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Discussions

Developing a nutrient pollution model to assist policy makers by using a meso-scale Minimum Information Requirement (MIR) approach

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Abstract

A model for simulating runoff pathways and water quality fluxes has been developed using the Minimum Information (MIR) approach. The model, the Catchment Runoff Attenuation Tool (CRAFT) is applicable to meso-scale catchments which focusses primarily on hydrological pathways that mobilise nutrients. Hence CRAFT can be used investigate the impact of management intervention strategies designed to reduce the loads of nutrients into receiving watercourses. The model can help policy makers, for example in Europe, meet water quality targets and consider methods to obtain "good" ecological status.

A case study of the 414 km^2 Frome catchment, Dorset UK, has been described here as an application of the CRAFT model. The model was primarily calibrated on ten years of weekly data to reproduce the observed flows and nutrient (nitrate nitrogen – N – and phosphorus – P) concentrations. Also data from two years of sub-daily high resolution monitoring at the same site were also analysed. These data highlighted some additional signals in the nutrient flux, particularly of soluble reactive phosphorus, which were not observable in the weekly data. This analysis has prompted the choice of using a daily timestep for this meso-scale modelling study as the minimum information requirement. A management intervention scenario was also run to show how the model can support catchment managers to investigate how reducing the concentrations of N and P in the various flow pathways. This scale appropriate modelling tool can help policy makers consider a range of strategies to to meet the European Union (EU) water quality targets for this type of catchment.

1 Introduction

The meso-scale is classed as catchments that vary between 10–1000 km^2 (Blöschl, 1996). Uhlenbrook et al. (2004), states "The satisfactory modelling of hydrological processes in meso-scale basins is essential for optimal protection and management of water resources at this scale". It is therefore important that government policies on pollution abatement must be implemented at this scale. The EU Water Framework Directive (WFD) (European Parliament, 2000) has increasingly required catchments to meet in-stream standards in order to obtain "Good" ecological status. Therefore, all surface water bodies must meet exacting water quality and ecological targets (Withers and Lord, 2002). Hence we require a framework that helps inform policy makers and regulators to understand the source of nutrient pollution at the scale of their interest. Numerous models have been developed to simulate water and nutrient fluxes at a catchment scale (e.g. INCA, Wade et al., 2002, 2006; PSYCHIC, Davison et al., 2008; SWAT, Arnold, 1995). INCA has been used to investigate compliance issues with the WFD in terms of water quality (Whitehead et al., 2013). These models have been used to underpin policy decisions and feed into the decision making processes with regards to the land use in catchments, and assess the impacts of any changes to this including source control or modified agricultural practices (Whitehead et al., 2013). However, these models tend to be too complex for average end users to use and the simulations are prone to high uncertainty (Dean et al., 2009; McIntyre et al., 2005). Conversely export coefficients (Johnes, 1996; Hanrahan et al., 2001) can be an over simplification of reality and omit the role of event driven nutrient losses.

Is there are a scale appropriate methodology that can simplify the models and still reflect the dominant observed processes observed in research studies? Can the models be more transparent with regards to the processes that are simulated and hence how they can be managed? It is vital that user friendly models should aid policy makers when considering the likely consequences of their decisions. This study will show that this model must include sufficient processes to reflect nutrient losses from the catchment which must be based primarily on soil and hillslope processes: such as overland flow; subsurface soil flow and slower groundwater dynamics. Hence the model must represent both chronic nutrient losses (seasonal fluxes), and acute losses (storm driven fluxes) (these terms were defined by Jordan et al., 2007). To this end we have developed an MIR (Minimum Information Required) modelling approach (Quinn et al., 1999; Quinn, 2004) which: uses the simplest model structure; that achieves the current modelling goals; that uses process based parameters that are physically interpretable to the users and the impact of any parameter change can be clearly interpreted by the end user. The CRAFT (Catchment Runoff Attenuation Flux Tool) has been developed to address these goals. Thus the methodology focusses less on the spatial pattern of land use given the mixing effects of meso-scale aggregation and homogenisation.

We are also living in a new era of high resolution datasets. These datasets may become invaluable to research-scale studies but at the meso-scale such detail may be less useful. More data are becoming available from high resolution monitoring using newly developed auto-analysers and sondes (for example: Owen et al., 2012; Bowes et al., 2012; Wade et al., 2012; Cassidy and Jordan 2011), and from use of high-frequency samplers (Bowes et al., 2009a, Evans and Johnes, 2004). This study will attempt to show that high-frequency data sets at this scale can help to justify the choice of a simpler MIR model. A case study will be shown that includes a sub-daily and weekly time series, collected at the River Frome catchment in the Dorset (Bowes et al., 2011; Marsh and Hannaford, 2008). The CRAFT model will demonstrate the simulation of flow (Q), nitrate (NO_3), total phosphorus (TP) and soluble reactive phosphorus (SRP) with a daily time step. Data analysis will aim to show that for this meso-scale dataset, the representation of the dominant hillslope processes and hydrological flow pathways is more important than the spatial distribution of parameters (Quinn, 2004). Equally, it will be shown that the hydrological flow pathways need representation rather than the detail of the nutrient cycling processes.

1.1 The MIR modelling methodology

The MIR approach was developed partly as a response to a perceived excessive number of parameters in the established water quality and sediment transport models (Quinn et al., 1999; Quinn, 2004), and partly to address the issue of excessive model complexity to end user needs. The principles of MIR models are based on how much information can be gained from localised and experimental studies on nutrient loss, so that the most pertinent process components can be retained in the model and be easily manipulated and assessed by an end user.

MIR models must be suitable for use in the decision-making process in order to become a valuable tool. The use of such an approach leads to the following research questions:

1. How complicated does a MIR model need to be in order to address catchment management issues?

2. How important is it that the MIR model represents how nutrients are lost from the catchment, through the dominant hydrological pathways?

3. Does the model reflect the importance of acute losses of nutrients from the catchment during storm events and chronic losses during inter-event periods (and also any non-agricultural component)?

The MIR approach provides a method for the evaluation of existing models and the selection of key generation and transport processes (e.g. nitrate leaching, Quinn et al., 1999 and sediment-attached P entrainment parameters, Quinn et al., 2008). The modelling of runoff is also kept as simple as possible to avoid excessive computation, although key runoff processes that influence nutrient and sediment loads are retained. By creating a meta model of more complex process based models, a minimum number of processes are retained in the model structure that are required to satisfy a model goal: in this case the simulation of meso-catchment scale diffuse pollution. A series of simple equations are implemented in MIR models with a parsimonious number of parameters. The TOPCAT MIR family of models (Quinn, 2004, Quinn et al., 2008) were developed using this approach to simulate various sources of sediments and nutrients. Heathwaite et al. (2003) developed a simple spatial index model for estimating diffuse P losses from arable lands into waterways called the PIT (Phosphorus Indicators Tool). A series of DSS-based models were developed in Australia: commencing with E2 (Argent et al., 2009), then WaterCAST and finally SourceCatchments (Storr et al., 2011; Bartley et al., 2012). These have similar features of a MIR including: a daily simulation timestep to predict sediment and nutrient concentrations (C); and fluxes (i.e. $C \times$ daily flow); containing only two flow and nutrient pathways termed "event" i.e. storm flow, and "dry weather" i.e. baseflow, both assigned fixed C values for each sediment and nutrient simulated. This prioritisation of processes is the basis of MIR modelling.

1.2 Models as catchment management tools

It is important that models are seen as useful in terms of the decision making process and its relationship to land use through a feedback mechanism between the regulators and the land owners (e.g. farmers) or holders of discharge consents into receiving watercourses (e.g. water companies) (Whitehead et al., 2013). Hence, there is a need to re-interpret broad scale planning decisions and assess their likely impact on a single farmer or farming community. The key research question arising from this process relates to how large scale catchment management decisions impact nutrient concentrations and fluxes at the scale of assessment. The model can highlight any potential problems such as changes in nutrient form, known as pollution swapping (Stephens and Quinton, 2009). In this study *pollution swapping* could show for example that SRP increases due to the mitigation measures that have reduced the concentration (and loads) of particulate P.

