Experience from recreational waters

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Management of recreational waters is changing rapidly to accommodate recommendations driven by World Health Organization (WHO) (WHO 2003). Parallel developments in drinking water quality management have also been evident (WHO 2004). These water quality Guidelines establish the concepts of ‘profiling’ of recreational water and ‘water safety planning’ for drinking water, both based on hazard analysis and critical control point (HACCP) principles. The parallel development of real time prediction of adverse water quality conditions to facilitate the control of public health risks through the implementation of an appropriate management system presents identical challenges and opportunities in both recreational and shellfish harvesting waters. This chapter outlines some lessons from studies designed to support implementation of the WHO approach to recreational water management and suggests key research and management questions in the risk assessment debate for both water types.
15.1 GUIDELINES DEVELOPMENT

The WHO Guidelines for Safe Recreational Water Environments (GSRWE) (WHO 2003) outline a radical new paradigm in environmental regulation which presents both a challenge and opportunity to the regulatory community. Stated briefly, the GSRWE develops two guiding principles of recreational water management. The first is that numerical microbiological standards should be based on epidemiological evidence of health risk and the second is that the Guidelines should be sensitive to the environmental processes and variability in the main parameters used to measure ‘compliance’ at bathing waters – the faecal indicator organisms (FIOs).

To assess the evidence on which to base the GSRWE, the WHO undertook an internal review of the international epidemiological literature (Pruess 1998). The objective was not to produce a meta-analysis of the health evidence base relevant to recreational water exposures but, rather, to facilitate selection of the most appropriate protocols and dose–response relationships for use in the standards design process. This process resulted in the formulation of numerical standards published as a WHO Consultation Draft in 1998 (WHO 1998).

The mathematical derivation of the risk-based standards utilized a new approach which combined a probabilistic measure of exposure, in the form of a probability density function for intestinal enterococci concentration in the recreational waters, with a dose–response relationship giving a continuous risk assessment in terms of the probability of a bather acquiring gastroenteritis from one bathing exposure (Kay et al. 1994; Fleisher et al. 1996; Kay et al. 2004). Box 1 illustrates this procedure for a ‘theoretical’ recreational water. Clearly, exact risk assessment would require both the geometric mean faecal indicator concentration and its log10 standard deviation for any site (assuming log10 normality for the FIO predicting illness, intestinal enterococci). An earlier set of water quality criteria, developed for the States of Jersey, utilized both distributional parameters (Wyer et al. 1999). However, this was felt to be too complex for international implementation and the WHO team of independent technical advisers suggested a single parameter, the 95th percentile, should be used in the published guidelines (Kay et al. 2004).

The numerical criteria published in the WHO Draft Consultation (WHO 1998) were more stringent than the mandatory standards in force in Europe and North America and it was clear to the scientific community that simple implementation of the numerical standards would result in very significant increases in beach failures world-wide. Furthermore, examples were cited of locations only marginally affected by human sewage discharges but which would still fail the proposed numerical guidelines due to ‘normal’ variability in

Detailed consideration of this aspect was undertaken at a further expert consultation in Annapolis, USA which led to the ‘Annapolis Protocol’ (WHO 1999). This established the process of sanitary inspection which has become

\[ \log_{10} \text{faecal streptococci (per 100 ml)} \]

**Box 1** Epidemiological dose–response relationship and probability density function to derive water quality guidelines from epidemiological results. Source: Kay et al. (2005).
known as ‘beach profiling’ and the linked process of risk management through real-time prediction of adverse water quality and the provision of timely public information through the provision of advisory notices. At the core of this ‘management’ approach was the view that remediation of the non-point pollution inputs to the bathing zone, which frequently dominated the total FIO flux at the crucial high flow periods when non-compliance was most evident, could not be achieved quickly and without significant change to current farming practice (Kay et al. 2007). However, the pollution loading from the farming sector could not be ignored in view of its likely loading of zoonotic pathogens derived from livestock (such as *Giardia* spp., *Cryptosporidium* spp. and *E. coli* O157). Thus, public health protection was best achieved through real-time warnings to inform the public of this risk of predominantly non-human FIO sources. It was considered acceptable for the numerical compliance assessment not to use any sample results acquired during the period for which an advisory was in force in the calculation of the 95th percentile value. Thus, the risk assessment implied by the calculation of the 95th percentile value would only relate to the period during which the public were exposed to the recreational water with the approval of the regulatory authorities.

