Improvement Programme for England's Natura 2000 Sites (IPENS) – Planning for the Future IPENS008a

Application of a cross sector pollutant source apportionment modelling framework to protected sites

Covers multiple Natura 2000 sites

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Foreword

The **Improvement Programme for England's Natura 2000 sites (IPENS)**, supported by European Union LIFE+ funding, is a new strategic approach to managing England's Natura 2000 sites. It is enabling Natural England, the Environment Agency, and other key partners to plan what, how, where and when they will target their efforts on Natura 2000 sites and areas surrounding them.

As part of the IPENS programme, we are identifying gaps in our knowledge and, where possible, addressing these through a range of evidence projects. The project findings are being used to help develop our Theme Plans and Site Improvement Plans. This report is one of the evidence project studies we commissioned.

Water pollution has been identified as one of the top three issues in all Natura 2000 rivers. It also affects many terrestrial and some marine and coastal Natura 2000 sites.

Diffuse water pollution is the release of potential pollutants from a range of activities that individually may have little or no discernable effect on the water environment, but at the scale of a catchment can have a significant cumulative impact. The sources of diffuse water pollution are varied and include agriculture, urban run-off, highways drainage and non mains sewage discharges.

Often sites are affected by multiple sources of pollution, and in many cases a better understanding is required of the pollution issue to inform and guide the actions required.

This report develops a national scale framework for assessing the multiple sources of sediment and nutrient pollution impacting on a number of aquatic sites designated for wildlife. It is one of four produced by the IPENS project "Meeting local evidence needs to enable Natura 2000 Diffuse Water Pollution Plan Delivery."

The results have been used by Natural England and others to help develop and implement the Diffuse Water Pollution Theme Plan and will be used to develop and implement individual Diffuse Water Pollution Plans for target catchments.

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Application of a cross sector pollutant source apportionment modelling framework to protected sites

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Abstract

Diffuse water pollution represents a major environmental issue for the European Union. Attempts to provide a coordinated approach to the management of the freshwater environment require appropriate tools for spatial analysis to deliver the evidence base for informing targeted decision making and interventions. In this context, this report outlines the application of a new national multiple pollutant (sediment and nutrients) source apportionment screening framework to a select list of aquatic sites designated for wildlife (Sites of Special Scientific Interest; SSSIs) in England. The cross sector modelling framework includes emissions to the aquatic environment from both diffuse (agriculture, urban, river channel banks, atmospheric) and point (sewage treatment works (STWs), septic tanks, combined sewer overflows (CSOs), storm tanks) sources. The outputs from the modelling exercise will contribute to the planning of interventions designed to improve the status of the designated sites.

Introduction

Pollution of the aquatic environment, including rivers and lakes, remains a persistent and widespread problem in many parts of the world (UN-Water, 2011; Patterson et al., 2013). Freshwater ecosystems deliver services crucial to human survival and wellbeing and yet globally, their degradation has outstripped the success of remedial programmes (Ormerod et al., 2010). More specifically, agricultural pollution has been widely recognised as one of the key contributors to the degradation of river water quality and aquatic biodiversity (Carpenter et al., 1998; Smith, 2003; Berkes et al., 2003; Poole et al, 2013). Agricultural emissions of various pollutants contribute to environmental problems including those resulting from excessive loss of sediment (Collins et al., 2011) and nutrients (Hilton et al., 2006) to freshwaters. Sediment pollution, in particular, has been identified as impacting adversely on some of the most important sites across England designated for wildlife. The need to tackle excessive pollutant emissions to freshwater habitats, including designated sites, has resulted in the introduction of significant water policy instruments including the Water Framework Directive (WFD) in Europe (European Commission, 2000).

In England and Wales, a number of mechanisms are being used to help deliver the WFD, through improved management of the agricultural sector and its pollutant emissions (McGonigle et al., 2012). These include baseline regulations for farmers such as those in Cross Compliance or Action Programmes for Nitrate Vulnerable Zones, targeted advice or training such as Catchment Sensitive Farming or the Campaign for the Farmed Environment (CFE, 2011) and incentives delivered by agri-environment or payments for ecosystem services schemes. Balanced approaches are required for the management of freshwater environments to help achieve multiple goals (McGonigle et al., 2012). A key challenge facing the agricultural sector is the need to increase productivity to feed a growing population in the context of minimising environmental burden (Foresight, 2011).

Attempts to provide a coordinated approach to environmental management such as that targeting excessive pollutant influx to aquatic sites designated for wildlife require appropriate tools for spatial analysis and informing decision making (Giupponi and Vladimirova, 2006). Computational methodologies for characterising and assessing pollution pressures on the aquatic environment differ profoundly in terms of data requirements, process or pathway representation and complexity. The use of modelling has gained momentum and these approaches range from simple export coefficient frameworks (Beaulac and Reckhow, 1982; Johnes, 1996; Shaffner et al., 2009; Ma et al., 2011) to regression models (Alexander et al., 2002) to more complex deterministic tools for individual or multiple pollutants (Horn et al., 2004; Rao et al., 2009; Coffey et al., 2010; Rivers et al., 2013; Parajuli et al., 2013).

Aquatic pollutant source decomposition has been undertaken using empirical load apportionment modelling founded on the fundamental contrast in the timing of point (e.g. water treatment discharges) and diffuse (e.g. agricultural) emissions and corresponding associations with flow (Bowes et al., 2008; Howden et al., 2009; Greene et al., 2011). Source apportionment screening tools are, however, more appropriate for macro-scale analyses and for guiding targeted decision making (Navulur and Egel, 1996; McLay et al., 2001; Margane, 2003; Collins and Anthony, 2008; Collins et al., 2009a,b; OECD, 2012; Comber et al., 2011, 2013). These screening tools in their most rudimentary format represent a simplification of the DPSIR (Driving force – Pressure – State – Response) conceptual framework (EEA, 1999) by focussing on the key pollutant pressures on the aquatic environment. Previous additional examples include Brouwer and Van Pelt (2003), Giupponi and Vladimirova (2006), Anthony et al. (2006) and Brouwer and De Blois (2008). Since macro-scale pollutant screening tools can be used to appraise primary sources of emissions to the aquatic environment, they are ideally placed to ensure that no individual sector is unduly burdened with abatement costs in the context of the 'polluter pays' principle.

Evidence for significant and sustained improvements in river water quality and aquatic ecology in response to mitigation programmes remains scant due to a number of confounding factors. These include, amongst others, failure to take sufficient account of cross sector or multiple source contributions to pollutant loadings (Collins et al., 2014) and the need for substantial reductions in pollutant pressures before ecological responses are observed (Bowes et al., 2011). On-farm interventions including those in agri-environment schemes across England and Wales have been used to help reduce the detrimental impacts of agricultural pollutant emissions (Natural England, 2012), but since such schemes are funded by public tax revenue, it is important that they are optimised spatially to help maximise the delivery of multiple outcomes (Poole et al., 2013). An increasing focus on the protection of aquatic sites designated for wildlife from excessive pollutant loadings represents one dimension of ongoing attempts to maximise environmental improvement in priority catchments.

