle trapping in drying cells, and inaccurate predictions. An advection-dispersion approach where numerical dispersion is significant can also lead to inaccurate predictions.Use of a particle tracking approach will also require consideration of the number of particles that can be represented in the model. Each particle is associated with a certain pollutant load and therefore particle numbers should not be so low that one, or a few, particle(s) could cause a significant change in predicted concentration. Conversely, there may be a point when too many particles will be required to preserve resolution, and at this stage particle tracking may not be the best option.

- **Far-Field Modelling**

Far-field modelling uses the same basic principles as mid-field modelling, but refers to pollutants which persist and potentially accumulate over a longer period (days/weeks/months). The same modelling tools as for mid-field modelling may be used, but the types of pollutants are those that persist, or that interact with other substances that continue to have a potentially polluting effect. This could refer to nutrients, for instance, or potentially to the modelling of viruses, which typically have much longer survivability than bacteria.

The domain of the model needs to be considered over the longer time periods involved, to ensure that pollutant is not lost over the boundary of the model.

A decision to build a 'regional' domain for a model, as a strategic long-term tool, generally means that all the potential considerations for modelling are covered.

- **Compliance Modelling Tools**

To understand the impact of discharges on the potential of a receiver (for example a bathing water) to meet a given standard, the modelling approach needs to be able to produce a prediction of the standard that can be attained. For example, for bathing waters, this could mean predicting a 90^{th} or 95^{th} percentile value and comparing this with the reference value of the standard. Table 2.8 of this manual shows how the risk of non-compliance is taken into consideration when these bathing water standards are used in the UK. Where detailed compliance assessment predictions are carried out then these are usually based on at least ten years (or ten bathing seasons), such that a representative set of rainfall/wet weather spill conditions can be assessed.

Discharges from intermittent sources can occur at any time; they are not regular or predictable, as they are determined by independent rainfall events. In complex cases, to produce predictions that account for all potential impact scenarios, model runs over all stages of tide (as spills can occur at any stage of tide), wind, rainfall, and possibly river flows need to be undertaken. Model runs should account for all known sources, represented in a way that is suitable for the study being undertaken. Transparency in how estimates or assumptions have been calculated for sources with limited data is very important, as it allows sensitivity testing to be developed, and a reasonable interpretation of results.

- **Unit impact approach**

Undertaking assessments of the impacts of intermittent discharges on tidal water quality directly within the water quality modelling software is likely to have run-time and data generation issues. It is also likely to be inflexible when considering scenario changes, sensitivity, or in responding to input errors. To address this, so that detailed compliance studies can be modelled and analysed within feasible timescales, a unit impact approach is normally adopted.

A unit impact approach describes the modelling of arbitrary flows and concentrations, over a specified period, to establish the impact relationship between a source of pollution (for example a CSO) and a receiver (for example a bathing water). These unit runs are undertaken for all scenarios of tide, wind, etc. This generates a database of unit impacts, across all potential impact scenarios. By using information on actual spill events under specified conditions (for example, the output from a sewerage network model and a specified rainfall time series), the unit run impacts are scaled for actual spill events.

As with all modelling approaches, consideration should be given as to whether this approach is appropriate for the study being considered.

5.6.4 Building, Calibrating and Validating Tidal Water Models

This section discusses building, calibrating and validating tidal water models. There should always be an agreed set of criteria and threshold values upon which the model should be developed against, and assessed. These would include, for example, current speed and direction, tidal level, as well as water quality elements such as dye patches. Demonstrable meeting of those agreed criteria would then deem the model 'fit-for-purpose'. A UK example of such criteria is 'A framework for marine and estuarine model specification in the UK' (WRc, 1993).

During **Calibration**, the internal model parameters are adjusted to find the best fit between model predictions and field data. For example, the parameters adjusted might include bed roughness (hydrodynamics) and dispersion coefficients (water quality). Adjustments should only be made within reasonable bounds, based on physical reality and evidence based. This is not a force-fitting process.

During **Validation**, the model predictions are tested, with the model parameters unchanged, against an independent field dataset. In coastal models, calibration can be undertaken with data taken on a neap tide, and validation undertaken with spring tide data. These two datasets are independent.