This is perhaps the most radical aspect of the new guidelines, specifically, the departure from the ‘traditional’ regulatory approach which requires strict numerical limits derived from agreed sampling and analytical protocols. It does, however, provide a means of establishing health evidence-based numerical guidelines at the times of bather exposure, even in an area where environmental background variability would cause a beach to fail because of adverse water quality during periods when bather exposure was likely to be low due to the very climatic conditions causing the water quality deterioration.

### 15.2 CURRENT DEVELOPMENTS

The Commission of the European Communities (CEC) have been the first agency to incorporate elements of the numerical values and the ‘management’ approach into a revised EU Directive on bathing waters (Wiedenmann 2003; CEC 2002, 2004, 2005, 2006). The Commission also utilized research findings from German epidemiological work (Wiedenmann et al. 2006) initiated in response to the observation by the WHO expert advisers that the epidemiological base for the 2003 WHO GSRWE was uncomfortably narrow (because it was based on north European marine waters; Fleisher et al. 1996; Kay et al. 2004). The German epidemiological studies were conducted using, as far
as possible, an identical protocol and questionnaire survey to the earlier United Kingdom randomised trials to ensure data compatibility as recommended by WHO.

The process of European Union (EU) Directive revision has generated intense scrutiny by the policy and scientific communities which has focused on both the CEC proposals and the scientific basis of the WHO Guidelines themselves (Kay et al. 2001). Interestingly, the United Kingdom Government’s regulatory impact assessment for the proposed Directive suggested that the financial impact on the United Kingdom, as an EU Member State, would be almost neutral if a management system could be established to facilitate discounting of up to three samples per year during adverse weather conditions (DEFRA 2003a).

In the expanding EU, implementation of the Water Framework Directive (WFD), together with proposed changes to the systems of financial support for the farming community, offer potential means of reducing diffuse pollution from agriculture and its impact on both recreational and shellfish harvesting waters (Anon 2000, 2003a,b; Kay et al. 2007). Under Article 11 of the Directive, Member States are required to design a ‘programme of measures’ to achieve the standards defined in the daughter Directives listed in Annex VI which includes both the Bathing Water and the Shellfish Hygiene Directives. Thus, for the first time, the mandatory requirement for the integrated control of faecal indicator fluxes from both diffuse and point sources has been formally enshrined into environmental legislation in Europe.

This has significant implications for farming communities in Europe. However, there are policy developments which could offer significant opportunity for amended farming practices in key catchments draining to recreational and shellfish harvesting waters. The policy change is the revision of the Common Agricultural Policy and the principal drivers are the decoupling of farm support payments from production or ‘headage’ and the introduction of ‘cross compliance’ (a new explicit linkage between the farmer’s ‘performance’, in animal welfare and environmental management, and the receipt of the ‘single farm payment’). Whilst not currently linked to potential effects on ‘protected areas’ as defined in Annex VI of the EU WFD, it is difficult to envisage that this established mechanism would not be utilized if an EU Member State was at risk of infraction proceedings by the CEC for non-compliance with statutory criteria in Directives covering bathing and shellfish harvesting waters.

In the United States, the Clean Water Act (CWA) enshrines many of the same principles as the WFD (Horn et al. 2004) but its implementation precedes the EU legislation by over a decade, thus, providing some interesting insights. Where water quality is defined as impaired under the CWA and fails to reach target levels, the CWA requires that a Total Maximum Daily Load (TMDL)
assessment is undertaken to rectify the impairment (parallel to the WFD Article 11 ‘programme of measures’). Some 64,628 water quality ‘impairments’ were reported between January 1996 and June 25, 2007 and 25,255 TMDLs were approved by the United States Environmental Protection Agency (USEPA) over the same period (Elshorbagy et al. 2005). The top five reasons for water quality impairment leading to an agreed TMDL have been: ‘microbial pollutants and pathogens’ (in fact, FIOs impacting on bathing and shellfish harvesting waters) (5111 TMDLs); heavy metal pollution (5072 TMDLs); nutrients (3521 TMDLs); sediments and siltation (2682 TMDLs); and organic enrichment and low dissolved oxygen (1425 TMDLs). Some 4525 TMDLs, for all impairment causes, were approved by USEPA in the single fiscal year to 30 September 2006 (Hyer and Moyer 2004; Kay et al. 2006).

It is interesting to note that microbial water quality ‘impairments’ of bathing and shellfish harvesting waters were the most common reasons for US TMDL studies, suggesting a higher US prominence for this area than, for example, nutrients, pesticides and oxygen demand which have all received far more attention to date by the EU regulators and policy makers addressing the implementation of the WFD (DEFRA 2002, 2003b).