Against this background, a small science project was commissioned by Natural England to deploy a national scale framework for assessing the multiple sources of sediment and nutrient pollution impacting on a select list of aquatic sites designated for wildlife. The following sections detail the project key deliverables, fundamental components of the aquatic pollutant screening tool and the results of the source apportionment for the study sites.

Key deliverables

The key deliverables were:

a) to screen the SSSI sites shortlisted for the delivery team by Natural England to identify those where agricultural inputs of sediment and nutrients represent the dominant pollutant source.
b) to assess the ‘gap’ between contemporary sediment pressure on the SSSI sites and estimates of modern background sediment delivery to freshwater.

The study sites

The project focussed on 40 sites designated as freshwater Sites of Special Scientific Interest (SSSIs). These sites are listed in Table 1 and shown in Figure 1. The delivery team requested a GIS shape file for the catchment boundaries of these designated aquatic sites. Upon inspection, the shape file provided to the contractor by Natural England contained a total of 50 sites, but the match-up between these and the short list of SSSI designations was not 100% (Table 2). On this basis and following discussions with Natural England over the sites not in the GIS shape file, it was agreed that the catchment boundaries for a further 10 sites from the original list would be assessed (Table 3). These catchment boundaries were derived using the CatchmentsUK tool. Finally, seven catchment boundaries from the 10 were accepted by Natural England, with those for the remaining three designated sites being judged to be too small (<10 km^2).

Table 1: The 40 SSSIs shortlisted for this study.

Site ID	SSSI name
1000501	Ant Broads And Marshes
1003807	Aqualate Mere
1006622	Avon Valley (Bickton To Christchurch)
1002462	Barnby Broad & Marshes
1003782	Bassenthwaite Lake
1000880	Bure Broads and Marshes
1002654	Chesil & The Fleet
1001594	Hawes Water
1002380	Hornsea Mere
1001669	Leighton Moss
1001818	The Mere, Mere
1004035	Marazion Marsh
1000503	Ouse Washes
1002185	Portholme
2000183	River Avon System
2000139	River Axe
1005993	River Beult
2000151	River Camel Valley And Tributaries
2000452	River Dee (England)
1003398	River Derwent
2000214	River Derwent And Tributaries
2000215	River Eden And Tributaries
2000147	River Ehen (Ennerdale Water To Keekle Confluence)
2000220	River Frome
2000227	River Itchen
2000164	River Kennet
2000335	River Kent And Tributaries
2000155	River Lambourn
1006616	River Lugg
2000416	River Mease
1006323	River Nar
2000102	River Teme (inc R. Clun)
2000170	River Test
1006328	River Wensum
1006327	River Wye
2000479	Slapton Ley
2000355	Trinity Broads
2000455	Tweed Catchment Rivers - England: Lower Tweed And Whiteadder
2000288	Tweed Catchment Rivers - England: Till Catchment
1005779	Upper Thurne Broads And Marshes

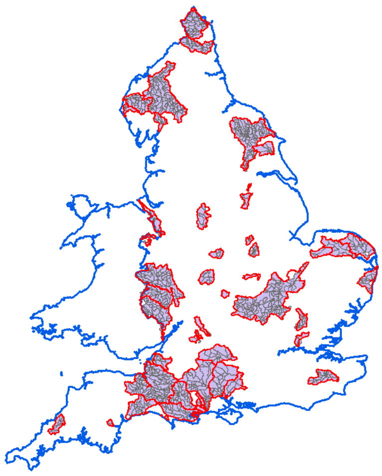

Figure 1: The distribution of the 40 SSSIs shortlisted for this study.

Table 2: SSSIs in both the GIS catchment boundary shape file and the short list of national grid references (NGRs) provided by Natural England.

SSSI name	NGR
River Avon System	SZ141985
Ouse Washes	TL490879
River Axe	SY275963
River Beult	TQ736488
River Camel Valley And Tributaries	SX117756
River Dee (England)	SJ418501
River Derwent	SE704300
River Derwent And Tributaries	NY265209
River Eden And Tributaries	NY450218
River Ehen (Ennerdale Water To Keekle Confluence)	NY050158
River Itchen	SU476240
River Kennet	SU337695
River Kent And Tributaries	SD500964
River Lugg	SO440625
River Mease	SK276118
River Nar	TF834169
River Teme (inc R. Clun)	SO507745
River Test	SU378386
River Wensum	TG023176
River Wye	SO519384

Table 3: The 10 sites (in bold) not in the national shape file selected in agreement with Natural England for the derivation of contributing catchment boundaries to augment the shape file provided by the funder.

SSSI	NGR
Ant Broads And Marshes	**TG365193**
Aqualate Mere	**SJ773204**
Barnby Broad & Marshes	TM477910
Bassenthwaite Lake	**NY218290**
Bure Broads and Marshes	**TG348159**
Chesil & The Fleet	SY611805
Hawes Water	**SD476767**
Hornsea Mere	TA188468
Leighton Moss	**SD482749**
The Mere, Mere	**SJ733819**
Marazion Marsh	SW512316
Portholme	TL236708
River Frome	SY756906
River Lambourn	**SU417722**
Slapton Ley	**SX828441**
Trinity Broads	**TG457133**
Tweed Catchment Rivers - England: Lower Tweed And Whiteadder	NT969516
Tweed Catchment Rivers - England: Till Catchment	NU006297
Upper Thurne Broads And Marshes	TG435211

The cross sector pollutant source apportionment modelling framework

The modelling framework integrates information on pollutant emissions from multiple sectors to provide source apportionment. Table 4 summarises the key sources and corresponding pollutant emissions characterised in the framework. The outputs can be readily mapped in GIS to support policy on the spatial targeting of interventions to address water quality targets. On this basis, the modelling framework provides a basis for supporting integrated environmental assessment (cf. Laniak et al., 2013). Coastal water bodies and therefore any corresponding designated aquatic sites are not represented in the modelling framework outputs.

Table 4: A summary of the key sources and their pollutant emissions represented in the cross sector modelling framework.

Source	Sediment	Total Phosphorus	Total Nitrogen
Diffuse agricultural	X	X	X
Diffuse urban	X	X	X
Diffuse river channel banks	X	X	X
Diffuse atmospheric deposition to water	-	X	X
Sewage treatment works (STWs)	X	X	X
Septic tanks	-	X	X
Combined sewer overflows (CSOs)	-	X	X
Storm tanks	-	X	X

Pollutant emissions from diffuse agricultural sources

Diffuse agricultural pollution emissions (sediment, total phosphorus, total nitrogen) were generated using the Agricultural Pollutant Transfer (APT) framework, which has been developed for national scale modelling for water quality policy support under a number of Defra funded research projects including WQ0128. APT builds upon the existing, validated PSYCHIC model (Collins et al., 2007; Davison et al., 2008; Stromqvist et al., 2008; Collins and Anthony, 2008; Collins et al., 2009a,b) for phosphorus and sediment emissions and NIPPER model (Shepherd, 2007; Gooday et al., 2008) for nitrogen losses. By combining these two process-based models within a single framework it is possible to produce estimates of multiple pollutant losses which benefit from shared input data and common hydrological and crop growth sub-models.