Where detailed compliance modelling is carried out then **Verification** is the comparison of the output of compliance assessments undertaken with calibrated and validated tidal waters models to field or sampling data, without adjusting model parameters. For example, the comparison of bathing water monitoring data to model predictions.

Once an acceptable level of calibration, validation and where appropriate verification have been achieved, the model will be 'fit for purpose' (that is, a validated model which is demonstrated as meeting the criteria agreed for use in assessments), if there are no gross changes in the area likely to significantly affect hydrodynamics and/or dispersion. For each study undertaken, there is a process of ensuring the model is fit for purpose for a specific investigation. This may not be the case, for example, if all the hydrodynamic and dispersion field data were collected far from the new investigation site. In this case there may be a requirement for additional field data collection, or it may be that previous studies and/or existing data will provide sufficient information. This should be appraised on a case-by-case basis. Conversely, the fact that a model has been verified using bacterial data does not mean that it is unsuitable for use to appraise other pollutants, although some further calibration and validation or verification may well be necessary.

Calibration and validation can be, and usually should be, a staged process, being carried out first on the hydrodynamic model and then on the water quality model. This reduces the number of independent variables to be considered at any one time and provides a closer focus.

- **Building a detailed tidal waters model**

  The first step in building any detailed 2D or 3D tidal waters model is to build the underlying hydrodynamic model, a major task starting with a decision on the dimensionality to be adopted.

A summary of the key components of the hydrodynamic model build is set out below:

DOMAIN – Model extent needs to be sufficient to cover all points of concern and to retain all pollutant inside the domain. The boundaries should also be sufficiently far from the area of interest to minimise the impact of any boundary errors. The use of a strategic regional model versus a local domain should be considered. Broadly, a large domain will include more areas of interest, and allow them to be modelled together, at the expense of run-time and the generation of larger output datasets.

GRID RESOLUTION – Cell size is a balance of resolution and computational time. Flexible mesh type models can provide increased resolution in key areas but are constrained by the size of the smallest element in the mesh, and the overall number of elements should be considered. Resolution should not lead to issues with numerical dispersion. Numerical dispersion is the process of pollutant being distributed through a model cell and is particularly an issue where grid cells are large and so concentration of pollutant may quickly be inaccurately lowered.

BATHYMETRY – The bathymetry of the study area is required. A key consideration is the mobility of the seabed, and whether any historical data is still representative of current conditions. Where bathymetry is mobile, then consideration should be given as to how this is accounted for as part of the overall modelling investigation.

HYDRODYNAMIC DATA – The applicability of historical measured data should be considered together with the collection of new data.

Examples of data that may be required for a hydrodynamic model include:

- Salinity;
- Temperature;
- Wind data;
- Current direction data;
- Current meter data (magnitude);
- Water level data;
- Drogue data;
- Acoustic Doppler Current Profiler (ADCP) data.

BOUNDARY DATA – This needs to be suitable for the purposes of the model. Boundary data can be generally considered to be in two forms, the outer tidal boundaries and point source boundaries such as rivers and large discharges. If tidal boundaries are extracted from larger scale models, then it should be ensured that the validation of these models is acceptable and appropriate for the study being considered. For rivers and other large sources boundary information can be obtained from hydrology model or recorded data. In estuaries it is especially important to consider if river sources will affect the hydrodynamics/hydrology. This is considered unlikely in many cases, but there will be circumstances where this could be important. In these cases, these sources must be integrated into the hydrodynamic model, and not left for inclusion at the water quality modelling stage.

It is important to stress the need to check and test assumptions (for example domain and grid resolution). Checking and testing these key issues at this stage minimises the risk of the model not meeting all user requirements at a more critical stage of the investigation.

Calibration and validation of the hydrodynamic model requires, as a minimum, tidal level and salinity observations over tidal cycles at several points in an estuary. For 2D and 3D models, comparisons of predicted and observed velocities is usually also expected. Temperature may also be a factor for 3D models.

Once the hydrodynamic model is completed dispersion coefficients are identified. Initially default model coefficients can be applied which will be subsequently adjusted to provide a best fit with available field data.