Kay et al. (2006) reviewed the operation of microbial TMDLs in California, USA and concluded that the longer US regulatory experience with examination of catchment microbial dynamics through TMDL assessments had not, to date, produced more operationally useful empirical science or modelling approaches which could be applied in the United Kingdom. In effect, many US authorities were defining FIO ‘discharge’ concentration limits for discharges to streams and coastal waters (in TMDL terminology the ‘concentration-based pollutant allocations’) which were simply set at the allowed environmental ‘receiving water’ concentrations required for recreational and shellfish harvesting waters for relevant discharges (a geometric mean faecal coliform concentration in agricultural and surface drainage discharges to tributary streams of <200 100 ml$^{-1}$ and a 90$^{th}$ percentile for faecal coliforms in direct discharge to the coastal water of <43 100 ml$^{-1}$). Waste water treatment plants and boats were required to achieve a faecal coliform median of zero 100 ml$^{-1}$. The TMDL study examples described by Kay et al. (2006) did not address the spatial and temporal characteristics of the inputs or their fluxes which are more relevant than ‘concentration’. Nor was the feasibility of achieving these criteria addressed. Additionally, sampling programmes were recommended which could not capture data on the hydrological events which studies world-wide have suggested account for >90% of the catchment-derived faecal indicator flux from diffuse source pollution (Kay et al. 1999; Lee et al. 2002; Lee and Kay 2006).
15.2.1 Real-time prediction of water quality

15.2.1.1 Simple univariate ‘trigger’ systems

The United Kingdom regulatory community has been quick both to assess the impacts of the WHO ‘management’ approach to bathing water regulation and to field-test systems designed to communicate the results of real-time prediction. In Scotland, a new system of ‘signage’ has been tested which relays the predicted water quality on each day during the bathing season to the beach front or adjacent car park (McPhail and Stidson 2004). This simple system is based on river flows and rainfall in upstream catchment areas and the Scottish Environmental Protection Agency (SEPA) simply use whichever trigger parameter best predicts non-compliance at the identified bathing beach. Data from telemetric rainfall and stream flow gauges in the contributing catchment are received in a central control room where scientific staff decide if the information received would be expected to produce non-compliance based on historical data (Plate 15.1). After initial misgivings within local communities dependent on visitors using bathing waters, subsequent reaction to this signage system has been generally positive.

Plate 15.1 Real time prediction signs as installed at bathing water locations in Scotland, United Kingdom.
15.2.1.2 Multivariate regression based systems

Multivariate regression to underpin prediction was first used in the United Kingdom at the Fylde beaches in the North West of England where the intention was to explore the reasons for continued non-compliance following significant infrastructure expenditures (Crowther et al. 2001). The multivariate approach has, however, progressed in the United Kingdom with studies in the Cardiff Bay impoundment where a simple operational spreadsheet model has been employed to predict real-time water quality from calibration data acquired in the previous year. Figure 15.1 shows set of predicted and observed data for 2004 (Stapleton and Kay 2004). Artificial neural network modelling has also been employed with good prediction within the calibration range but unproven inter-year transferability (Brion et al., 2005; Kashefiipour et al. (2005). Others have employed a hydrodynamic modelling framework to underpin associated health risk assessment (Elliott 1998; Harris et al. 2004).

In North America, the US Geological Survey have issued guidance on real-time prediction options for the beach management communities (USGS 2003, 2006a, 2006b) which recommends a multivariate approach to prediction. A similar conclusion was reached following a three-year United Kingdom research programme funded by the EU (Anon 2006). Perhaps the principal limitation in any modelling study where the outcome variable used to calibrate the modelling system is a microbial determination in water or shellfish flesh. The principal reason for this is the imprecision in microbial enumeration which increases the random stochastic disturbance term and, thus, reduces the

![Figure 15.1](image-url)

**Figure 15.1** Actual and predicted concentrations of *E. coli* in Cardiff Bay, United Kingdom; plotted against EU recreational water criteria (Directive 160/76/EEC).
explained variance. There is, thus, considerable research potential in enhancing the precision of this dependent variable by, for example, replicate enumerations where new data sets are being specifically generated to underpin predictive modelling.

15.2.2 Source apportionment studies

Most modelling attention to date has focused on the nearshore zone (Jin et al. 2003) with very little research and monitoring effort directed to define the complex and highly episodic mix of inputs from both point and diffuse terrestrial sources (Ferguson et al. 1996; Fraser et al. 1998; Ferguson et al. 2003a; Ferguson et al. 2003b; Ferguson et al. 2005; Ferguson 2005; Jamieson et al. 2003; Jamieson et al. 2004a; Jamieson et al. 2004b; Jamieson et al. 2005a; Jamieson et al. 2005b).