The APT framework predicts pollutant losses from agricultural land and woodland, including pollutant emissions delivered to watercourses. It operates at a daily time step and can output at a 1km^2 spatial resolution. The APT framework predicts losses at field scale, with a waterbody represented as a large number of fields which are then subject to landscape scale retention to estimate delivery of pollution from agricultural land to rivers. Land drainage as a pollution delivery pathway is represented, as well as surface runoff. This is important given that ~40% of agricultural land across England and Wales has some form of assisted under-drainage.

The APT framework requires three core types of data; daily weather information, physical attributes of the land, and crop and livestock management data. The daily weather data was interpolated for each from existing UK Meteorological Office records using an inverse distance weighting function in the IRRIGUIDE tool (Bailey and Spackman, 1996). A catchment is represented by a small number of major soil types taken from the NSRI Natmap Soils Database. Other physical data required as input include slope and altitude, plus field boundary features (based on the Countryside Survey; Hornung, 1998) which are a key control on land-to-river connectivity.

The crop areas are based upon the 2010 June Agricultural Census completed by farmers, which has been mapped to a 1 km grid using the approach described in Comber et al. (2008). Manure and excreta distribution and management are calculated using the Manures-GIS system (ADAS, 2008), which uses livestock numbers from the June Agricultural Census. Data on fertiliser application rates for different crop types were taken from the 2010 British Survey of Fertiliser Practice (BSFP; Thomas, 2011). The APT framework models crops as either part of a 3 year rotation, or (primarily for permanent grassland) as continuous cropping. The primary benefit of this is that it allows the predictions to include the effects of crop and manure management in previous years on the nitrogen cycle. APT runs covered a 20-year period (1991-2010) and annual average pollutant losses over this period were calculated for inclusion in the pollutant source screening.

Some previous source apportionment studies for England and Wales have used the NEAP-N model (Lord and Anthony, 2000) to estimate nitrate emissions from the agricultural sector (e.g. Hughes et al., 2011). The NEAP-N and APT models both use agricultural census and soils data as key inputs, but differ in terms of temporal resolution and process representation. NEAP-N is an export-coefficient based model reflecting average climatic conditions under typical farming practice, with coefficients determined from experimental data. For each arable crop, a coefficient represents the N at risk of leaching over winter, largely controlled by post-harvest mineralisation. The potential N loss from un-grazed grassland is assumed to be small, with a per capita coefficient used for each livestock type, reflecting the manure from that livestock and the fertiliser required to grow their feed. A simple relationship is used to determine the proportion of the N at risk that is actually leached, based upon an annual average volume of drainage and the water capacity of the soil. In contrast, the APT model by considering nitrate leaching over a whole crop rotation, explicitly captures the legacy effects of previous cropping and manure management. Crop growth and nitrogen uptake, evapotranspiration, denitrification and mineralisation are calculated daily, with the movement of water and nitrate simulated through a discretised soil profile. The amount of nitrogen in managed manure and excreta is calculated explicitly using the Manures-GIS model (ADAS, 2008) based upon livestock numbers and detailed information on livestock management. Managed manure is applied appropriately within the crop rotations represented within the APT model.

Pollutant emissions from diffuse urban sources

The diffuse urban source category lumps information on pollutant emissions from open urban spaces, industrial and commercial areas, residential zones and main highways. Annual average runoff L (mm) from urban land areas is calculated using the Wallingford Modified Rational Method (DoE, 1981):

L = R * (0.829 * P +0.078 * U – 20.7)

where:

R = annual average rainfall (mm)

P = proportion of land area that is impermeable (%)

U = catchment wetness index determined from annual average rainfall (Mitchell et al., 2001).

Annual average runoff from urban areas is combined with representative event mean concentrations (EMCs) of 123 mg/L for sediment, 0.31 mg/L for TP and 2.1 mg/L for TN. These nationally representative EMCs were calculated by taking the overall average of published (Mitchell, 2005) data on sampled pollutant concentrations (sediment: 126.3 mg/L for open urban; 50.4 for industrial/commercial; 85.1 mg/L for residential' 194.5 mg/L for motorways and 156.9 mg/L for main highways / TP: 0.22 mg/L for open urban, 0.30 mg/L for industrial/commercial; 0.41 mg/L for residential; 0.28 mg/L for motorways and 0.34 mg/L for main highways / TN: 1.68 mg/L for open urban, 1.52 mg/L for industrial/commercial; 2.85 mg/L for residential and 2.37 mg/L for main highways). The use of EMCs to estimate pollutant loadings has been widely reported (e.g. Collins and Anthony, 2008; Collins et al., 2009a,b; Lee et al., 2011; Jung et al., 2013).

Pollutant emissions from river channel bank sources

Inputs from eroding river channel banks were estimated using a modified version of the approach reported by Collins and Anthony (2008) and Collins et al. (2009a,b). The original index reported by these studies is based on river regime and the duration of excess shear stress as a percentage of the runoff year. Flow duration curves are generated using the method reported by Gustard et al. (1992) with representative Q_{95} values assigned to each soil series using the HOST (Hydrology of Soil Types) classification scheme (Boorman et al., 1995). Shear stress on river banks for a given flow depth is estimated using Guo and Julien (2005) assuming a rectangular channel profile. The relationship published by Julian and Torres (2006) was used to estimate the critical shear stress threshold for river banks across England and Wales. Channel density was included in the original index to take account of the opportunity for river bank erosion and the index was calibrated using empirical source fingerprinting data from 22 catchments (Walling and Collins, 2005). To update the index and reflect geomorphological control on bank erosion rates associated with curvature, the Detailed River Network (DRN; Environment Agency) was used to estimate channel density and sinuosity in all waterbodies. Regression analysis demonstrated a positive relationship (r^2 = 0.66) between bank erosion yield and channel density multiplied by sinuosity, where the proportion of the channel network with a sinuosity between 1.3 and 1.7 exceeded 10%. On this basis, the original index was applied in waterbodies with <10% channel sinuosity of 1.3-1.7 and the modified index in those waterbodies with >10% corresponding sinuosity. The revised index was calibrated using source fingerprinting data from 30 study catchments (Collins pers comm.). Measured national average TN (3350 mg/kg) and TP (550 mg/kg) contents in river channel banks provided by sediment source fingerprinting sampling programmes (Collins pers comm.) were used in conjunction with the estimated sediment loss from this source to generate corresponding layers for these nutrients.