In this particular study we assess whether a particular water body is likely to become compliant within key regulations such as the WFD, although any other water quality standards could be used. CRAFT is meant to be just one of many tools that can be used to aid the planning process and address several catchment management issues. If the aim of the modelling study is to determine a total export of nutrients from the catchment outlet then simulating all the processes within the catchment may not be required and an export coefficient model (e.g. Johnes, 1996; Hanrahan et al., 2001) may be useful. However, the provenances of the fluxes still need to be linked back to local sources, pathway and nutrient loading factors. Hence the "when" and the how' of nutrient losses seem to be key to the management aspects. Localised applications of the MIR can help address “where” nutrient arise from or could compliment a spatial index tool such as the PIT model (Heathwaite et al., 2003).

The goal here is to develop a model that contains a useful and parsimonious set of parameters resulting in a “visual thinking tool” that can provide a semi-quantitative risk-based assessment of management decisions. The CRAFT model described below is written in a MS Excel spreadsheet and the results, graphs and load calculations update instantaneously; hence the consequence of changing the parameters on all the outputs can be seen immediately: e.g. runoff and nutrient load. Instead of expecting the end user to perform an explicit uncertainty analysis, they are encouraged to investigate the sensitivity of the output fluxes to a wide range of parameter choices and “see” what these actions actually mean in terms of likely land management policies. The onus is thus still on the user to think through the meaning of the parameters and the implications of changing their values.

1.3 The spatial and temporal scales of the data

High-frequency water quality monitoring has become achievable over the last decade, firstly with the availability of automatic water samplers (Bowes et al., 2009a) enabling several measurements per day to be taken (e.g. sub-daily measurements of concentrations of sediments and nutrients). Recent examples of long term monitoring at a high temporal frequency include the DTC (Demonstration Test Catchments www.edendtc.org.uk) study in the UK, based in the Eden catchment in Cumbria (Owen et al., 2012), the monitoring in the Blackwater catchment, Ireland (Cassidy and Jordan, 2011), and the Irish Agricultural Catchments project (www.teagasc.ie/agcatchments), and the monitoring of the Enborne and Kennet subcatchments of the Thames by Wade et al. (2012). These studies were made possible by the development of bankside nutrient auto-analysers (Jordan et al., 2007) which have allowed very high-frequency (hourly/sub-hourly) data sets to be assembled. These data have enabled better estimation of nutrient export from catchments to be made for the first time (Bowes et al., 2009a; Johnes, 2007). The growth of these data sets allows us to pose an additional

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research question as to what is the value of collecting high-frequency data to parameterize models at the medium-large catchment scale (100–500 km^2). However these high frequency measurements may be prone to localised noise can introduce errors to the observations (Bowes et al., 2009a). Unravelling trends, seasonality and noise may require signal processing techniques to extract meaningful time series data and perform trend analysis (e.g. Kirchner and Neal, 2013).

The modelling process seeks to link science and process knowledge gained at the local "research scale" (1 m^2–10 km^2) with a larger (meso-scale) catchment (100–500 km^2) "applied science" scale (Haygarth et al., 2005). Hence, the astute choice of model structure and timestep allow a scale appropriate MIR model to be set up.

At larger catchment scales mixing processes may dominate the final observations at the outlet, and the choice of sampling frequency will still be important if load estimates are required (Johnes, 2007). The temporal fluctuations in runoff and water quality observed in headwater research catchments may not necessarily be observed at the outlet of the larger catchment area (Haygarth et al., 2005, 2012; Storr et al., 2011). As a rule therefore, the smaller the catchment the more detail is required in the model to define processes, but as the catchment size increases then in-stream processes associated with channel routing and the effect of point sources (especially of P) will tend to take over from nutrient generation processes in influencing the signal observed at the outlet of a larger catchment (Haygarth et al., 2005, 2012).

2 Methods

The MIR structure of the new CRAFT model described below lends itself to catchment applications as the simple model structure to satisfy the project goals. MIR allows the modifications to be made to the model to improve simulations by adding or removing processes as required.

2.1 Model description

The structure of the CRAFT model is shown in the upper pane of Fig. 1 comprising three dynamic storages and the associated flow and transport pathways, representing a MIR representation of a more complex hydrological system. The lower pane shows the flow and nutrient transport pathways that exist in a catchment such as the Frome using a conceptual hillslope cross-section. Here, inputs and outputs of N and P in the catchment are shown diagrammatically. There are three flow pathways shown: (i) an overland flow component which also represents processes in the cultivated near surface layer (down to several centimetres depth); (ii) a faster subsurface component encapsulating agricultural soils that may have been degraded by anthropogenic activities and perhaps enhanced flow connectivity (e.g. through field drains); (iii) a slower groundwater component encapsulating any background flow in the catchment due deeper flow pathways and also to Wastewater Treatment Plants (WWTP)s and other constant, non-rain-related discharges. We will refer below to the pathways as: (i) overland flow (OF); (ii) as fast subsurface soil flow (SS); and (iii) as the slow, deeper groundwater flow pathway (DG) respectively.

There are six parameters that require estimation or calibration to control the flow pathways. The values are shown in Table 2 below.

The uppermost surface and cultivation store (SCS) is conceptualized to permit both crop management runoff connectivity options to be examined. The SCS store is split into two halves with the upper half representing a cultivation (tillage) layer that generates overland flow, and the lower half accounting for controlling ET and the drainage control rate to the lower stores. Firstly, a water balance updates the storage (SS) and then computes the overland flow from the surface store (QOF) through the following equations, where R is rainfall, D drainage to the lower half of the store. Note that all stores are in units of length (e.g. m) and all flux rates (e.g. R, D, QOF) are in units of length per time step (e.g. m day^{-1})

$$\mathrm{SS}\,(t) = \mathrm{SS}\,(t-1) + R(t) - \mathrm{QOF}\,(t-1) - D(t-1) - \mathrm{QCSR}\,(t) \tag{1}$$

$$\mathrm{QOF}\,(t) = (\mathrm{SS}\,(t) - D(t)) \cdot \mathrm{QUICK}. \tag{2}$$

The drainage rate (D) is calculated as the smallest of: (i) KSURF, the maximum drainage rate per time step; and (ii) the current storage SS(t), in the upper part of the store. Therefore, the user can force the model to hold more water in the store by specifying a small value of KSURF, which will generate more overland flow according to Eq. (2). A large value of KSURF will cause the store to drain out in one time step and increase the drainage rate to the subsurface stores and reducing the overland flow. Thus the parameter can be used to deliberately partition excess water between surface and subsurface flows which is crucial for investigating connectivity options and possible pollution swapping effects.

$$D(t) = \mathrm{Min}\,(\mathrm{KSURF}, \mathrm{SS}(t)) \tag{3}$$

The lower half of the SCS represents the soil layer (below the cultivated layer) and also accounts for ET in the model. The parameter limiting the size of the store is called SRZMAX. The storage of water in the store (SRZ) at each time step is updated by the following mass balance:

$$\mathrm{SRZ}\,(t) = \mathrm{SRZ}\,(t-1) + D(t) - \mathrm{ET}(t) - \mathrm{PERC}\,(t). \tag{4}$$

Any excess water present in the store above SRZMAX will form percolation (PERC) which then cascades into the subsurface DS and DG stores:

$$\mathrm{PERC}\,(t) = \mathrm{MAX}\,(0, (\mathrm{SRZ}\,(t) - \mathrm{SRZMAX})). \tag{5}$$

Both the SS and DG stores are dynamically time varying and generate fast (QSS) and slow groundwater flows to the outlet (QGW) respectively. A dimensionless parameter

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SPLIT (0,1) apportions active drainage from the lower surface store towards either store, i.e. a water balance for the storage (SSS) in the SS store can be written as

$$\mathrm{SSS}\,(t) = \mathrm{SSS}\,(t-1) - \mathrm{QSS}\,(t-1) + \mathrm{PERC}\,(t)\cdot \mathrm{SPLIT}. \tag{6}$$

The equation for the storage in the DG store (SGW) is identical except that (1 – SPLIT) is substituted for SPLIT.

The flow (QSUB) from either subsurface store is described by Eq. (7) where K is a recession rate constant (d^{-1}) and S is the storage (in m). Therefore QSUB at time t, is given by

$$\mathrm{QSUB}\,(t) = K\cdot S(t-1). \tag{7}$$

In the DG store the initial storage SGW0 is set by the user by specifying an initial value of the resulting flow (QGW0, where we are using the suffix "GW0" to denote initial value of slow groundwater flow) rather than explicitly defining the storage (which is difficult to estimate in a complex catchment). It is convenient to commence the model simulation during a dry spell, where the slow groundwater component is usually relatively constant and most of the runoff consists of this flow. Therefore, rearranging Eq. (7) to invert its terms gives

$$\mathrm{SGW0} = \frac{\mathrm{QGW0}}{\mathrm{KGW}} \tag{8}$$

where QGW0 ≡ Observed runoff on first day of simulation ($\mathrm{m\,d}^{-1}$), following the assumption above.