The classic method of determining dispersion coefficients is the use of readily measured tracers such as Rhodamine WT dye. These have the advantage of being directly measurable in the field, allowing the creation of a dye contour or dye patch which can then be compared with model output.

Dye studies should be designed with care, and should cover a number of scenarios in order to provide data on a wide range of conditions and potential differences in dispersion. High water, low water releases on both spring and neap tides as a minimum should be considered. Experienced contractors with expertise in following, tracking and measuring dye patches are also important in obtaining useful data. Licensing issues, potential restrictions on use and the amount of dye required should all be considered carefully. Dye can also be affected by local conditions. Spot releases can be instructive as they show the general dispersion pattern quickly and relatively cheaply. However, they can be less important in charecterising the dispersion of a continuous discharge or dischrges of several hours duration. Thus, continuous tracer releases over at least one hour should also be considered.

The use of alternative tracers, for example bacteriophage or spore tracers, can provide similarly useful data and may provide additional information on the connectivity between sources and receiving waters. These types of tracers must be sampled and analysed in a laboratory, but in a robustly designed field programme can provide further calibration and validation information.

A complimentary approach is the use of salinity data in instances where there is measured data of changing salinity in different tidal stages (for example where a river influences a bathing water, or in an estuary). Salinity data is often regularly taken with bathing water samples, and there may therefore be a long-term dataset available. Although not an approach available in areas where there are no freshwater inputs, it should be used where available and where the data is appropriate.

The importance of salinity (and the use of tracers) is their conservative nature and their predictably. Where freshwater enters seawater, there is generally a well understood and replicable mixing of waters.

Salinity changes in the vertical profile is an important element for 3D models, and data should be collected accordingly.

The rate of bacterial die off (T_{90}) is generally the key consideration for bacterial modelling. T_{90} values should be set sensibly since adjusting T_{90} without consideration for reality will provide any answer required. Experience and reference to previous studies and academic literature will help in the selection of appropriate T_{90} rates. The scale of the study will affect choices regarding a varying decay rate (for example the modelling of a specific scenario) or a representative value (for example compliance modelling of bathing waters over several years).

- **Other Water Quality Modelling**

Although bacterial modelling has been the principal interest in many previous studies, modelling of other water quality parameters may also be required. For example, to model DO accurately, sources of oxygen demand need to be identified and represented. A key process in many estuaries is Sediment Oxygen Demand (SOD), and if it is omitted, DO values are likely to be over-estimated.

Ammonia is a potentially acute pollutant with toxicity issues, but as part of the nitrogen cycle is also a factor in nutrient enrichment.

For complex issues, the modeller may be required to represent relatively complex interactions. There is therefore a consideration regarding parameterisation (simplified representation of processes), probably coupled with sensitivity, to be had.

- **Water Quality Verification**

Where detailed bacterial compliance modelling is being carried out, using the calibrated and validated hydrodynamic and water quality model, water quality processes (such as decay or chemical interaction) are then incorporated for the study for which the model is to be employed. Bacterial sampling data would be used for source characterisation, spill volume and spill rates would be obtained from sewer models or measurements, and estimates or assumptions used for decay rates (using available local data or published research evidence). With correctly characterised sources, when the water quality model is run, the model output will be comparable with sample data. This is essentially a process of historical verification, using monitoring data. It is important to analyse the monitoring or historical data to ensure that there have been no step changes in source character that will affect model comparisons.

During this verification process it is sometimes necessary to adjust factors such as decay, without considering a re-calibration of the transport model. T_{90} is best represented as a narrow band of values (to initially test). Anything outside this range will need further justification and consideration of the specific application, the available field source apportionment data or rarely further model calibration. Bacterial decay is dominated by exposure to ultra-violet light, and this will be affected by turbidity, insolation, latitude, and salinity. All these factors need to be considered when determining an appropriate range of T_{90} values.

Where assumptions or estimates are used, then the sensitivity of the model to these assumptions should be tested, where it is considered appropriate.

It is therefore at this stage that the accuracy of the prediction of bacterial pollutant impacts is assessed. It is undertaken at this stage because it is only at this stage that a full characterisation of sources has been undertaken, and so it is reasonable to expect the model to represent bacterial or other water quality parameter sampling undertaken in the field.