Pollutant inputs from direct atmospheric deposition to water

Water pollution source apportionment needs to take account of direct atmospheric inputs to freshwater (e.g. Hunt et al., 2004; Bealey et al., 2007). The NEAP-N model (Anthony et al., 1996; Lord and Anthony, 2000; Silgram et al., 2001) was used to estimate direct deposition of N to open water. For TP, a representative concentration of 0.045 mg/L was used in conjunction with annual average rainfall (AAR) for the period 1961-90 to estimate direct atmospheric deposition. The representative TP concentration of 0.045 mg/L is derived from the monitored average soluble phosphorus content (0.022 mg/L) in rainfall provided by the UK Environmental Change Network (ECN) monitoring sites and a correction factor for the ratio of soluble to total phosphorus for precipitation (Neal et al., 2004). The proportion of atmospheric N or TP deposited directly onto open water was estimated using information on catchment land cover, including the percentage of open water, in the ADAS national database (Comber et al., 2008).

Pollutant emissions from sewage treatment works (STWs)

Sewage treatment work (STW) emissions to rivers were estimated using a national register (n = 6790) of consented effluent discharges together with measured flows and pollutant concentrations for the period 2010-2012 provided by the Environment Agency. Those STWs with mean daily discharges <3 m^3 or without measured discharge data (dry weather flow, maximum flow, mean daily) were removed from the analysis. The ratio between daily actual and consented discharges was used to correct flow data and these estimates were combined with measured pollutant concentrations to calculate emissions. For sediment, regional specific concentrations were estimated from monitored data for two broad categories of STWs; water company and non-water company works. The national average sediment concentration for water company STWs was 13 mg/L compared with 32 mg/L for non-water company discharges. Total N in STW discharges was estimated by combining regionally averaged data on total oxidisable N (nitrate-N plus nitrite-N) and ammonia representing the total inorganic N fraction with organic N data. In the case of the inorganic N fraction, the national monitored means for STWs were 17 mg/L (minimum 0.018 mg/L, maximum 54.8 mg/L) for TON (n sites = 235) and 4.7 mg/L (minimum 0 mg/L, maximum 11.3 mg/L) for ammonia (n sites = 3015). Monitored organic N data for STWs was far more sparse (n sites = 2 with 12 samples) but the limited evidence suggested typical values of ~3 mg/L. These limited data on organic N were consistent with those in published literature (e.g. Jiminez et al., 2007). The corresponding monitored national average TP content in STW discharges was 4.38 mg/L (minimum 0.011 mg/L, maximum 54.6 mg/L). A default pollutant concentration of 5 mg/L for TP agreed with Environment Agency experts was used to fill gaps in the national database. Total STW loads were further filtered on the basis of distance between outfalls and receiving rivers and by selecting only those discharge environments (freshwater river, onto land/into watercourse, lake/reservoir with outlet, canal) with the potential to deliver sediment and nutrient pollution to the national river network.

Pollutant emissions from septic tanks

Septic tanks serve the estimated 2.1 million people in England and Wales not connected to the mains sewer network (Comber et al., 2011). The locations of septic tanks were taken from a recent study on phosphorus emissions (Environment Agency, 2010). Pollutant contributions from septic tanks to surface water concentrations were estimated at 1 km^2 resolution using:

$$LST_{hij} = ILST_{hi} \times R_i \times SSC_{hj}$$

where:
LST_{hij} = the load delivered to the receiving surface water from septic tank $_h$ in grid $_j$ of pollutant $_i$
$ILST_{hi}$ = the input load of pollutant $_i$ into septic tank $_h$
R_i = removal efficiency for pollutant $_i$ (assumed to be 20%; Comber et al., 2011)
SSC_{hj} = surface water connectivity for the 1 km^2 grid $_j$ within which septic tank $_h$ occurs (taken from the PSYCHIC model; Davison et al., 2008)
Estimates of the influent pollutant load into each septic tank were calculated on the basis of:

$$ILST_{ij} = P_h \times C_j$$

where:
P_h = population served by each septic tank (assuming 2.36 people per household; Comber et al., 2011)
C_j = annualised per capita usage for each pollutant (1.2 g/capita/day for N and 2.2 g/capita/day for P; Comber et al., 2011)

Pollutant emissions from combined sewer overflows (CSOs)

Information on pollutant discharges from CSOs was taken from the SAGIS framework (Comber et al., 2011, 2013). This framework assumes that combined sewers can carry six times dry weather flow (DWF) before CSOs discharge to surface waters. CSO discharge volume is assumed to be a function of rainfall intensity, sewer capacity, surface permeability and the proportion of surface runoff entering the sewerage system. The rainfall intensity threshold (RIT; mm) at which sewer capacity is exceeded is estimated using:

$$\mathrm{RIT_{CSO}} = \frac{\mathrm{SC} \times \mathrm{TDWF}}{\mathrm{TISA}}$$

where:
SC = sewer capacity as a multiple of dry weather flow (DWF; L)
TDWF = national scale DWF for all STWs across England and Wales
TISA = national scale impermeable surface area (m^2).
CSO discharge at WFD waterbody scale is estimated on the basis of:

$$CSO_i => RIT_i \times ISA_i$$

where:
$CSO_i =$ = total CSO discharge for waterbody $_i$
$ERIT_i$ = total rainfall (mm) in waterbody $_i$ in excess of the RIT
ISA_i = total impermeable surface area in waterbody $_i$

CSO water discharges were combined with published values for concentration (Comber et al., 2011) for TP (0.27 mg/L in surface runoff and 13.9 mg/L in raw sewage) and TN (1.93 mg/L and 39.7 mg/L, respectively).

Pollutant emissions from storm tanks

The approach to estimating storm tank overflow pollutant emissions is reported in Comber et al. (2011) and is similar to that used for CSOs. This work assumes that storm tanks can retain three times the volume of DWF before discharging to surface waters. The national scale rainfall intensity threshold ($\mathrm{RIT_{ST}}$; mm) at which storm tanks emit pollution to surface waters is estimated as:

$$\mathrm{RIT_{ST}} = \frac{(\mathrm{SC} \times \mathrm{TDWF}) + (\mathrm{PE} \times \mathrm{STC})}{\mathrm{TISA}}$$

where:
$\mathrm{RIT_{ST}}$ is a national scale rainfall intensity threshold (mm) at which the on-site storage capacity will be exceeded

SC is the sewer capacity (L) as a multiple of dry weather flow (DWF). This value reflects the flow retention capacity of the sewer

TDWF is an estimate (L) of the national scale DWF (i.e. the sum of DWF for all STWs across England and Wales)

PE is the population equivalent sewage treatment capacity

STC is the on-site storage capacity (68 L per population equivalent)

TISA is an estimate of the total (national scale) impermeable surface area (m^2)

The estimated discharges for storm tank overflows were combined with the same measured values of TP and TN content used in conjunction with the CSO data layers (Comber et al., 2011).