Lastly, the total modelled runoff at each timestep, at the outlet is calculated (QMOD)

$$\mathrm{QMOD} = \mathrm{QOF} + \mathrm{QSS} + \mathrm{QGW}. \tag{9}$$

The user must now add a sensible range of input nutrient levels to the model and it is assumed the user has some nutrient pollution knowledge. The "informed" user is

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encouraged to set and alter these values and see the impact instantaneously. The nutrients are all conservative in the inputs and the losses and we require user to understand the link between land use management and the level of nutrient loading.

In general nutrients are modelled in the CRAFT by either a constant concentration assigned to each flow pathway or by using an uptake factor (or "rating curve") approach (e.g. Cassidy and Jordan, 2011; Krueger et al., 2009), where the concentration is directly proportional to the flow rate. A conceptual model of the flow and transport pathways in the catchment that are incorporated in the CRAFT is shown in the lower part of Fig. 1.

In the uptake factor approach, the concentration (units $\mathrm{mg\,L^{-1}}$) of a nutrient (N) in a flow pathway, in this example in overland flow (COF) is a function of QOF and given by

$$\mathrm{COF}\ (N) = \mathrm{MAX}\ (\mathrm{COFMIN}\ (N) + K(N) \cdot \mathrm{QOF}, \mathrm{COFMIN}\ (N)) \tag{10}$$

where: QOF is the overland flow; $K(N)$ represents the slope of the relationship between flow and nutrient concentration in the observed data (i.e. uptake factor) and COFMIN (N) is the minimum concentration. This is included in Eq. (10) to prevent unrealistically low concentrations being used in the model during low flow periods, and has a physical basis in that it is equivalent to the y-intercept of a C–Q plot for the nutrient in question. Krueger et al. (2009) used this type of equation to model TP concentrations in high flows generated by enrichment of sediment with P.

The daily nutrient load is calculated by the mixing model described by Eq. (11), where $L(N)$ is the load, CSS and CGW are the constant concentrations in the dynamic soil and dynamic groundwater zones respectively

$$L(N) = \mathrm{COF}(N) \cdot \mathrm{QOF} + \mathrm{CSS}\ (N) \cdot \mathrm{QSS} + \mathrm{CGW}(N) \cdot \mathrm{QGW}. \tag{11}$$

The concentration of the nutrient in the catchment outflow ($C(N)$) can be calculated directly from $L(N)$ using Eq. (12)

$$C(N) = \frac{L(N)}{\mathrm{QMOD}}. \tag{12}$$

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Nitrate and SRP concentrations are calculated at each timestep using Eqs. (11) and (12). The TP concentration is calculated by adding together L(SRP) (calculated using Eq. 11) and L(PP) (particulate P load; calculated by multiplying the concentration calculated by Eq. 10 by QOF), and then dividing the TP load by the modelled (total) flow

$$C(\mathrm{TP}) = \frac{(L(\mathrm{SRP}) + L(\mathrm{PP}))}{\mathrm{QMOD}}. \tag{13}$$

2.2 Case study description

The 414.4 km^2 River Frome catchment (Fig. 2) drains into Poole Harbour with its headwaters in the North Dorset Downs (Bowes et al., 2011; Marsh and Hannaford, 2008; Hanrahan et al., 2001). Nearly 50 % of the catchment area is underlain by permeable Chalk bedrock, the remainder consists of sedimentary formations such as tertiary deposits along the valleys of the principal watercourses (including sand, clay and gravels). There are some areas of clay soils in the lower portion of the catchment. However, most of the soils overlaying the chalk bedrock are shallow and well drained. The land use breakdown is dominated by improved grassland (ca. 37 %, comprising hay meadows, areas grazed by livestock and areas cut for garden turf production), and ca. 47 % tilled (i.e. arable crops primarily cereals) usage (Hanrahan et al., 2001).

The mean annual catchment rainfall from 1965 to 2005 was 1020 mm and mean runoff 487 mm (Marsh and Hannaford, 2008). The major urban area in the catchment is the town of Dorchester (2006 population over 26 000, Bowes et al., 2009b) otherwise the catchment is predominantly rural in nature. At East Stoke the UK Environment Agency (EA) has recorded flows since 1965. The Centre for Ecology and Hydrology (CEH) and Freshwater Biological Association have collected water quality samples at this same location at a weekly interval from 1965 until 2009 (Fig. 2) (Bowes et al., 2011). Hanrahan et al. (2001) presented both export coefficients for diffuse sources of TP, and load estimates for diffuse and point sources (comprising: WWTPs (serving Dorchester plus other towns); septic systems; and animal wastes). The total annual

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TP (total phosphorus) export from diffuse sources in the catchment was estimated to be 16.4 t P yr^{-1}, a yield of 0.4 kg P ha^{-1} yr^{-1}. Point source loads from WWTPs, septic systems and animals added an extra 11.5 t P yr^{-1} (from the data in Table 2 in Hanrahan et al., 2001) to the catchment export, giving a total load of 27.9 t P yr^{-1}. Nitrogen (as nitrate) export from the catchment in the mid-1980s was estimated by Casey et al. (1993) to be 21.6 kg N ha^{-1} yr^{-1}, with 7 % of this originating from point sources in the catchment. Based on a long timeseries of nitrate concentrations also collected at East Stoke (Bowes et al., 2011), the N load probably increased to a maximum during the 1990s and stabilised during the following decade.

A report by the Environment Agency from their "Making Information available for Integrated Catchment Management" project (EA, 2007) provided spatial predictions of N in addition to diffuse P and sediment yield, on a 1km grid covering the entire catchment using the models: PSYCHIC (for P) and NEAPN (for N; EA, 2007). Based on these predictions, N export varied from 0–63.4 kg N ha^{-1} yr^{-1} (of a similar order of magnitude to the figure of 20.2 kg N ha^{-1} yr^{-1} TON (Total Oxidisable N) from high resolution monitoring data estimated by Bowes et al. (2009a)) and TP export varied from 0–2 kg P ha^{-1} yr^{-1}, which is lower than the range of TP export coefficients quoted in Hanrahan et al. (2001) for their baseline land use and management scenario, probably due to improvements in phosphorus treatment at the Dorchester WWTP in 2002 (Bowes et al., 2009b).

2.2.1 Hydrological data

Forcing data (precipitation) was supplied by the EA for the period 1997 to 2006 which was therefore chosen as the modelling period. Daily mean flow was also provided from East Stoke gauging station for the same time period. Potential Evapotranspiration (PET) was derived using an algorithm developed to estimate daily PET based on monthly temperature patterns, to estimate a daily PET which when totalled for the year would match the known annual PET (465 mm yr^{-1}).

Daily rain gauge data was obtained from Kingston Maurwood (ST718912) located ca. 4 km downstream of Dorchester. Earlier studies have noted some spatial variation in precipitation across the catchment (Bowes et al., 2011), and Smith et al. (2010) reported that between 1993 and 2008 there were 3–5 gauges operational in the catchment. Therefore, model errors sourced from rainfall are likely to be significant and may influence predictions of overland flow (where rainfall is an important factor) and the associated nutrient transport by this pathway.

2.2.2 Monitoring datasets

Two sets of water quality monitoring data were used in this study (Table 1 below shows the statistics relating to long term concentrations) along with daily flows recorded by the Environment Agency at East Stoke gauging station. The data were compared and analysed so that the MIR model could be defined.

1. The CEH/Freshwater Biological Association long-term dataset (LTD) of water quality for the River Frome (Bowes et al., 2011; Casey, 1975; open access via gateway.ceh.ac.uk) was collected from 1965 to 2009 at a near-continuous weekly interval (average number of observations per year = 48) and thus represents one of the longest (relatively) high frequency datasets on water quality in existence from the UK. In this study we analysed their nitrate-N (nitrate) from 1997 to 2006, and their TP and SRP data between 1997 and 2002. After March 2002 the introduction of P-stripping measures at Dorchester WWTP reduced SRP loads by up to 40 %, according to Bowes et al. (2009b, 2011), which produced a step reduction in stream SRP concentrations. The statistics for the periods of analyses are shown in Table 1.