A ‘fit for purpose’ model must be validated or verified for the range of weather or operational conditions that are being investigated. For example, for a compliance investigation of a bathing water, a full range of weather conditions would need to be accounted for in a bathing season. For a shellfish study, the full range across a year (or years). For a direct modelling assessment of, for example, a specific bathing water failure, then the environmental conditions (weather, tide, river flow, etc) at the time of the failure would need to be accounted for.

- **Potential Issues with Calibration, Validation and Water Quality Verification**

  There are several issues which can arise during the calibration/validation/verification process. A number are considered below, to provide general guidance for commonly encountered issues. Again, a familiarity with the general process of calibration and validation is assumed, and so issues directly associated with the application of tidal models to the UPM process are discussed. Issues with observed data are often key sources of error.

  It should be remembered that all calibration processes are inherently iterative. Testing and adjusting are part and parcel of the process.

- **Verification of the water quality impacts not matching**

  Issues to consider when modelled and observed information do not match include:

  - Checking the performance of other model inputs (river, sewer);
  - Examining the output data, to see whether the data suggests a missing source, or that the characterisation of identified sources is incorrect;
  - Ensuring that key model areas function correctly – not just one point. Conversely, there may be areas of the model where accurate performance is not required and therefore this does not need to be focussed on;
  - Whether significant changes may have occurred to the bathymetry in the area being studied over the period being studied.
  - Reliability of field data/collection or analysis errors, etcUseful outputs that can inform overall process
  - Knowledge of residuals and residence/oscillations (estuaries) – drogue releases in model, for instance, as well as field data;
  - Including supporting evidence/data along with comparison of fitness criteria;
  - Demonstrate understanding of limitations of both models and fitness criteria, and field data.

UPM models are management tools which should deliver accurate outputs regarding the transport and impact of pollutants. As with all complex modelling, the degree of representation of all processes in detail is a balance which needs to be achieved in a pragmatic manner.

As with all modelling, the degree of fit should be assessed considering that the tool and the study is not necessarily designed to the same level of detail as, for example, an academic study seeking to understand in detail processes and relationships. The concept of parameterisation and the representation of detail by proxy or aggregation and estimate should not be discounted, if it is clearly explained and documented, and checks, such as sensitivity testing, are delivered to understand the implications of assumptions.

5.6.5 Using Tidal Waters Impact Models

Broadly, there are two approaches to consider:

Compliance modelling: This is the generation of data to calculate the statistical distribution and estimate of compliance covering a period of several years or more. This will produce a compliance prediction directly comparable to the given percentile standard over the complete range of potential environment impact scenarios. It generally requires the integration of impacts from all identifiable sources on a given receiver, and, taking a bathing waters study as an example, would typically be undertaken over at least ten years.

This is achieved using a combination of tidal model, sewer model (and potentially river models) together with a means of integrating multiple unit impacts to collate and analyse the data from the water quality modelling assessment.

Direct modelling: Modelling of a shorter period - a discrete event or scenario, directly within the tidal waters model. Typically to test the performance of a given source or sources under prescribed conditions. While, equally applicable to single or multiple source testing, this approach is more difficult to use to produce estimates of compliance covering a long period.

- **Source identification**

There are several potential sources to be considered, depending on the precise requirements of the study:

Continuous discharges

Intermittent discharges (Surface water discharges are included in this category.)

Diffuse sources

Source identification is a critical stage, and as many stakeholders as is practicable should be consulted. This will include operational staff in water companies (knowledge of how assets perform), Regulator staff (may be particularly important for diffuse sources), and may include other sources such as Internal Drainage Boards, Universities, industrial bodies, etc.

- **Supporting Data**

An important consideration in any study is the amount of good data which exists, and the level of new data collection that will be required. This is almost certainly a project specific consideration, and the following points and issues will need to be considered to establish specific requirements:

- The aims of the study;
- The degree of accuracy, or resolution between sources required (for example, whether there is a requirement solely for the provision of a 'river' source, or additionally for a breakdown of sources within that river);
- Any step change in source characterisation (for example asset investment) which may render historical data obsolete;
- Sources of local data which may be of use – academic studies or PhD research can be a valuable source of data.