Estimates of pollutant source apportionment

Sediment source apportionment

Table 5 presents the estimates of sediment source apportionment for the study sites. It should be noted that these estimates relate to sediment delivery to the watercourses and do not take into account any instream processing (e.g. sediment storage or remobilisation). The highest contribution from agriculture (96%) was estimated for the Slapton Ley study catchment and the lowest (47%) for the River Mease. In the case of sediment contributions from diffuse urban areas, the highest contribution (25%) was estimated for the Trinity Broads study area and the lowest (0%) for the Aqualate Mere, River Teme and Slapton Ley study catchments. Contributions of sediment from eroding channel banks ranged from 4% (Slapton Ley) to 52% (Aqualate Mere). The corresponding contributions from STW outfalls were generally the lowest from the sediment source types, with the highest contribution (10%) being estimated for the River Wensum.

Table 5: Sediment source apportionment estimates for the study sites.

SSSI name	% sediment from agriculture	% sediment from diffuse urban	% sediment from channel banks	% sediment from STWs
Ant Broads And Marshes	54	5	39	2
Aqualate Mere	48	0	52	0
Avon Valley (Bickton To Christchurch)	Uncertainty over catchment boundary			
Barnby Broad & Marshes	Uncertainty over catchment boundary			
Bassenthwaite Lake	87	1	12	0
Bure Broads and Marshes	60	4	35	1
Chesil & The Fleet	Uncertainty over catchment boundary			
Hawes Water	Catchment area <10 km^2			
Hornsea Mere	Uncertainty over catchment boundary			
Leighton Moss	Catchment area <10 km^2			
The Mere, Mere	Catchment area <10 km^2			
Marazion Marsh	Uncertainty over catchment boundary			

Ouse Washes	80	5	12	3
Portholme	Uncertainty over catchment boundary			
River Avon System	57	4	37	2
River Axe	80	1	19	0
River Beult	61	1	37	1
River Camel Valley And Tributaries	81	2	17	0
River Dee (England)	50	6	39	5
River Derwent	70	1	28	1
River Derwent And Tributaries	86	1	13	0
River Eden And Tributaries	70	1	29	0
River Ehen (Ennerdale Water To Keekle Confluence)	91	2	7	0
River Frome	Uncertainty over catchment boundary			
River Itchen	65	15	20	0
River Kennet	65	4	24	7
River Kent And Tributaries	90	2	8	0
River Lambourn	88	2	7	3
River Lugg	74	1	24	1
River Mease	47	5	46	2
River Nar	76	2	21	1
River Teme (inc R. Clun)	67	0	32	1
River Test	63	4	33	0
River Wensum	58	9	23	10
River Wye	70	1	29	0
Slapton Ley	96	0	4	0
Trinity Broads	57	25	18	0
Tweed Catchment Rivers - England: Lower Tweed And Whiteadder	Uncertainty over catchment boundary			
Tweed Catchment Rivers - England: Till Catchment	Uncertainty over catchment boundary			
Upper Thurne Broads And Marshes	Uncertainty over catchment boundary			

Total phosphorus source apportionment

Table 6 presents the estimates of phosphorus source apportionment for the study sites. It should be noted that these estimates relate to total phosphorus (TP) delivery to the watercourses and do not take into account any instream processing (e.g. associated with phosphorus adsorption and desorption / phase changes). The highest contribution from agriculture (87%) was estimated for the Slapton Ley study catchment and the lowest (6%) for the Ant Broads and Marshes study area. In the case of TP contributions from STWs, the highest contribution (83%) was estimated for the River Dee (England) study area and the lowest (0%) for the River Ehen and Trinity Broads study catchments. Contributions of TP from diffuse urban sources ranged up to 8% (Trinity Broads). The corresponding contributions from storm tanks ranged up to 29% (River Itchen) and those from septic tanks up to 15% (Trinity Broads). The highest TP contribution from CSOs (57%) was estimated for Trinity Broads. Eroding channel bank contributions to TP loadings were highest (8%) for the Aqualate Mere and River Eden and tributaries study sites. Atmospheric deposition was consistently one of the smallest sources of TP in the study areas but was estimated to contribute 10% of this pressure on watercourses in the Trinity Broads study catchment.

Table 6: Total phosphorus source apportionment estimates for the study sites.

SSSI name	% TP from ag	% TP from STWs	% TP from urban	% TP from storm tanks	% TP from septic tanks	% TP from CSOs	% TP from bank erosion	% TP from atmospheric deposition
Ant Broads And Marshes	6	75	1	2	7	6	2	1
Aqualate Mere	53	28	0	1	7	2	8	1
Avon Valley (Bickton To Christchurch)	Uncertainty over catchment boundary							
Barnby Broad & Marshes	Uncertainty over catchment boundary							
Bassenthwaite Lake	85	5	1	1	1	1	3	3
Bure Broads and Marshes	18	56	2	3	8	9	3	1
Chesil & The Fleet	Uncertainty over catchment boundary							
Hawes Water	Catchment area <10 km^2							
Hornsea Mere	Uncertainty over catchment boundary							
Leighton Moss	Catchment area <10 km^2							
The Mere, Mere	Catchment area <10 km^2							
Marazion Marsh	Uncertainty over catchment boundary							
Ouse Washes	35	40	2	6	2	13	1	1
Portholme	Uncertainty over catchment boundary							
River Avon System	11	75	1	4	2	5	2	0
River Axe	71	15	1	2	4	3	4	0
River Beult	37	49	1	1	6	2	4	0
River Camel Valley And Tributaries	68	4	2	8	7	6	4	1
River Dee (England)	10	83	1	1	1	3	1	0
River Derwent	37	52	0	1	2	4	4	0
River Derwent And Tributaries	81	8	1	1	1	2	3	3
River Eden And Tributaries	67	15	1	3	2	3	8	1
River Ehen (Ennerdale Water To Keekle Confluence)	86	0	3	4	2	1	2	2
River Frome	Uncertainty over catchment boundary							
River Itchen	20	12	7	29	3	27	2	0
River Kennet	15	74	1	2	1	6	1	0
River Kent And Tributaries	82	2	2	6	2	4	2	0
River Lambourn	26	64	1	1	1	6	1	0

River Lugg	34	57	0	1	3	2	3	0
River Mease	16	61	1	5	3	11	3	0
River Nar	30	55	1	2	3	6	2	1
River Teme (inc R. Clun)	46	40	0	1	5	2	6	0
River Test	45	11	3	15	6	14	6	0
River Wensum	9	81	1	1	1	6	1	0
River Wye	41	35	1	9	4	5	5	0
Slapton Ley	87	7	0	0	3	0	1	1
Trinity Broads	9	0	8	0	15	57	1	10
Tweed Catchment Rivers - England: Lower Tweed And Whiteadder	Uncertainty over catchment boundary							
Tweed Catchment Rivers - England: Till Catchment	Uncertainty over catchment boundary							
Upper Thurne Broads And Marshes	Uncertainty over catchment boundary							