2. A high frequency data set (HFD) described in Bowes et al. (2009a), was also collected at East Stoke between 1 February 2005 and 31 January 2006, using a stratified sampling approach and EPIC™ water samplers (Salford, UK). The statistics related to nutrient concentrations are shown in Table 1. The frequency

of the water samples varied between two to four times daily during dry periods with up to eight samples per day during rainfall events. The average number of samples was 3.7 per day. Also in the dataset were river flow (Q) values taken from the Environment Agency 15 min interval flow data. In this study we used the Q, TON, TP and SRP data. A more detailed discussion of the two datasets follows in order to justify several MIR simplification assumptions.

Firstly, the flow timeseries of the LTD (daily mean flows; DMF) and HFD (sub-daily) flows were compared over the course of the high resolution monitoring period described in Bowes et al. (2009a) and both time series of flows are shown in Fig. 3a. For most of the period both sets of flows closely matched ($\rho = 0.98$) except perhaps during runoff events of less than a day where the HFD flows were sometimes higher. The analysis suggests that for modelling purposes and load estimation that a daily timestep is probably sufficiently short to capture the variability in the observed data without the need to use hourly forcing data to enable a sub-daily timestep to be used.

For nitrate it is assumed that nitrite concentrations were negligible in the LTD dataset (Bowes et al., 2011) so that TON concentrations (equivalent to nitrate plus nitrite) were effectively equal to nitrate. This allows the HFD TON data to be directly compared against the observed (weekly LTD) nitrate data. The patterns observed graphically in the weekly and high resolution nitrate/TON timeseries were very similar indicating that the weekly monitoring data were probably sufficient to estimate the range of nitrate/TON concentrations in the catchment, in order to assess compliance with EU WFD quality standards (in this case ensuring that $C \leq 11.9\,\mathrm{mg\,L^{-1}}$ N). The monitored periods overlapped (Fig. 3b) and there were a few spikes in the HFD above concentrations measured by the LTD with those measured during recession spells in the flows, generally less than $1\,\mathrm{mg\,L^{-1}}$ N. The correlation between C and Q was weak (in the HFD $\rho = 0.12$), due to the complex SRP concentration/flow relationships caused by point source dilutions at low flows and increasing diffuse inputs at higher flows (Bowes 2009b). Therefore, it would not be possible to develop a Q vs. C rating curve to estimate loads from this dataset using the methods used by Cassidy and Jordan (2011). There was also no evidence that high flows would generate correspondingly high nitrate concentrations. In Fig. 3b a dilution effect can be clearly observed during several events in autumn 2005 (indicated by "1", and the dashed blue line linking the chemograph to the corresponding events in the hydrograph in Fig. 3a), with lower concentrations lasting for several days in some cases during the subsequent period of high baseflow. This indicates that concentrations of nitrate in the combined slower baseflow/sewage effluent must be higher than concentrations in rapid overland flow.

For phosphorus the HFD SRP data were compared visually with the LTD SRP data in Fig. 3c and again the patterns in both datasets were broadly similar, with increasing concentrations during the summer period between May and November 2005. HFD TP concentrations are also shown in Fig. 3c by the red line. Between November 2004 and March 2006 there was a gap in the LTD TP data for operational reasons discussed in Bowes et.al (2011). Flow data from the upper panel (Fig. 3a) will be used to illustrate several key points arising from the HFD data:

1. Spikes in TP concentration for example in February and mid-December 2005 were during the falling limb of the flow hydrograph and were not associated with significant storm runoff events. Corresponding spikes in SRP concentration were not significant at these times. (Examples are indicated by "2" on Fig. 3c.) Some spikes were also observed during the falling limb of the flow hydrograph on several occasions in summer 2005, without corresponding SRP spikes but during a period where SRP concentrations were increasing. This is surprising because rapid overland flow is normally thought to generate correspondingly high PP concentrations in washoff (Haygarth et al., 2012). (Examples are indicated by "3" on Fig. 3c.)

2. Three events between October and December 2005 did generate high concentrations in PP that coincided with the storm peak in the flow hydrograph. This could indicate a faster mobilisation of PP into the channel system during wet conditions in autumn–winter 2005 compared to summer storms. Haygarth et al. (2012) have

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observed similar peaks in PP in smaller headwater catchments due to sheet flow events. (Examples are indicated by “4” on Fig. 3c.)

3. Some SRP concentration spikes were not simultaneously observed in the TP concentrations, these may have been due to WWTP discharges or leaky septic tanks (the high-frequency sampling methods permitted this to be observed). Examples of this are indicated by “5” on Fig. 3c. SRP concentrations during the summer months tended to increase by approximately 0.07 mg L^{-1} P indicating chronic sources of nutrients in the catchment whereas acute sources tended to be associated with runoff events or other events in the catchment not associated with high flows. Bowes et al. (2011) also observed this in the LTD dataset and suggested that the probable cause was a combination of lower flows with less dilution of SRP in the river originating from point sources (WWTPs) in the catchment. Jordan et al. (2007) attributed acute sources of TP in their 5 km^2 agricultural catchment in Northern Ireland to applications of slurry and inorganic P during periods of low rainfall (with no associated runoff events).

2.3 Modelling and calibration

Flow and nutrients were simulated with the CRAFT for a ten year baseline period, 1 January 1997 to 31 December 2006 using a daily timestep. The performance of the calibrated CRAFT model at reproducing observed flows was assessed by a combination of visual inspection of the modelled against observed runoff and the use of the Nash-Sutcliffe Efficiency (NSE) evaluation metric.

The model parameters were assumed to be constant over space and time. A comparison of the model performance at predicting the SRP and TP concentrations was curtailed at the end of February 2002. However, for nitrate the model performance over the full 10 yr period was assessed. The daily timestep was used in the CRAFT for reasons discussed above.

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The hydrological model calibration aimed to maximise the value of the NSE whilst ensuring that the MBE (mass balance error) was less than 10 %. The parameters QUICK, KGW, KSS, SPLIT, SRMAX and KSURF were adjusted iteratively to enable this and create a single "expert" parameter set.

The sensitivity of the model was then assessed by running a Monte Carlo analysis of 100 000 simulations, where the six parameters were randomly sampled from a uniform distribution (the upper and lower bounds are shown in Table 2). The performance metric used to compute a likelihood function (Beven, 2009); the Sum of Square of Errors (SSE) was chosen here, in order to identify which simulations were "behavioural".

Simulations with a MBE greater than 10 % were also rejected. The top 1 % of simulations meeting both criteria were thus chosen as "behavioural" and a normalised likelihood function ($L(Q)_i$) was calculated using Eq. (14) with the SSE values determined above for each simulation i. Lastly, weights were assigned to the behavioural flows based on the likelihood of each simulation. These weighted flows were then used to compute the upper and lower bounds (here the 5th and 95th percentile flows were chosen) applied to the modelled flows (QMOD).

$$L(Q)_i = \frac{\mathrm{SSE}_i}{\Sigma_{\mathrm{SSE}}} \tag{14}$$

The nutrient model parameters were calibrated by assessing the performance of the model against the weekly concentration data in the LTD, using the following metrics to determine an "expert" parameter set:

– Visually comparing the time series of nitrate, SRP and TP against the observed data and adjusting the most sensitive nutrient model parameters to obtain a best fit between modelled and observed time series.

– Optimising the errors between modelled and observed mean and 90th percentile concentrations with the aim of reducing these below 10 % if possible. The mean and 90th percentile concentrations were chosen as these represent the concentrations over the range of flows (mean) and events (90th percentile), and therefore

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allow the model performance under all flow regimes to be assessed. This should be carried out alongside the previous step.

If satisfactory nutrient model outputs were not obtained by adjusting the nutrient parameters in the first step then it was necessary to adjust the hydrology model parameters, particularly QUICK and SPLIT, to increase or decrease the proportions of the different flow pathways.

A further sensitivity analysis was then performed using the flows from the behavioural hydrology simulations (discussed above) and re-running the nutrient model (without adjusting the "expert" parameter values for nutrients) to determine a set of upper and lower bounds (5th and 95th percentile values) to the predicted concentrations and their associated loads ($Q \cdot C$). A full uncertainty analysis investigating the water quality parameters (as performed for P modelling by Dean et al., 2009 and Krueger et al., 2009) was not carried out due to the difficulties in defining "behavioural" water quality models. Please note that the uncertainty analysis has been done for the benefit of this study and we would not expect an end user to perform this task.