Early catchment-scale investigations in this area were initiated to explain continued compliance problems (impairments) following very significant expenditures on sewage treatment. Perhaps simplistically, it had been assumed that emerging effluent treatment technologies such as ultraviolet (UV) disinfection would effectively ‘cure’ impairment problems by removing the bulk of culturable faecal indicators from the effluent stream. When the first UV treatment system installed in Europe failed to guarantee bathing water compliance with Directive 76/160/EU criteria for coliform organisms, the search for non-outfall sources of faecal indicators and the associated ‘integrated catchment studies’ commenced (Wyer et al. 1994; Wyer et al. 1996; Wyer et al. 1997; Wyer et al. 1998; Wyer et al. 1999).

These investigations have produced a series of key observations and findings which can be summarized as follows:

- non-compliance or impairment is most often associated with rainfall events and the associated transport of faecal indicators into the nearshore zone;
- such short-term ‘events’ occupy a small proportion of the harvesting (and/or bathing) ‘season(s)’ but the flux of FIO pollution causing impairment will be delivered in these discrete periods;
- historical archive data describing this condition (the crucial rainfall induced fluxes of pollutants) is often absent, more importantly, historical ‘compliance’ data, collected according to a regular sampling programme will systematically under-represent this condition and, as a consequence, the management utility of such historical archive data is often limited, if not potentially misleading;
most catchment systems have a mix of human (sewage) and animal (agriculture and wildlife) sources of faecal indicator organisms; 
under dry weather conditions, streams transporting diffuse pollution fluxes from livestock in catchment systems exhibit very low FIO concentrations, (faecal coliform concentrations of 104 to 103 per 100 ml); 
even very small and apparently pristine stream waters draining livestock areas, with little or no human sewage inputs, can exhibit FIO concentrations in higher flow conditions similar to those observed in a dilute sewage spilling from a combined sewage overflow (coliform concentrations of 105 to 106 per 100 ml); 
treated sewage effluent will exhibit FIO concentrations determined by the treatment systems and the flow through the treatment plant, but, during low flow conditions, the treated sewage effluents are often the dominant source; 
treated sewage effluent may exhibit very different FIO concentrations following rainfall events, both concentration reductions (due to dilution) and increases (due to increased plant loadings) have been reported (Wyer et al. 1998) and generalizations in this area are inappropriate due to the site specific nature of the sewerage systems installed; and 
where the sewage system is designed to accommodate ‘combined’ surface drainage and foul sewage there will generally be some system of overflows from the sewerage system if it becomes full (termed ‘combined sewage overflows’ – CSOs), or from a holding tank used to provide buffering storage before the sewage plant (termed storm tank overflows – STOs). Under event conditions such CSOs and STOs will discharge to rivers or directly to the coast and these may represent a considerable flux of organisms which commonly enter a river or stream during the early part of the high flow event.

Management information on this complex input pattern is required to target appropriate expenditures on point (mainly human) and diffuse (mainly animal) source control strategies. The key management information required is the proportions of the flux derived from all potential inputs during both low and high flow conditions. Clearly, this requires samples to be acquired from streams and the sewerage infrastructure during event conditions which is logistically difficult and requires aseptic hand sampling if the resultant data are to be credible for operational purposes (Anon 2002).

Simple pie charts can represent this source apportionment and Figure 15.2 shows this representation for the Irvine catchment in Scotland, United Kingdom. This represents all inputs to the bathing zone comprising a crude discharge via...
a long sea outfall, CSOs and STOs and diffuse agricultural sources. This catchment is predominantly used for livestock rearing but also contains the settlements of Kilmarnock and Irvine which, at the time of the study, produced a screened primary treated effluent stream from approximately 200 000 persons. Figure 15.3 shows the flux pattern for this catchment represented in hourly FIO delivery over an eight-week period in the bathing season. This pattern suggests that in dry weather conditions, the marine discharge from the sewage treatment works dominates the input but, during wet conditions after rainfall, the total flux (middle plot) increases rapidly and the diffuse component becomes a much larger contributor to the total FIO flux to the bathing zone (lower plot). Interestingly, the CSO component (principally from the Kilmarnock sewerage infrastructure) represents a relatively small component of the total flux and does not dominate even during the brief periods of CSO discharge.