As a rule, a study that attempts to represent a complex situation will require comparatively complex supporting data - a good level of knowledge of local sources and their characterisation will be essential. In many cases, this may be met by existing data. Where it is not, then applicable field surveys will be necessary.

- **Considerations of Model Data**

In addition, and to some extent dependent on the modelling tools being used, there will be a need to specify extraction points or areas in the model domain. Extraction points are those points or co-ordinates where data is output from the model for further detailed analysis. In some software, extraction points need to be identified prior to model runs. In these cases, the agreement and selection of appropriate extraction points will be critical. Extraction of data will generally be:

Single extraction point, e.g. a Designated Sampling Point at a bathing water.

Multiple extraction points, e.g. several designated sampling points if several bathing waters are modelled concurrently.

Areal - To understand overall compliance or impact over a designated area, e.g. shellfish water, the whole bathing water area or estuary for water quality issues.

The amount of data that needs to be extracted may have an impact on model run times, and the volume of output data generated. For some model approaches, extraction points may need to be identified prior to the running of the tidal waters model. If further points are required, then the model must be run again. It is therefore crucial that requirements are identified at the outset of the project.

- **Ensuring sources and receivers are modelled for all stages of the assessment**

For all tidal models, the cell that a discharge occurs in, or a data extraction point is identified by, must be 'wet'. That is, the cell must have the minimum depth of water which has been identified by the modeller as the minimum depth to indicate 'wetness' (i.e. for transport and pollutant processes to be modelled). When water levels fall below this depth, the cell is effectively turned off for that timestep. Discharges will not occur, and impacts in that cell will not be calculated. The model will not necessarily indicate when and where this is happening. For discharges which occur in the intertidal area (and many intermittent discharges fall into this area), there is a risk that for part of a tide, there will be no discharge from these sources, when there should be. This discharge will therefore be under-represented in any assessment. For extraction points, impacts will be under-represented if the cell becomes dry at any point during an impact phase.

One approach for overcoming this is to use the concept of a 'wandering discharge' or 'wandering extraction point'. The points move to the next wet cell as the tide advances and recedes, ensuring that discharge or impact occurs in a cell that is both 'active' in the model and as close to the actual discharge or extraction point as possible. The track of this wandering line should be assessed and decided according to local information, experience and expert judgment. An alternative less useful approach is to alter the location of the outfall or extraction point in the model to a cell that is wet all the time. Some sensitivity testing may be required.

- **Environmental data – wind and rainfall**

It is likely that wind data will be required for any assessment. Wind data will be required to test the sensitivity of a given situation to wind effects impacting on the direction and rate of transport of pollutant (advection).

The wind data should be representative of the local conditions, cover a sufficient period, and should be used in a manner appropriate to the study being undertaken.

If rainfall is not integrated into network or river model output for use in the tidal waters assessment, then appropriate rainfall data may be required. Often this will be accounted for in other models.

In all studies where rainfall is required, it is essential that all elements of the assessment (tidal waters modelling, sewer modelling, river modelling) use rainfall data on a common basis. This may mean that a tidal waters modelling assessment will require the production of an upland rainfall series, where a whole catchment is being assessed. Historical rainfall is preferable to stochastic datasets, as it can help understand temporal and spatial extent and variability. To provide confidence in the integration of sewer model and river model output used in the tidal waters model, it is useful to calculate annual spill volumes and spill frequency for each discharge for checking against monitored sewer discharge and river modelling data.

It is essential that sewer, river and tidal modellers work together to ensure that a full understanding of the issues exist and all parties understand what is required of each other.

- **Sewerage network data**

It is likely that sewer network model data will often be used as part of a coastal modelling investigation, and particularly for compliance assessments. To ensure that the network model output is fit for purpose it is recommended that the following are checked:

Duration – at least ten years of contemporary data is likely to be required.

Timestep – the timestep of the output should be appropriate to the tidal waters modelling being undertaken. It is probable that a timestep of 15 minutes will be acceptable.