2.4 Management intervention scenarios

For a model to be effective at the management level it needs to be able to link back to processes at the local scale. The creators of the model are thus conveying their key findings to catchment managers to inform them of the consequences of local scale changes at the catchment scale. Here the local land use change is assumed to occur at all locations. Nevertheless, the CRAFT model can show the magnitude and proportion of the nutrients lost by each hydrological flow pathway. Equally it is possible to show the concentration of each nutrient at each time step as this helps educate the end user. However, for simplicity, here a combination of land use changes and express the output as the change in export loads for each pathway at the outlet will be shown.

In order to demonstrate the impact of a catchment management intervention strategy, the following changes were made to the catchment as a runoff and nutrient management intervention (MI) scenario: (i) the modelled overland flow was reduced by reducing the value of the QUICK parameter to 0.012, representing a management intervention that remove or disconnects the agricultural pollution "hotspots"; (ii) nutrient loads in the rapid subsurface zone were reduced by reducing the values of CSS(SRP) and CSS(NO_3) by 50 % (i.e. halving the impact of diffuse sources linked to the outlet by this flow pathway) to represent improved land management with reduced fertilizer loads. No change to the DG nitrate concentration was made as firstly, any changes in land management may take decades to be observed in the deeper groundwater (Smith et al., 2010); and secondly, recent improvements to WWTPs have only targeted reducing SRP loads and not nitrate loads (Bowes et al., 2009b, 2011).

(iii) Background loads of SRP in the catchment are reduced by lowering CGW(SRP) to represent the reduction in deeper groundwater concentration caused by both lower leaching rates from the soil store and making further improvements to WWTPs in the catchment to reduce SRP loads. Bowes et al. (2009b) found that a 52 % reduction in the SRP export from point sources had taken place since 2001 in the catchment (up to 70 % of the SRP loads from each improved WWTP is assumed to be stripped out). In terms of the total (point and diffuse) SRP load, Bowes et al. (2011) estimated that between 2000 and mid 2009 it had been reduced by 58 % which was due to further improvements to the smaller WWTPs in the catchment as well as a reduction in diffuse sources of up to 0.1 $kg\,P\,ha^{-1}\,yr^{-1}$.

3 Results

Essentially we can compare the modelled and observed data sets and the core statistics (Table 1) or by visually assessing model performance firstly from the "expert" calibration. Figure 4 shows the time series plots of modelled and observed flow at East Stoke and then the modelled ("expert" calibration) and observed nitrate, TP and SRP concentrations. To further illustrate the model performance in terms of predicting flow and concentrations Fig. 5 shows scatter plots of the modelled against observed values

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with the goodness of fit of the simulations (R^2) shown along with the fitted linear trendline for each predicted variable (flows and concentrations).

3.1 Expert calibration

The hydrology model parameters from the final "expert" calibration are shown in Table 2. The model results from the CRAFT were as follows: The NSE for the baseline hydrology simulation was 0.80; the mass balance error was +1.0 % (over prediction), less than the 10 % limit that is considered acceptable for assessing the model performance as "satisfactory". In the Frome catchment the percentage of overland flow (which includes surface runoff and near-surface runoff through the ploughed layer) according to the calibrated model was very small (2.2 % of the total runoff of 516 mm yr^{-1}). This value may be low but as stressed before it is difficult to see the overland flow signal at the meso-scale. Here, an overland component has been retained due to an assumption that P is being lost via this process i.e. from the knowledge arising from research studies (e.g. Bowes et al., 2009a; Heathwaite et al., 2005; Owen et al., 2012). Values for the parameters KSR(PP) and KSR(SRP) were set in the "expert" calibration based on some events where both runoff and TP spikes were observed.

3.2 Runoff

It is possible of course to optimise the model parameters to generate either a smaller mass balance error or a larger value of the NSE metric (over 0.8 is possible with this model and data, as evidenced by the Monte Carlo simulation results). Here a compromise was sought between both and to in terms retain the overland flow signal (discussed above) and a good visual fit with the observed flows.

The behavioural flows from the Monte Carlo simulation are shown in Fig. 6 as dotted lines representing the upper (95th percentile) and lower (5th percentiles). There were 511 simulations classed as "behavioural". The envelope of the predicted flows indicates that most of the observed flows during the ten year period of data could be reproduced,

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supporting the choice of runoff processes represented in the CRAFT for this particular catchment. Some events may have been either missed or over predicted which could be due to limitations with using a single rain gauge in the forcing data for the model. Table 5 shows the minimum, median and maximum flows from these timeseries. The table shows that the model outputs are sensitive to the parameters and the end user needs to retain this fact.

3.3 Nutrients

3.3.1 Nitrate

The HFD observed nitrate concentrations in Fig. 3b indicated that concentrations of nitrate in overland flow are much smaller than concentrations in baseflow, and the model parameter COFMIN(NO_3) (see Eq. 10) was set to 0.4 $mg\,L^{-1}$ N. *In the baseline scenario* the proportion of nitrate loads generated by overland flow was thus fairly negligible (< 1 %) and the nitrate loads were split fairly evenly between the SS and DG pathways according to the model. The load from the DG contributed around 31 % of the total load, compared to 43 % of the modelled runoff originating from this pathway. This implies that a significant proportion of nitrate drains from the shallow subsurface (SS) immediately after storm events, probably through either enhanced connectivity due to agricultural drains or recharge into the underlying chalk aquifer. The DG component includes nitrate loads from the WWTPs in the catchment which were estimated to contribute around 7 % (1.5 $kg\,N\,ha^{-1}\,yr^{-1}$) of the total load based on monitoring data from the mid-1980s (Casey et al., 1993).

In terms of the sensitivity of the nitrate results to the flow model parameters, SPLIT was obviously important since it controlled the proportions of mixing the slow and fast nitrate in the total runoff. Overall, the nitrate model has reproduced a moving average of the observed LTD concentrations reasonably well and mean concentrations were within 10 % of the observed (Table 4). The fit between modelled and observed nitrate (Fig. 5) was not so good probably due to timing errors in predicting the onset of dilution,

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although visually (Fig. 4) the model appears to model the seasonal pattern of nitrate fairly well. The CRAFT modelled baseline nitrate export was 33.2 kg N ha^{-1} yr^{-1}, which was higher than the TON export estimated from the HFD of 20.2 kg N ha^{-1} yr^{-1} (Bowes et al., 2009a). Table 5 shows the uncertainty in nitrate loss arising from the hydrological model in terms of the 5th, 95th percentiles and medians of modelled concentrations and yields.

3.3.2 Phosphorus

Bowes et al. (2009b) estimated that between 1991 and 2003, SRP provided 65 % of the TP load in the Frome catchment. In the *baseline scenario*, the DG component generated almost four times the load of SRP than the SS component. This seems reasonable as the DG also includes the SRP loads from the WWTPs, in addition to the SRP originating from springs and seeps from shallow groundwater. Again, the SPLIT parameter in the flow model had a large influence on SRP, by adjusting the ratio between the SS and DG SRP loads. The poorer scatter plot fit depicted in Fig. 5 compared to the visual timeseries fit (Fig. 4) again could be caused by timing issues leading to periods of overprediction and underprediction of concentrations. Visually the SRP concentrations were fitted well using on average and the seasonal patterns and trends were simulated (Fig. 4). Any spikes in the observed data which were not reproduced by the model were probably not based on actual hydrological runoff events (as seen in Fig. 3).

In the baseline scenario the modelled proportion of TP (i.e. PP) generated by overland flow was about 11 % which is quite high considering only 1.2 % of the modelled runoff was generated via this pathway. The PP concentrations generated by the model by the were calibrated by adjusting the value of the KSR(PP) parameter (Table 3). Of the flow model parameters QUICK influenced the PP generated by overland flow the most. The model predicted some spikes in the PP (and therefore TP) during runoff events of up to 2 mg L^{-1} P, shown in Fig. 4. The LTD dataset did not contain many spikes of this magnitude in the TP concentrations, however the HFD data did measure occasional high concentrations of TP associated with runoff events (e.g. those indicated by a “4” on Fig. 3c). Overall, the model underpredicted the observed TP concentrations by up to 60 % (Table 4), despite these spikes being generated, this may be due to additional source(s) of P not being accounted for in the model (e.g. within-channel river channel dynamics and/or conversion of SRP to entrained particulate forms of P as suggested by Bowes et al., 2009a). There may also be a missing source of P from the catchment (e.g. organic P) unaccounted for in the current model (as stated above SRP was estimated to only contribute 65 % of the TP load during the period 1991–2003). Figure 3c, and the simulation results in Figs. 4 and 5, show that the issue of fitting TP at the meso-scale is problematical.