Total nitrogen source apportionment

Table 7 presents the estimates of nitrogen source apportionment for the study sites. It should be noted that these estimates reflect total nitrogen delivery to the watercourses and do not take into account any instream processing. The highest contribution from agriculture (98%) was estimated for the River Test and Slapton Ley study catchments and the lowest (59%) for the River Dee (England) study area. In the case of TN contributions from STWs, the highest contribution (43%) was estimated for the River Kennet study area. Contributions of TN from diffuse urban sources ranged up to 1% (Ouse Washes, River Ehen, River Itchen, River Kent and tributaries). The corresponding contributions from storm tanks ranged up to 2% (River Itchen) and those from septic tanks up to 1% (River Beult). The highest TN contribution from CSOs (2%) was estimated for the Ouse Washes and River Mease study areas. Eroding channel bank contributions to TN loadings were up to 1%. Atmospheric deposition was consistently one of the smallest sources of TN in the study areas but was estimated to contribute 6% of this pressure on freshwater in the Bassenthwaite Lake study catchment.

Table 7: Total nitrogen source apportionment estimates for the study sites.

SSSI name	% TP from ag	% TP from STWs	% TP from urban	% TP from storm tanks	% TP from septic tanks	% TP from CSOs	% TP from bank erosion	% TP from atmospheric deposition
Ant Broads And Marshes	93	6	0	0	0	0	0	1
Aqualate Mere	96	2	0	0	0	0	1	1
Avon Valley (Bickton To Christchurch)	Uncertainty over catchment boundary							
Barnby Broad & Marshes	Uncertainty over catchment boundary							
Bassenthwaite Lake	91	2	0	0	0	0	1	6
Bure Broads and Marshes	97	2	0	0	0	0	0	1
1Chesil & The Fleet	Uncertainty over catchment boundary							
Hawes Water	Catchment area <10 km^2							
Hornsea Mere	Uncertainty over catchment boundary							
Leighton Moss	Catchment area <10 km^2							
The Mere, Mere	Catchment area <10 km^2							
Marazion Marsh	Uncertainty over catchment boundary							
Ouse Washes	77	18	1	1	0	2	0	1
Portholme	Uncertainty over catchment boundary							
River Avon System	87	10	0	1	0	1	0	1
River Axe	96	2	0	0	0	1	1	0
River Beult	91	5	0	0	1	1	1	1
River Camel Valley And Tributaries	97	0	0	1	0	1	0	1
River Dee (England)	59	39	0	0	0	1	0	1
River Derwent	96	3	0	0	0	0	0	1
River Derwent And Tributaries	92	2	0	0	0	0	1	5
River Eden And Tributaries	92	4	0	1	0	0	1	2
River Ehen (Ennerdale Water To Keekle Confluence)	95	0	1	1	0	0	1	2
River Frome	Uncertainty over catchment boundary							
River Itchen	95	1	1	2	0	1	0	0
River Kennet	56	43	0	0	0	1	0	0
River Kent And Tributaries	95	0	1	1	0	1	0	2
River Lambourn	86	13	0	0	0	1	0	0

River Lugg	87	12	0	0	0	0	1	0
River Mease	82	14	0	1	0	2	1	0
River Nar	97	2	0	0	0	0	0	1
River Teme (inc R. Clun)	75	23	0	0	0	0	1	1
River Test	98	0	0	1	0	1	0	0
River Wensum	75	23	0	0	0	1	0	1
River Wye	96	3	0	1	0	0	0	0
Slapton Ley	98	1	0	0	0	0	0	1
Trinity Broads	96	0	0	0	0	1	0	3
Tweed Catchment Rivers - England: Lower Tweed And Whiteadder	Uncertainty over catchment boundary							
Tweed Catchment Rivers - England: Till Catchment	Uncertainty over catchment boundary							
Upper Thurne Broads And Marshes	Uncertainty over catchment boundary							

Placing the source apportionment estimates in the national context

Water policy in England in a similar fashion to that in other parts of the world, places emphasis on the evidence base for informing the prioritisation of management investment (Keene and Pullin, 2011; Patterson et al., 2013). This intention of current policy to protect and enhance the aquatic environment for societal benefit demands new and improved tools for characterising the natural environment and the pressures thereon (Vlachopoulou et al., 2014). Cost-benefit proportionality exercises require reliable information on where sector-specific interventions such as those for agriculture should be targeted (Vinten et al., 2012). The new cross sector modelling framework described herein offers strategic scale updated water pollutant source apportionment estimates to help address these policy challenges by providing a means of screening shortlisted catchments across England and Wales to identify those where agricultural emissions of sediment and nutrients are dominant (i.e. ≥51%) compared to those from other sectors and sources.

For comparison, using the same modelling framework, national scale source proportions (with water body ranges) for sediment were recently estimated to be in the order; agriculture (72%, 0-100%) > river channel banks (22%, 0-96%) > diffuse urban (5%, 0-100%) > STWs (1%, 0-91%). This national scale source apportionment is consistent with that reported previously (Collins and Anthony, 2008; Collins et al., 2009a,b) but differs slightly on account of an updated model being used for the agricultural sector, a modified index for channel bank erosion and the fact that the monitored sediment concentrations for STW outfalls for the period 2010-2012 (10-30 mg/L) used by this study were lower than those used by the previous work (30-70 mg/L) reflecting stricter consents. The corresponding national scale estimates for total phosphorus (TP) were recently estimated to be; STWs (47%, 0-100%) > agriculture (31%, 0-100%) > CSOs (9%, 0-94%) > storm tanks (6%, 0-100%) > diffuse urban / septic tanks / river channel banks (all 2%, 0-100%, 0-70%, 0-71%) > direct atmospheric deposition (1%, 0-65%). Previous assessments have estimated the agricultural contribution to the national TP loading delivered to rivers to range between