Source – it is preferable to use an historical rainfall series where possible, as opposed to synthetic series.

- **River Data**

River flow and quality data may be the product of river model output, or it may be the result of field data collection and the use of local gauging data.

There is likely to be a requirement for both flow and quality data. There needs to be a sufficient resolution to capture flow changes that arise from the wet weather events of interest. We would suggest that daily average flows are not suitable for example..

Attention must be paid to the fact that decay of pollutants will be different in freshwater and saline environments (for example bacterial decay is longer in rivers).

It is essential that river and tidal modellers work together to ensure that a full understanding of the issues exist and all parties understand what is required of each other.

5.6.6 Interpretation of results

Tidal models, especially when used to deliver compliance results, offer the opportunity to gain large amounts of information regarding the transport and impact mechanics of pollution and the importance of specific sources. Used in a direct mode, they provide clear and unequivocal evidence of impact and significance – provided they have been set up and used correctly, with the right degree of experience and expertise involved at each stage.

The other key element is their use in solution evaluation. The outputs can be used to direct solution development, and then to test solution options. Models offer the potential to deliver optimised solutions, using post processing or other complimentary analytical tools. The optimisation process can be applied over multiple sources. There are several issues to consider in the interpretation of modelling results:

- **Sensitivity**

  Processing power, data availability and experience in modelling studies has improved the understanding of how to apply coastal models to impact analyses, and as such the default position that models are likely to be inaccurate, and therefore large amounts of sensitivity or conservative assumptions are required are outdated. There is of course, an important role for sensitivity analyses, especially when the potential assumptions required for the characterisation of sources is considered. Some sensitivity testing is essential because the modelling system is, by definition, a simplified representation of reality.

  A reasonable starting point for sensitivity testing is to consider those assumptions or estimates that could reasonably introduce uncertainty into the modelling study.

  Sensitivity tests should therefore be included as a part of the whole modelling study, with agreement as to which parameters should be tested, or the conditions in which sensitivity testing will be required. Common considerations for sensitivity testing would be:

  - Bacterial decay rate;
  - Pollutant concentration;

  Other parameters could be considered as necessary. There may be situations where the bathymetry of the model may be subject to sensitivity. This is more likely in areas of mobile mudflats or sandbanks, where there can be gross changes in the channel morphology, and as such the transport of pollutant can be significantly changed. Models can still be useful in such situations, but the validated model bathymetry will only represent one set of conditions, and tests should be undertaken to investigate the significance of channel migration.

- **Source Apportionment**

  Output from the models can isolate each source, as well as combining all the sources to predict overall impact. This means that the models can provide insight into the proportional significance of each source to performance at or in a sensitive receiver. An understanding of the proportional significance can help in improving performance or environmental quality. Apportionment can also demonstrate the relative importance of, for example, point and diffuse sources.

  The maximum value will arise from this analysis where sources are resolved to an individual source level, but there is value in a study delivering data regarding the overall contributions of assets, private sources and diffuse, for example (where it may be that diffuse pollution is represented as a single representative source).

- **Solution Evaluation**

  Where discharges are identified as contributing to or causing a breach of a water quality standard, then mitigation solutions will need to be considered. Dependent on the nature of the impact and the key sources, these might entail:

- Enhanced levels of wastewater treatment;
- Improvements to intermittent discharges (for example by increased storage to reduce spills);
- Better operational interventions;
- Catchment improvements (for diffuse sources).

In developing solutions, tidal waters models provide the initial starting point of designs, as these models have determined either the level of non-compliance, the key sources or both. Where detailed compliance modelling is done then the models provide predictions of the amount of pollutant which needs to be removed from the environment to deliver compliance.

- Where spill frequency or spill duration solutions are a regulatory requirement, modelling studies may still offer several benefits:

- Such studies help identify those intermittent discharges that are significant to compliance;
- They can test the extent to which solutions based on meeting different types of standards will provide compliance;
- They also present a means of testing potential design solutions, and therefore solutions can be optimised.
- Providing underpinning evidence to understand, articulate and maximise the benefits versus costs of potential improvements.