We also calculated the export yields (load per unit area) for each nutrient to show the impact of the flow pathways at transporting nutrients (see Fig. 7 and Table 5). This aggregation lends itself to comparisons with previous studies. The baseline simulation predicted a TP export of 0.69 kg P ha^{-1} yr^{-1} which is slightly more than both the export rate estimated by Hanrahan et.al (2001) for diffuse and point sources in the catchment of 0.62 kg P ha^{-1} yr^{-1} (for calendar year 1998), and the export calculated from the HFD (0.63 kg ha^{-1} yr^{-1}). SRP loads were modelled by Bowes et al. (2009b) and the SRP export was predicted to be 0.44 kg P ha^{-1} yr^{-1} between 1996–2000 (of which WWTP discharges accounted for 49 %), compared to the CRAFT modelled baseline SRP export of 0.62 kg P ha^{-1} yr^{-1} (between 1997 and February 2002). Table 5 shows the uncertainty in terms of the 5th, 95th percentiles and medians of modelled concentrations and yields.

3.4 Management intervention (MI) scenario

The yields of nitrate and TP are summarised by the use of bar charts in Fig. 7 which illustrate the fluxes under the baseline conditions (left bars) and the MI scenario (right bar), and the relative contribution of each of the three flow pathways to these.

The results show that the amount of PP generated by the overland flow pathway (denoted by the blue rectangle in the baseline scenario bar in Fig. 7) has reduced to almost zero due to the reduction in overland flow, and the difference between TP and

SRP export is negligible as a result. This indicates that a limited amount of "pollution swapping" is predicted so that the proportions of PP and SRP comprising TP have changed from 8.8 and 92.2 % to 0 and 100 % respectively under the MI scenario. Nitrate and TP loads are predicted to decrease by 34.4 and 65.0 % respectively. Under the MI scenario, the nitrate concentration in the DG flow component (which includes point sources) was not reduced (it was assumed that WWTP improvements targeted P and not N). Both nitrate and SRP loads in overland flow were negligible (< 0.1 %) under the baseline scenario and have been reduced to effectively zero by drastically reducing the amount of overland flow generated. SRP loads due to point sources are included in the DG component, the predicted load from this component reduced by 63 %. The export of SRP via the faster SS component also reduced by 55 % (to 0.045 $\mathrm{kg\,P\,ha^{-1}\,yr^{-1}}$) under the MI scenario. These reductions in the SRP loads from different components compare well to the overall reductions since the 1990s in point and diffuse sources in the catchment (Bowes et al., 2009b, 2011).

4 Discussion and conclusions

This paper has attempted to explore the role of scale appropriate modelling methods at the meso-scale. It has explored the information content of flow and nutrient data within a case study, that helps justify the choice of model structure and timestep. The MIR approach is thus the minimal parametric representation to model phenomena at the meso-scale as a means to aid decision making at that scale. The model is based on either a simplification of a more complex model or is based on observations made in research studies. The chosen MIR thus focussed on key hydrological flow pathways which are observed at the hillslope scale. The nutrient models were kept very simple ignoring all nutrient cycling aspects. The astute choice of a daily timestep also reduced the burden to route flows through the system. The model deliberately avoids a spatial representation of local land use. This implies that the lumping process is appropriate for circumstances where the local variability disappears when aggregated.

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High resolution data (such as the HFD) for all nutrient parameters are desirable at all scale if it were affordable. However, it is shown here that at the meso-scale these data tends to reflect the noise, incidental losses and within-channel diurnal cycling in the system and hence a lower sampling rate may be suitable in this scale. For the Frome case study a daily flow model could simulate the dominant seasonal and storm driven nutrient flux patterns and thus aid the user in considering a variety of policy decisions. It is stressed that collecting the longest possible high resolution dataset particularly for all forms of nutrients is still of the utmost importance for effective water quality monitoring and identifying the full range of observed concentrations including incidental losses (see Fig. 3c). There may be some evidence here that collecting higher resolution data for nutrients helps to explain the distribution values and addressing the issues of "noise" and diurnal variability (e.g. the fluctuations in P concentrations observed in the River Enborne by Wade et al., 2012 and Halliday et al., 2014) in the datasets. Even so, it may still be beneficial to aggregate sub-daily data to daily data as a optimising the capabilities of a process based model such as the CRAFT and using all the policy relevant information actually contained in the HFD data.

The Frome study revealed a number of interesting factors, leading to a management change scenarios that could be explored. The mean annual SRP concentration that has to be attained in order to comply with the WFD standards for P is 0.06 $mg\,L^{-1}$ P, which was achieved by the MI scenario (modelled mean = 0.053 $mg\,L^{-1}$ P) by reducing the appropriate SRP concentrations in the model's flow pathways to reduce the modelled SRP load by 61.7 %. There are no explicitly defined guidelines for nitrate, except that the maximum concentration must not exceed 11.9 $mg\,L^{-1}$ N, which is imposed on all surface waters in the EU under the terms of the 1991 Nitrates Directive. In terms of nitrate management in the Frome catchment, the observed data from 1997 to 2006 indicated that concentrations were below the limit without any reductions due to nutrient and/or runoff management. The CRAFT model was able to reproduce the seasonality in the observed concentrations and also make predictions of the likely reductions in nitrate concentrations and yields, due to improved management of diffuse sources in the catchment. This MI scenario reduced concentrations from 6–4.3 mg L^{-1} N at the outlet of the Frome. Recent studies of long term trends (Smith et al., 2010; Bowes et al., 2011) showed that nitrate concentrations were observed to be rising in the Frome since the 1940s, however over the simulation period the rate of increase has slowed down and the CRAFT model could predict the weekly time series reasonably well as a result. The MI scenario shows that management interventions to reduce concentrations of nitrate in rapid subsurface flow can have a significant impact at reducing the total nitrate load by 34 %. Management interventions to reduce the concentration of nitrate in deeper groundwater were not investigated as these improvements could take decades to be observable at the monitoring point at the catchment outlet (Smith et al., 2010).

The results of the CRAFT model may best be viewed as event driven export coefficients when the origin of the nutrient is tied to the pathway that generated it. This informs the end user as to the aggregate effect of local policy changes and the importance of storm size and frequency. Whilst we have shown that those impacts are still uncertain it could perhaps encourage more intervention in order to guarantee the success of new policy. Equally, locally observed environmental problems caused by high nutrient concentrations may well be lost due to mixing effect at the meso-scale (i.e. catchment outlet).

The model has been shown to fit the dominant seasonal and event driven phenomena. This is a simple transparent way to convey the mixed effect of land use and hydrological process at the meso-scale for policy makers. The model assumes that the policy maker or informed end user will then manipulate the model to see the likely impacts of regulations. The burden is still on the user to translate the policy into the likely local impact, for example: reduction in N and P loading; more efficient use of N and P in soils and the acute loss of P from well-connected flow pathways. Once the parameters are changed, the net effect at the meso-scale can then be seen instantaneously. The user is encouraged to try many scenarios and to explore the parameters space. The simple Excel interface also allows an instantaneous view of the change which in itself is educational. The range of the fluxes seen can inform the user about the uncertainty of the model when taking decisions and can alert them to unexpected outcomes such as pollution swapping.

The sensitivity and uncertainty analysis carried out on the hydrological model shows the impact on the resultant nutrient fluxes. The output of the uncertainty does suggest that the "expert" choice of a (hydrology and nutrients) model parameter set is not unreasonable. The interactive nature of the tool allows the user to explore ideas and gain confidence in using the tool for scenario testing. This tool is intended to be just one of many required for setting policy at the meso-scale. Equally, despite the uncertainty in the model, the outputs should encourage the user that a range of local scale polices can have a large impact on the final nutrient flux at the meso-scale. The underlying message that lowering nutrient mobilisation risk, lowering flow connectivity and the improvement WWTPs are all beneficial at the meso-scale.

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Table 1. Long term nutrient concentration statistics in the LTD and HFD datasets.