These flux diagrams were produced by intensive sampling of high flow and low flow water and effluent qualities for a period of eight weeks, together with stream and effluent flow monitoring to calculate the hourly flux information. With this base information for a site it is relatively easy to insert alternative concentrations for different sewage treatment options. Characteristic concentrations for such effluents can be derived from Kay et al. (2008).

The rural pattern seen in the Irvine catchment contrasts with more urban delivery from conurbations. An urban catchment flux study has been completed in the Ribble catchment area in Lancashire, United Kingdom (Wither et al. 2005; Stapleton et al. 2008). Pie charts, hourly flux plots and a bar chart representation for an eight week summer period in this catchment are shown in Figures 15.4, 15.5 and 15.6. This urbanised area also exhibits a rapid increase in the diffuse source, catchment-derived ‘River Ribble’ contribution following rainfall events. However, the bar chart in Figure 15.6 clearly suggests that the majority of high flow inputs to the estuary are in fact attributable to CSOs and STOs. This was certainly a counter-intuitive finding but clearly the balance of the various inputs is important management information if expenditures are to be appropriately targeted between, for example, (i) storage to limit CSO discharges weighed against (ii) disinfection of treated sewage effluents and (iii) implementation of pollution control through farm-scale ‘best management practices’ (BMPs).

15.2.2.1 Using satellite data for catchment delivery modelling

The approach described below has employed catchment models of faecal indicator delivery to provide hourly input sequences from riverine and infrastructure sources to act as input variables for nearshore mathematical modelling (Fraser et al. 1998; Kay et al. 2005). The catchment FIO flux models presented
below are based on multiple regression equations used to predict the low and high flow faecal indicator geometric mean concentrations entering the nearshore zone. These are calibrated from empirical data describing low flow and high flow geometric mean faecal indicator organism concentrations (dependent variables) and satellite-derived land use data (predictor variables). These data
have been used in both rural and urban catchments (Crowther et al. 2002; Crowther et al. 2003; Kay et al. 2005) to predict high and low flow FIO geometric mean values which, when combined with hourly discharge volumes, can provide FIO flux estimates.

In the United Kingdom studies, this process has utilized a digital terrain model (DTM), at a cell resolution of 25 m, derived from Ordnance Survey (OS)

Figure 15.3 Hourly flux plot of faecal coliform for Irvine Bay, Scotland, United Kingdom.
digital contour data. This is modified to create a DTM using standard ARC/INFO procedures to fill in anomalous sinks that often occur along the valley floors in the DTM. This ‘filled’ DTM can be used as a basis for flow path analysis and derivation of a raster drainage network. Subcatchment outlet points are positioned on this network as close as possible to the actual sampling points. Standard ARC/INFO routines are then used to derive the topographic watershed boundaries for each subcatchment.

The digital map of land cover, at 25 m resolution, used to date, is derived form of the Centre for Ecology and Hydrology (CEH) 1990 (or 2000) land cover maps. These are generated from remotely-sensed (Landsat) imagery and divide land cover into 17 classes (Table 15.1). Additional digital land cover data for the United Kingdom are available from OS 1:50 000 colour raster maps. A common 25 m resolution cover can be generated based on these maps as described below.

Accurate land cover and associated water quality data from previous detailed field mapping programmes are available for >200 United Kingdom

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**Figure 15.4** Pie charts representing the proportion of different riverine and sewage infrastructure inputs to the Ribble estuary, Lancashire, United Kingdom.
subcatchments in five study areas in England and Wales. The 17 CEH land cover classes are categorized according to the seven principal land use classes attributed during field surveys (Table 15.1). It should be noted, however, that comparison of the CEH land cover data with the field survey data in the 200 subcatchments showed some significant discrepancies, particularly with regards to built-up land, woodland and improved pasture, suggesting inaccuracies in the remotely sensed data.

Figure 15.5 Hourly flux of faecal coliform organisms derived from different riverine and sewage treatment plants impacting on the Ribble estuary, Lancashire, United Kingdom.
These limitations in the satellite land cover data have been addressed as follows. First, maps of built-up land and woodland are generated based on the unique colours used to depict these land use types in the OS 1:50 000 map. The woodland area extracted from the OS 1:50 000 map data was found to correspond very closely with the field survey data from the 100 subcatchments. Three problems were identified in extracting the built-up land: (i) only buildings are identified and not roads, gardens, and similar infrastructure which would conventionally be classified as part of built-up areas; (ii) some public buildings are excluded because they are depicted using different colours; and (iii) lettering is often superimposed on built-up areas, further reducing the area of built-up land extracted. To quantify this under-estimation in one United Kingdom study area (the Ribble catchment; Kay et al. 2005), 25 500 × 500 m squares, which would be conventionally mapped as 100% built-up, were selected from the OS 1:50 000 raster map set for England and Wales. The area of built-up land was extracted as outlined and, on average, was under-represented by a factor of 3.11.