18-24% (Anthony and Lyons, 2006), 23-28% (White and Hammond, 2006) and 43-53% (SDIA, 1989; Morse et al., 1993; Defra, 2004). Direct comparisons are hampered by various inconsistencies in terms of the sources appraised and the methods or tools used to estimate the loadings from specific sectors. Thus, for example, the work of Anthony and Lyons (2006) did not take into account TP emissions from septic tanks or from eroding channel banks and used a different pressure model for the agricultural sector. Anthony et al. (2008) and Duethmann et al. (2009) extended the work of Anthony and Lyons (2006) in a source apportionment exercise for P in lakes across England and Wales by including emissions from agriculture, channel banks, STWs, septic tanks, diffuse urban sources, groundwater and atmospheric deposition. In addition, P stripping has been, or is planned to be, introduced at more STWs (n= ~600) in response to the need to comply with the Urban Wastewater Treatment Directive and the Habitats Directive, thereby lowering the relative contribution from such point sources over time and critically since the above previous source apportionment studies. For total nitrogen (TN), the national scale source proportions were recently estimated to be in the order; agriculture (81%, 1-100%) > STWs (14%, 0-95%) > CSOs (1.5%, 0-73%) > direct atmospheric deposition (1.3%, 0-93%) > diffuse urban and storm tanks (both 1%, 0-80% and 0-93%) > septic tanks (0.2%, 0-30%) > river channel banks (~0%, 0-1%). %). Previous work by Hunt et al. (2004) estimated that agriculture contributed 60.6%, STWs 32.1%, other land 4.1%, direct industrial emissions 1.8%, septic tanks 0.7%, direct atmospheric deposition 0.4% and CSOs 0.3%. Subsequent work by Hughes et al. (2008) suggested that agriculture contributed 49.2% of TN delivery to freshwater in England and 59.5% in Wales, compared with corresponding respective contributions of 30.3% and 17.4% from sewage and industrial sources, 9.9% and 12.6% from woodland and natural areas, 6.0% and 3.4% from urban runoff and leaching, 3.8% and 6.0% from particulate sources and 0.7% and 1.0% from direct atmospheric deposition to water. Agricultural contributions were recently estimated to dominate water pollution by sediment in 76% (104, 434 km^2) of WFD cycle 2 non-coastal water bodies across England and Wales, compared to 58% (68, 434 km^2) in the case of TP and 93% (130, 384 km^2) in the case of TN. In combination, agricultural contributions of all three of these pollutants were recently estimated to be dominant in 53% (63, 030 km^2) of all WFD cycle 2 non-coastal water bodies.

Limitations of the pollutant source apportionment framework and its estimates

The outputs from the cross sector modelling framework need to be interpreted in the context of a number of limitations and uncertainties. A critical challenge in policy support is to integrate evidence at the national scale but with a view to providing information at local scale to help inform decision making. The outputs of the modelling framework are primarily designed to be summarised by WFD cycle 2 non-coastal water body (n = ~4500) but despite this, it is important to note that the results for individual catchments with areas <25 km^2 should be treated with caution. This limitation primarily reflects issues associated with the accuracy of statistical or regionally averaged data used to drive the emission layers including those for the agricultural sector. The pollutant emissions from the agricultural sector represent a baseline with no prior (i.e. current) implementation of mitigation methods. Ongoing work is estimating existing uptake of on-farm diffuse pollution interventions (e.g. in conjunction with agri-environment schemes) in order to improve the estimation of present day agricultural emissions. In contrast, the STW emissions are based on monitored data and so reflect the tightening of discharge consents and the gradual introduction of nutrient stripping (currently at or planned to be at ~600 works by 2015 in conjunction with the Urban Wastewater Treatment or Habitats Directives) over time. Nonetheless, there are significant uncertainties associated with the flow and concentration data for the smallest STWs and a national relationship between actual and consented discharges (n = 2393 pairings) for the period 2010-2012 has been used in conjunction with regionally averaged pollutant concentrations to calculate loads from this point source category. In addition, the database of STW locations includes multiple outfalls and permits for some works and these needed to be resolved in discussions with Environment Agency experts for the purpose of the load estimation. The temporal coverage of the estimates for the individual sectors or sources is not entirely consistent (e.g. 1991-2010 for agriculture, 2010-2012 for STWs) although the modelling framework does include the most up to date datasets available to researchers. The estimates of channel bank contributions to sediment and nutrient pressures do not take into account channel margin protection works. The cross sector modelling tool provides a framework for updating emissions or source apportionment as new datasets become available over time. The layers in the modelling framework represent pollution delivery to watercourses and do not take account of retention in the fluvial system, nor do they include nitrate inputs from groundwater (cf. Wang et al., 2012), although the latter will be available soon using coupled agricultural nitrate and groundwater modelling. The modelling framework provides primary source apportionment information as opposed to integrated pollutant budgets summarising the exchanges between sectors and media (cf. Leip et al., 2011 for integrated nitrogen budgets for European countries). The next stage in the development of the modelling framework will involve the construction of a pollutant concentration modelling function on the basis of structured regression modelling (e.g. Collins and Anthony, 2008) and this will include an estimate of retention in the fluvial system. The modelling framework does not currently include biogeochemical cycling.

Estimating modern background sediment pressures on the study sites

Although high sediment loads can lead to a deterioration in water quality, catchment-scale targets for sediment transport by European rivers do not exist (Collins and Anthony, 2008). The annual mean suspended sediment concentration of 25 mg L^{-1} cited by the European Freshwater Fish Directive (FFD) which was repealed in 2013. An annual average concentration of 25 mg L^{-1} equates to a sediment yield of 7.5 t km^{-2} yr^{-1}, assuming a runoff of 300 mm yr^{-1}. Collins and Anthony (2008) summarised the key problems associated with using the guideline annual mean suspended sediment concentration cited in the FFD as a global target for all of England and Wales. In addition, Collins et al. (2011) summarised the problems and uncertainties associated with existing international approaches to setting sediment targets using water column or river substrate metrics, suggesting that the use of sediment measures such as yields can overcome some, but not all, of these limitations.

Paleoenvironmental data can be used to establish sediment yields under reference conditions. Such information can provide a basis for quantifying the impact of catchment sediment mitigation strategies and for correcting the gap between current or future projected reductions in sediment pressures required to meet good ecological status (GES). Even an ideal mitigation programme should not endeavour to eliminate all fine sediment from rivers because healthy aquatic habitats require some fine sediment input to avoid homogenised and featureless river channel substrates (Yarnell et al., 2006). Estimates of background sediment pressure could therefore be taken to represent this ecological demand for sediment. This approach, albeit simplistic in terms of assessing safe sediment pressures for aquatic ecology, requires identification of a time in the sediment record that pre-dates the most significant recent phase of agricultural intensification in river catchments. An implicit assumption in using background sediment pressure metrics to set targets for catchment mitigation strategies is that there have been no changes in hydrological conditions or efficiency of the river network in delivering sediment to the recipient water body. Two key issues must be resolved to utilise paleoenvironmental reconstruction in the above context. First a time period that provides representative background conditions must be identified. Second, there must be sufficient lakes and reservoirs available that are spatially representative of landscape types and that are old enough to have experienced appropriate background conditions.