Dataset/Nutrient (time period)	Number of Observations	10th Percentile concentration ($mg\,L^{-1}$)	Median concentration ($mg\,L^{-1}$)	Mean concentration ($mg\,L^{-1}$)	90th Percentile concentration ($mg\,L^{-1}$)
LTD Nitrate (7 Jan 1997–21 Nov 2006)	384	4.6	5.6	5.6	6.9
LTD TP (7 Jan 1997–28 Feb 2002)	176	0.13	0.22	0.21	0.30
LTD SRP (7 Jan 1997–28 Feb 2002)	183	0.08	0.14	0.14	0.20
HFD TON (12 Dec 2004–31 Jan 2006)	1454	4.5	5.4	5.5	6.7
HFD TP (14 Jan 2004–31 Jan 2006)	2290	0.09	0.15	0.17	0.24
HFD SRP (1 Feb 2005–31 Jan 2006)	1340	0.06	0.09	0.09	0.14

Table 2. Hydrological model parameters: "Expert" values; bounds; and performance metrics (baseline scenario).

	KSURF ($m\,d^{-1}$)	SRZMAX (m)	QUICK (–)	SPLIT (–)	KGW (d^{-1})	KSSF (d^{-1})
"Expert" value	0.02	0.019	0.08*	0.56	0.0011	0.041
Lower Bound	0.001	0.001	0	0	0.0001	0.02
Upper Bound	0.1	0.5	5	1	0.02	1
NSE (–)	0.80					
MBE (%)	1.00					

* QUICK was reduced to 0.012 in the MI scenario.

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Table 3. Nutrient modelling parameters; from baseline and MI scenarios (only values that were modified from baseline in the MI scenario are shown in parentheses).

Parameter	Nitrate ($mg\,L^{-1}$ N)	SRP ($mg\,L^{-1}$ P)	PP ($mg\,L^{-1}$ P)
COFMIN	0.4	0.01	0.01
CSS	8.0 (4.0)	0.03 (0.15)	
CGW	4.5	0.22 (0.08)	
KSR(N)*	0	70	700

* Units ($mg\,day\,m^{-4}$) × 10^3.

Table 4. Nutrient modelling results; "Expert" calibration in baseline scenario (1997–2006*).

Dataset	C_{mod} Mean ($mg\,L^{-1}$)	Error (%)	C_{mod} 90th ($mg\,L^{-1}$)	Error (%)	R^2 (–)
LTD Nitrate	6.0	5.7	7.1	3.2	0.04
LTD TP*	0.14	−60	0.20	−54.2	0.02
LTD SRP*	0.13	−5.9	0.20	2.5	0.22

* Calculated up until 28 February 2002 only.

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Table 5. Sensitivity analysis results (1997–2006).

Eval, mean (min–max) C and Q	"Expert" Fit	5th Percentile Behavioural	Median Behavioural	95th Percentile Behavioural
Q (mm d^{-1})	1.4 (0.46–6.4)	1.0 (0.06–4.2)	1.4 (0.20–5.9)	1.8 (0.54–8.1)
TP C^* (mg L^{-1} P)	0.14 (0.06–0.88)	0.14 (0.06–0.21)	0.20 (0.10–0.22)	0.22 (0.20–3.6)
SRP C^* (mg L^{-1} P)	0.13 (0.06–0.22)	0.14 (0.06–0.21)	0.20 (0.10–0.222)	0.22 (0.20–0.35)
Nitrate C (mg L^{-1} N)	6.0 (1.7–7.5)	4.5 (1.2–4.8)	4.8 (4.5–6.8)	6.0 (4.6–7.5)
TP Yield* (kg P ha^{-1} yr^{-1})	0.69	0.72	1.11	1.31
SRP Yield* (kg P ha^{-1} yr^{-1})	0.62	0.72	1.10	1.28
Nitrate Yield (kg N ha^{-1} yr^{-1})	33.2	22.8	26.1	32.1

* Calculated up until 28 February 2002 only.

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Table A1. Nomenclature.

CRAFT	Catchment Runoff Attenuation Flux Tool
DTC	Demonstration Test Catchments
HFD	High Frequency data set of nitrogen and phosphorus, recorded several times per day in the River Frome.
LTD	Long term data set of weekly nitrogen and phosphorus measurements also in the River Frome, modelled by the baseline scenario.
MBE	Mass balance error
MIR	Minimum Information Required
NSE	Nash – Sutcliffe Efficiency (model performance metric)
SRP	Soluble reactive phosphorus (from samples filtered using 0.45 µm paper)
TON	Total oxidised nitrogen (nitrate + nitrite).
TP	Total phosphorus (soluble + insoluble forms)
WFD	Water Framework Directive
WWTP	Wastewater Treatment Plant (Sewage Treatment Works)

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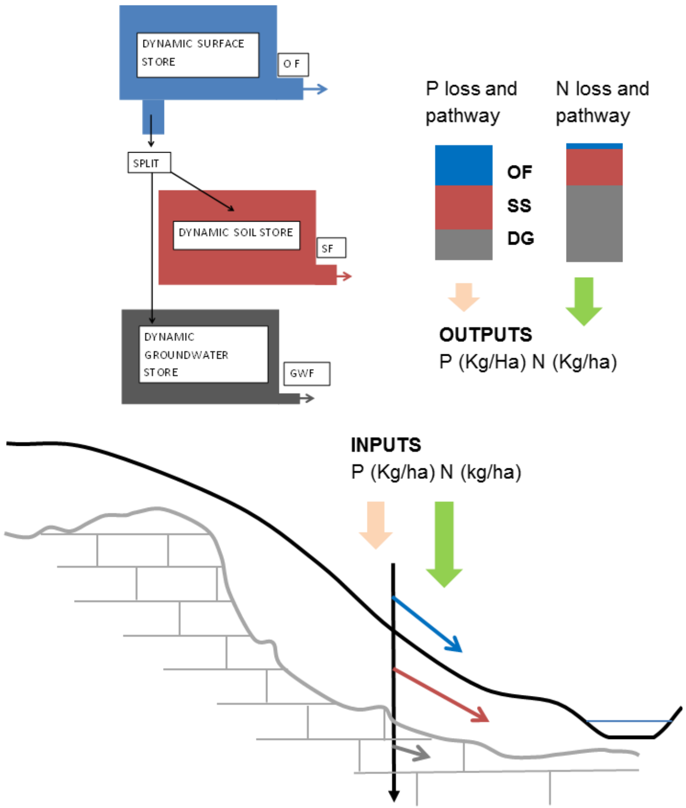

Figure 1. Conceptual diagram of a hillslope and the CRAFT model, showing the dominant flow and transport pathways.

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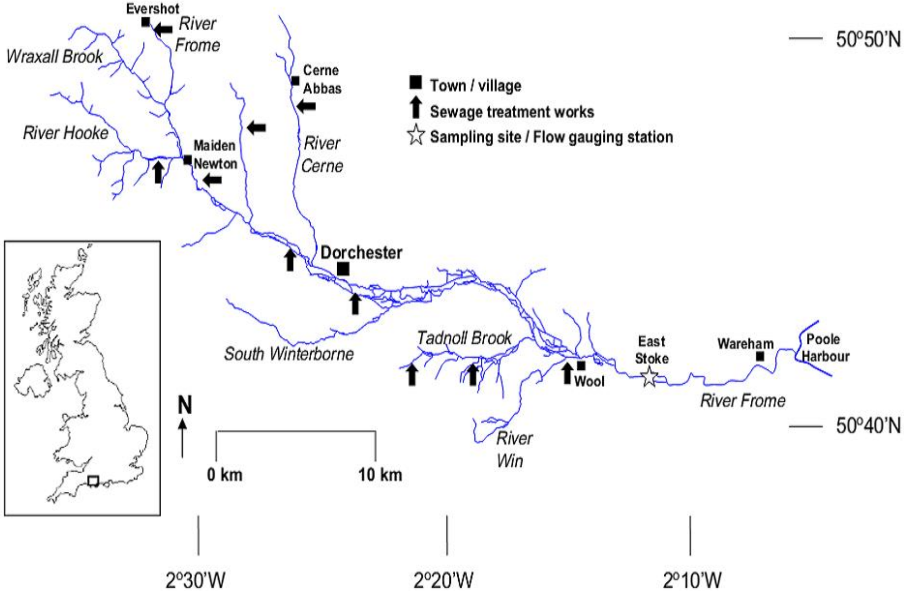

Figure 2. Schematic map of Frome Catchment showing monitoring points (from Bowes et al., 2009a).