Comparison of the proportion of improved pasture identified in the 100 subcatchments with that derived from the CEH satellite land cover data showed...
Table 15.1 Details of the Centre for Ecology and Hydrology (CEH) 1990 land cover classification (17 classes) and the corresponding land use type to which they have been attributed.

<table>
<thead>
<tr>
<th>CEH class</th>
<th>Description</th>
<th>Land use type to which the CEH class has been attributed</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>Unclassified</td>
<td>Unclassified</td>
</tr>
<tr>
<td>1</td>
<td>Sea, coastal waters and estuaries, inland to first bridging point or barrier</td>
<td>Other</td>
</tr>
<tr>
<td>2</td>
<td>Inland fresh waters and estuarine waters above the first bridging point or barrier</td>
<td>Other</td>
</tr>
<tr>
<td>3</td>
<td>Bare coastal mud, silt, sand shingle and rock, including coastal accretion and erosion features above high water</td>
<td>Other</td>
</tr>
<tr>
<td>4</td>
<td>Intertidal seaweed beds and salt marshes up to normal levels of high water spring tides</td>
<td>Other</td>
</tr>
<tr>
<td>5</td>
<td>Semi-natural, mostly acid, grasslands of dunes, heaths and lowland-upland margins + Montane/hill grasslands, mostly unenclosed nardus/molinia moorland</td>
<td>Rough grazing</td>
</tr>
<tr>
<td>6</td>
<td>Pastures and amenity swards, mown or grazed, to form a turf throughout the growing season + Meadows, verges, low intensity amenity grasslands and semi-natural cropped swards, not maintained as short turf</td>
<td>Improved pasture</td>
</tr>
<tr>
<td>7</td>
<td>Lowland marsh/rough grasslands, mostly uncropped and unmanaged, forming grass and herbaceous communities, of mostly perennial species, with high winter litter content + Ruderal weeds colonising natural and man-made bare ground + Felled forest, with ruderal weeds and rough grass</td>
<td>Rough grazing</td>
</tr>
<tr>
<td>8</td>
<td>Upland, dwarf shrub/grass moorland + Lowland dwarf shrub/grass heathland</td>
<td>Rough grazing</td>
</tr>
<tr>
<td>9</td>
<td>Upland evergreen dwarf shrub-dominated moorland + Lowland evergreen shrub-dominated heathland</td>
<td>Rough grazing</td>
</tr>
<tr>
<td>10</td>
<td>Bracken-dominated herbaceous communities</td>
<td>Rough grazing</td>
</tr>
</tbody>
</table>

(continued)
<table>
<thead>
<tr>
<th>CEH class</th>
<th>Description</th>
<th>Land use type to which the CEH class has been attributeda</th>
</tr>
</thead>
<tbody>
<tr>
<td>11</td>
<td>Deciduous scrub and orchards</td>
<td>Other</td>
</tr>
<tr>
<td></td>
<td>+ Deciduous broadleaved woodland and mixed woodlands</td>
<td></td>
</tr>
<tr>
<td>12</td>
<td>Conifer and broadleaved evergreen trees</td>
<td>Woodland</td>
</tr>
<tr>
<td>13</td>
<td>Lowland herbaceous wetlands with permanent or temporary standing water</td>
<td>Rough grazing</td>
</tr>
<tr>
<td></td>
<td>+ Lowland herbaceous wetlands with permanent or temporary standing water</td>
<td></td>
</tr>
<tr>
<td>14</td>
<td>Arable and other seasonally or temporarily bare ground</td>
<td>Arable</td>
</tr>
<tr>
<td>15</td>
<td>Suburban and rural developed land comprising buildings and/or roads but with some cover of permanent vegetation</td>
<td>Built-up</td>
</tr>
<tr>
<td>16</td>
<td>Industrial, urban and any other developments lacking permanent vegetation</td>
<td>Built-up</td>
</tr>
<tr>
<td>17</td>
<td>Ground bare of vegetation, surfaced with ‘natural’ materials</td>
<td>Other</td>
</tr>
</tbody>
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a Based on detailed notes that accompany the classification scheme.
a strong linear relationship. However, improved pasture is under-represented where a high proportion is present and over-represented where very little is present. The final map used to derive subcatchment land use needed to drive the FIO flux models requires percentage areas of improved pasture, rough grazing, arable and ‘other’ land use categories and this element is generated based on the CEH land cover data. However, given the uncertainties listed above, the built-up and woodland categories in the CEH data source are reclassified as ‘unclassified’ at this stage. This map is then amalgamated with the built-up and woodland areas extracted from the OS 1:50 000 map, with the latter map categories being given precedence (thus a cell of improved pasture on the CEH map classified as woodland on the OS 1:50 000 map would be classified as woodland in the final data set).