5.7 Integrated urban pollution modelling

This is the final section on modelling tools, and discusses the need to develop and build a tool that can combine the results from models of the individual components. The preparation of an integrated tool forms a discrete step within the process. Such a tool is required to allow the rapid simulation of many rainfall events so that compliance with the standards may be tested, in Section 6 of the UPM Procedure.

This section is subdivided as follows:

5.7.1 The need for an integrated modelling tool;

5.7.2 Audit trails;

5.7.3 Transferability of results between models;

5.7.4 Simplification.

5.7.1 The need for an integrated modelling tool

In carrying out a UPM study, planners/modellers must select the various components described in the previous sections, tailor them to the particular situation being studied and use them in an integrated way. The strength of the UPM approach lies in this holistic approach that seeks to take account of all the interacting factors that can contribute to wet weather pollution problems.

The complexity and multiplicity of the UPM tools is also a potential weakness of the approach and this needs to be recognised and addressed. Specifically, what is needed is a tool or procedure that allows the integration of the different UPM components so as to promote their efficient use in testing the compliance of proposed solutions with standards in Section 6.

Individuals and organisations approaching this issue will undoubtedly develop their own procedures and tools for this purpose - there is no prescriptive way of achieving efficient integration. However, the following points need consideration in developing such procedures and tools.

5.7.2 Audit trails

One of the complexities in UPM modelling is the large number of runs with different event details, different model components and different model configurations. Results data are often transferred from one model to become input data for another model and the results from, say, the river impact model for a large number of events are finally processed to reach a conclusion about compliance or non-compliance.

It is essential that a sound system is developed to record run and version details so that a clear audit trail exists. This is needed to track back from the final compliance assessment calculation to the event details that have been simulated to support the assessment.

5.7.3 Transferability of results between models

Spill volumes and loads generated by sewer models often become input data for river impact or marine impact models. There are similar transfer interfaces between sewer and STW models and between STW and river models. Problems can arise at these transfers because of different file formats, different units and different time steps. In addition, there can be incompatibility between the determinands used in different models.

Sound procedures are needed to identify and resolve these issues early in a study. The aim is to achieve efficient transfer of data between models and also to fully appreciate the implications of any differences in the way critical quality processes are represented.

5.7.4 Simplification

Many of the models described in earlier sections are detailed deterministic simulation models designed to provide an accurate representation of system performance under wet weather conditions. In theory, it is possible to run these models with large numbers of rainfall events and different background conditions to demonstrate the performance of sewer/STW systems and to compare this performance with the appropriate standards. Current performance can be assessed, and, by repeating these runs with, for example, extra sewer storage, potential solutions can be evaluated. However, this procedure can be very onerous because of the time and effort involved in multiple runs with complex models. Some compromises must be madeWith increased computing power, these problems will diminish.

An alternative approach, that can overcome the current limitation, is to create a simplified model of the urban system, calibrate this against a small number of detailed model results and then use the simplified model for multiple runs, and possibly for continuous simulation. This approach also involves compromises, as some accuracy is lost in the description of individual events. However, this loss of accuracy is more than compensated for by the greater range of event simulations that are possible and, hence, the greater overall confidence in performance assessment.

Using a simplified urban pollution model allows greater account to be taken of the variability in rainfall, river conditions, marine conditions and foul flow quality and how the interaction of these factors affects the probability of exceeding the critical standard thresholds. In some situations it may be appropriate to simplify just one component of the whole system. For example, a very detailed sewer flow model may be grossly simplified and yet retain good hydraulic representation at critical overflows. Dynamic river impact models with multiple cross-sections and complex water quality processes can also be greatly simplified once the critical aspects have been identified. Often it is possible to assess performance using steady-state runs rather than dynamic runs.

In other situations, it may be useful to simplify all the elements of the urban system (sewers, STW and river) into one integrated model. Potentially, a good integrated model will take account of all the data transfers between different elements and will provide a sound basis for establishing audit trails.

A simplified model can be very focused as a design tool in the task of checking compliance with specific standards. In developing such a model, a planner needs to have a good appreciation of what is important and what is not important for the task in hand. This will involve judgements based on a very good understanding of the water quality processes in operation in the study area.