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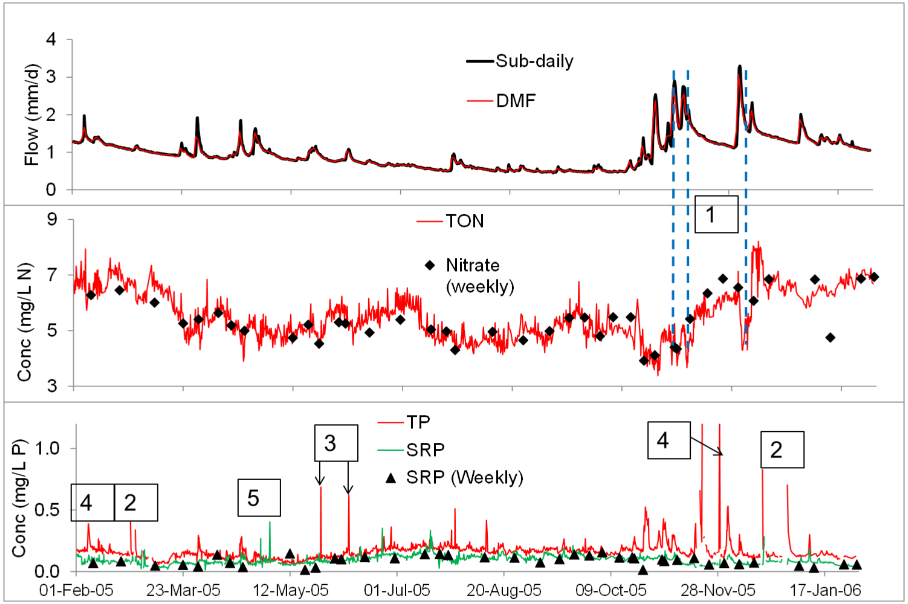

Figure 3. Timeseries plots from the sub-daily HFD dataset from the Frome at East Stoke monitoring point: (**a**, top panel) Flow data from the catchment outlet; (**b**, middle panel) TON and (LTD) Nitrate data; (**c**, bottom panel) TP, SRP and (LTD) SRP data. Refer to text for an explanation of the labelling on the figure panes.

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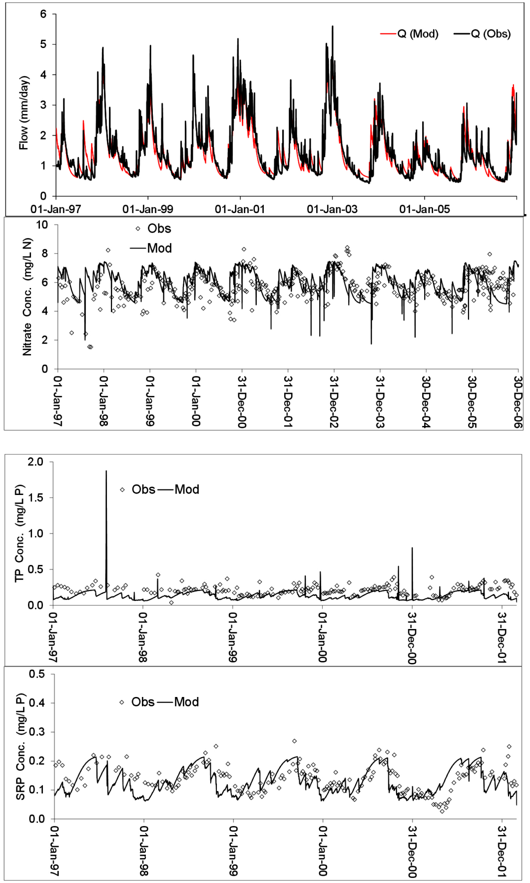

Figure 4. Timeseries plots of modelled (from "Expert" calibration) and observed (LTD) flows and nutrient data: (from top to bottom): **(a)** flows; **(b)** nitrate; **(c)** TP; **(d)** SRP.

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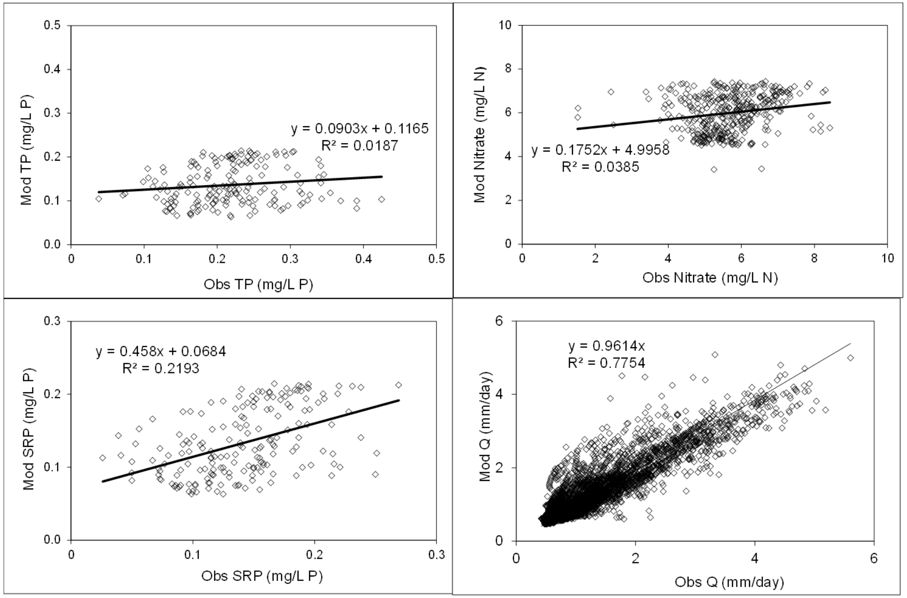

Figure 5. Scatter plots of modelled (from "Expert" calibration) and observed and nutrient data, bottom right panel: modelled and observed flows.

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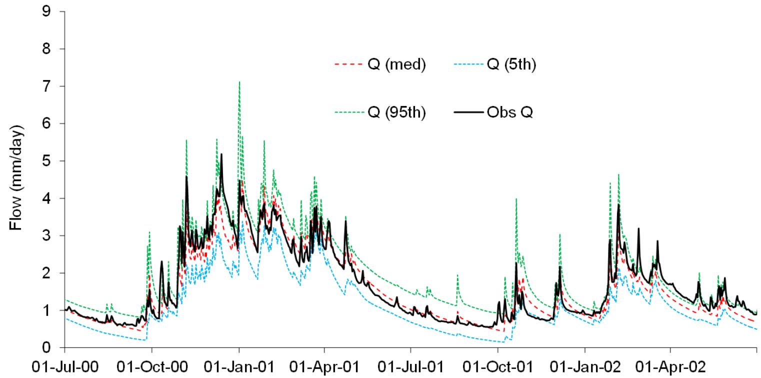

Figure 6. Timeseries plot of modelled (Monte Carlo) 5th and 95th percentile and median flows, and observed flows.

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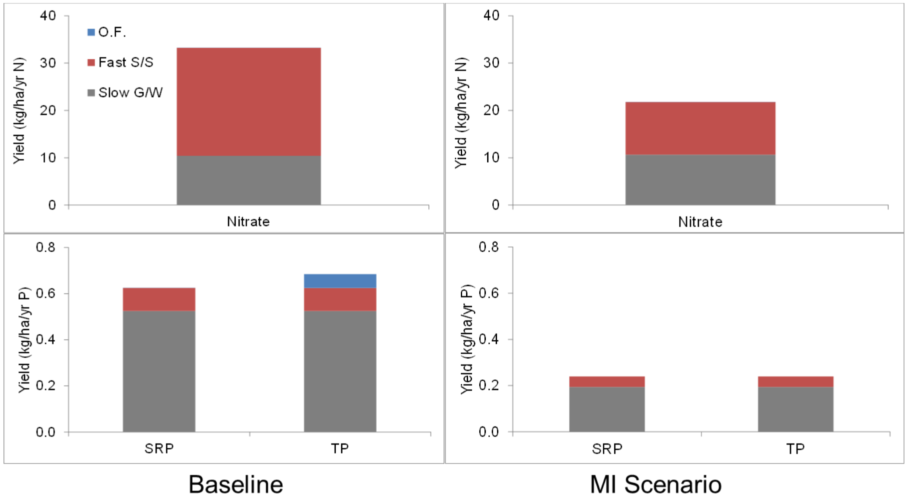

Figure 7. Comparison of nutrient yields from baseline (left panels) and management intervention scenarios (right panels).

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