The resultant areas of each land use type in each subcatchment are then adjusted. First, unclassified land is re-allocated to the improved pasture, rough grazing, arable and ‘other’ categories in proportion to the area of these land use types identified within each subcatchment (i.e. no adjustments are made to the areas of built-up and woodland categories). Second, the area of built-up land is increased by the factor of 3.11, the area of land required for this being subtracted proportionately from the areas of improved pasture, rough grazing, arable and other categories.

In addition, land areas upstream of the outlets of all identifiable lakes and reservoirs (from the OS 1:50 000 maps) is defined. Land use within these areas is reclassified as ‘reservoir catchment’. This additional classification attempts to account for low faecal indicator organism concentrations that would be associated with die-off and sedimentation processes within such water bodies (Kay and McDonald 1980) and the resultant effect of water quality at the subcatchment outlet not reflecting the land cover pattern within the subcatchment. It should be noted that, whilst these procedures produce much more accurate data on the overall proportions of different land use types within each subcatchment, some adjustments made (such as built-up land) are not location specific and cannot therefore be represented on a map.

Outputs of this approach for a recent study in the Ribble catchment, United Kingdom are shown in Figure 15.7 which also includes sewage treatment works in the study area.

15.2.3 Linked catchment and nearshore modelling

The compartmentalization of modelling into terrestrial (catchment) and nearshore (hydrodynamic) is often evident due to the different communities involved. This can, however be counter-productive because both bathing and
shellfish harvesting waters are impacted by terrestrial pollutant fluxes from the land surface. Previous modelling in this area has suffered from a lack of understanding of (i) the nearshore (i.e. shallow water) hydrodynamic and microbial fate and transport processes and (ii) the highly dynamic FIO fluxes derived from terrestrial point and diffuse sources. It could be argued that linked catchment and near shore models are required to facilitate accurate prediction of

Figure 15.7 Satellite derived land use data for the Ribble catchment, Lancashire, United Kingdom. Source: Wyer et al. (2003).
FIO concentrations at relevant compliance locations and to drive appropriate remediation strategies which will increasingly have to address issues of altered farming practice and their likely impacts on compliance (Kashefipour et al. 2006; Kay et al. 2007).

Linked catchment and nearshore modelling has been undertaken in three United Kingdom study sites with the aim of predicting recreational water quality. These are: the Ayrshire coast (Wyer et al. 2001; Kashefipour et al. 2006), the Severn estuary (Stapleton et al. 2004; Wyer et al. 2007; Yang et al. 2007) and Carmarthen Bay (Wyer et al. 2004).

These investigations have clarified the interaction between event driven water quality changes on microbial dynamics in riverine and nearshore waters, particularly the role of turbidity derived from entrained riverine sediments on faecal indicator survival, and the potential for real-time decay functions in hydrodynamic nearshore models for prediction of faecal indicators in the compliance zone for bathing and shellfish harvesting waters (Sinton et al. 2002; Kay et al. 2005; Wilkinson et al. 2006).

There has, to date been less attention to waters often favoured for shellfish cultivation such as sea lochs and inlets where tidal water exchange may be less pronounced and the hydrodynamic modelling challenges are significant.

15.2.4 Remediation of faecal indicator fluxes from coastal catchments

There have been very few long-term empirical studies which have sought to quantify the remediation potential of the principal policy levers available to reduce microbial pollutant fluxes (Shreeram and Mostaghimi 2002). Undertaking such assessments is complicated by: (i) seasonality in faecal indicator flux, with temperate livestock rearing areas exhibiting a summer peak in high flow stream water concentrations (Rodgers et al. 2003); (ii) poor information on the likely time taken for measures designed to reduce microbial flux to become effective at the catchment scale.

In one of the rare longitudinal (before and after) catchment-scale studies designed to quantify the effect of a BMP (in this case cattle exclusion from catchment streams) on FIO flux from a 56.7 ha drainage basin, Line (2003) reported data derived from a 7.5 years sampling period which suggested 65.9% and 57.0% reductions in faecal coliform and enterococci export respectively. They also reported that the provision of an alternate water supply without fencing was not effective in producing FIO reduction (see also Shreeram and Mostaghimi 2002). In a two-year United Kingdom longitudinal investigation of
FIO export through a period of de-stocking due to an outbreak of foot and mouth disease, Chalmers et al. (2005) and Sanders et al. (2005) reported a surprisingly slow improvement in water quality following the most drastic BMP of >95% stock removal from the 254.6 ha Caldew catchment in Cumbria, United Kingdom. A longitudinal study at Brighouse Bay in Scotland, United Kingdom, examined the effects of BMPs on water quality in catchment streams and at an adjacent bathing water beach. The principal BMP was stream bank fencing to create a riparian buffer strip (RBS) with associated provision of drinking troughs. Farm dirty water containment was also implemented. The stream water quality data suggested extreme seasonality with the summer period having markedly higher FIO concentrations in catchment streams. However, comparison with an unmodified adjacent control catchment suggested a 66% reduction in E. coli summer high flow export coefficient (in cfu.m$^{-2}$.hr$^{-1}$) with a parallel 81% reduction in intestinal enterococci export. Detailed monitoring through a rainfall event in the post-remediation period suggested that even this improvement would be insufficient to guarantee bathing water compliance with Directive 160/76/EEC (Dickson et al. 2005). The separate effects of RBS and steading dirty water control have been addressed in a longitudinal study of 60 monitored catchments in Scotland by Kay et al. (2005). Here, significant improvements were recorded in FIO flux when compared to ‘control’ catchments but a relatively high intensity of ‘measures’ was required (>30% of stream bank length protected by RBSs).

Bacterial source tracking has been employed by Hyer and Moyer (2004) to inform TMDL studies in the USA and Pond et al. (2004) provide an excellent overview of the potential for the source tracking methods currently available to contribute to FIO flux source apportionment. These methods use either (i) species and or sub-species of organisms thought to be associated with faecal matter from humans or defined animal groups or (ii) chemical markers indicative of human sewage. There is currently no single and definitive approach with which to identify exact proportions of human and animal derived FIOs, but this area is developing rapidly and may provide operationally useful data in the medium term. However, parallel testing of source tracking, where traditional source flux apportionment data are available, suggests that the essentially qualitative tracking information does not provide additional explanatory power (Stapleton et al. 2007).

15.3 CONCLUSIONS

A series of related policy and public health agendas are emerging in the general area of ‘catchment microbial dynamics’ (Kay et al. 2007) which is a relatively
immature field when compared to other water quality modelling areas such as nutrient flux assessment. This is of direct relevance to both shellfish and bathing waters. Effort, at the EU scale, is evident to develop integrated modelling strategies able to address the needs of WFD implementation (Moore and Tindall 2005) and scientists in the USA are similarly engaged through modelling platforms such as BASINS (Tong and Chen 2002). However, operationally useful, fully white box, deterministic and process based faecal indicator models able to predict the effects of individual remedial ‘programmes of measures’ or BMPs on catchment scale FIO fluxes simply do not exist at the present time.

Science activity is developing to address this emerging agenda but some fundamental questions remain such as:

i. How long do the FIOs live in river water under different flow and turbidity conditions? This is a vital question if this highly non-conservative parameter is to be modelled at a catchment scale. Recent developments in nearshore waters have employed real-time decay rates predicted by light intensity and turbidity and a similar approach for riverine matrices is needed (Kay et al. 2005).

ii. What are appropriate FIO export coefficients for different land use types to use in diffuse source models, how do such coefficients vary with season in different farming areas (Kay et al. 2008)?

iii. What are the likely reductions in FIO flux achievable through implementation of feasible land management interventions (BMPs) and will such interventions produce compliance of shellfish harvesting areas and bathing waters with existing and future standards?

iv. How do we balance the different interventions available to reduce faecal indicator fluxes from the sewerage network, principally disinfection of treated effluents against additional storage to reduce spills from CSOs and STOs; such deliberations often depend on sewer modelling studies but the poor precision of volumetric estimates from such models is often insufficiently based on transparent empirical data to facilitate robust assessment of model reliability.

v. To what extent are currently available commercial nearshore hydrodynamic models capable of real-time prediction? This question is being raised as the traditional engineering consultancies which have familiarity with hydrodynamic water quality modelling address the emerging agenda of real-time water quality management. The modelling tools available may, however, lack precision in shallow water environments and, more importantly, peer reviewed data to underpin assessment of key evaluation criteria such as the model explained variance or $R^2$ term.
15.4 REFERENCES


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