Bennion and Battarbee (2007) suggest that the early-to-middle 18th century may be the most appropriate period to represent ecological and chemical reference conditions for lakes. A similar time period was identified by Rose et al. (2011) in relation to sediment accumulation rates (SARs) for >200 European lakes. Rose et al. (2011) noted, however, that the most dramatic increases in sedimentation rate occurred after ~1950. Investigations of Holocene lake sediment in Britain show various phases of increased sediment accumulation that are strongly linked to agriculture during the Bronze Age, Iron Age and Romano-British periods, as well as during Medieval and Post-Medieval times (Edwards and Whittington, 2001; Chiverell et al., 2008). Similarly, Holocene floodplain accumulation rates in British rivers have been shown to be related to periods of rapid environmental change and vary spatially according to region and sedimentary environment (Macklin et al., 2009). The last 1,000 years have seen accelerated floodplain accumulation rates largely as a result of the agricultural revolution of the Middle Ages (Macklin et al., 2009). The sediment yield of a river over decades to centuries reflects climatic and anthropogenic drivers coupled with the internal regulation of storage and sediment delivery over a range of timescales (Foster, 2006; Trimble, 2010). Although the sediment yield of a river is a conceptually simplistic measure of the response of the river catchment to disturbance, it provides a quantifiable guide to the amount of sediment being transported by a river, which impacts directly on its ecological status.

Although Holocene sediment accumulation rates have been measured for natural lakes in Britain, these long-lived sedimentary basins are generally restricted to upland regions and are mostly a legacy of late Quaternary glaciations. The lowlands have few natural lakes, although reservoirs provide paleoenvironmental data for the last 100-150 years (Foster, 2006, 2010). From a practical perspective, establishing sediment conditions before the Medieval agricultural expansion as a ‘background target’ is not feasible for most of lowland Britain. It will be impossible to restore the agricultural landscape to the low-productivity system that characterised the region prior to the Medieval agrarian revolution, especially given current policy for food security. In a review of reconstructed sediment yields over the last 100-150 years, Foster (2006) noted that the most dramatic increase in sediment yields occurred after 1945. In the absence of longer-term records, the recent work of Foster et al. (2011) therefore proposed that values for the early 20th Century up to ~1940 should be used to establish provisional ‘modern background sediment pressures’ for catchments across England and Wales.

Sediment delivery to rivers and lakes decreases with increasing catchment size because there are opportunities for a greater proportion of the transported sediment to be deposited in intermediate stores. It is likely that estimates of modern background sediment yields based on small lake catchments should be further refined as a function of catchment area. Insufficient data are currently available to estimate long-term sediment storage and construct sediment budgets (Walling and Collins, 2008) for basins of contrasting size across England and Wales. On this basis, Foster et al. (2011) proposed that the estimates of modern background sediment yield should be taken as indicative of ‘modern background sediment delivery’ to watercourses for spatial extrapolation and up-scaling purposes and should not be considered representative of net downstream yields in larger drainage basins.

Mapping modern background sediment delivery to rivers across England and Wales

Only a limited number of late Holocene sediment yield reconstructions have been undertaken across England and Wales. The recent work of Foster et al. (2011) reviewed these studies for the purpose of identifying estimates of ‘modern background sediment delivery to rivers’ (MBSDR) that might help define catchment sediment targets in line with the EU WFD. The intention of this work as part of Defra project WQ0128 (Collins et al., 2012) was to help find a measure that could substitute the EU FFD, which was repealed in 2013. Large areas of England and Wales, however, have no sediment yield reconstructions and, on that basis, it proved necessary to extrapolate the available yields in order to produce tentative national MBSDR maps.

On the basis of the synthesis of lake-based sediment yield reconstructions available for England and Wales undertaken by Foster et al. (2011), and bearing in mind the relatively high uncertainty associated with extrapolation across large areas of the UK where no reconstructed data are available, two categories of MBSDR were defined and mapped as part of Defra project WQ0128:

1. The ***target modern background sediment delivery to rivers*** (TMBSDR) is the recommended target based on best scientific knowledge.
2. The ***maximum modern background sediment delivery to rivers*** (MMBSDR) has been introduced in order to recognise uncertainty in the sediment yield reconstruction and the extrapolation of these data to un-gauged areas across England and Wales.

Table 8 presents the estimates of MBSDR for England and Wales produced on the basis of the work during Defra project WQ0128 (cf. Foster et al., 2011). For the majority of the lake-based case studies reviewed, it was possible to estimate MBSDR on the basis of the dominant land use in the upstream catchment. For national spatial extrapolation of these values, the 1km Raster data summary of the Land Cover Map (LCM2000) (Fuller et al. 2002) was used to assess land cover in each 1 km^2 across England and Wales. The widespread broad habitat classes were grouped into five types (Table 9) compatible with the land cover criteria given in Table 8 and consistent with the LCM2000 land cover for the lake catchments.

Table 8: Estimates of TMBSDR and MMBSDR for England and Wales.

Land use category	TMBSDR ($t\ km^{-2}\ yr^{-1}$)	MMBSDR ($t\ km^{-2}\ yr^{-1}$)
Forested	<5	10
Mixed forest / moorland / upland rough grazing	<5	10
Upland moorland / rough grazing	<5	15
Peat	<<50	65
Lowland agriculture (A)	<10	15
Lowland agriculture (B)	<20	35

Table 9: Sub and aggregate classes from the CEH LCM2000 used to derive percentage cover of forest, peat, upland moorland and lowland agriculture for each 1 km^2 across England and Wales.

LCM 2000 Broad habitat classes	Class for sediment yield mapping	Note
12	Peat	Bog
13,17,18,19,20,21,22 and unclassified	Non-applicable area	No estimate of sediment yield
8,10,15,16	Upland moorland/rough grazing	Mountain, heath and acid grassland
4,5,6,7,9,11	Intensive agriculture	Arable, horticulture and grassland (excluding acid grassland)
1 and 2	Forest	Broadleaved/mixed and coniferous woodland

Extrapolation of MBSDR is more problematic in the case of lowland agricultural catchments given the wide range in possible MBSDRs indicated by the existing lake studies. Evans (1990) proposed a typology of the susceptibility of soils to water erosion for England and Wales based on soil associations (Mackney et al., 1983; Table 10) and slope gradient (> or < 4 degrees). Steep slopes were taken to be representative of more incised landscapes with greater slope-channel connectivity. This typology was used for extrapolating the lake-based MBSDR estimates for lowland agricultural catchments. Soil associations identified as being at very low, low or moderate erosion risk by water were used in the definition of lowland agricultural MBSDR category A (Table 8). Soil associations identified as being at high or very high risk of accelerated soil erosion by water (Table 10) were used to define lowland agricultural MBSDR category B (Table 8). Soil associations were mapped using the NATMAP vector soil map (National Soil Resource Institute, Cranfield University). For consistency with the current soil erosion risk typology used by the Department for Environment, Food and Rural Affairs (Defra, 2005), a slope of 3 degrees was used as the slope threshold separating categories A and B with the median slope for each 1 km^2 being assessed using a 50 m digital elevation model (DEM). Target and maximum MBSDR were then calculated as a weighted average based on the proportion of each of the land cover categories in each kilometre square. Finally, non-applicable areas (i.e. more than 90% classed as open waterbody or unclassified) and urban areas were removed from the spatial extrapolation using a corresponding GIS mask layer (Shepherd and Bibby, 2004).The tentative maps of MBSDR produced on the basis of the above extrapolation scheme are presented in Figure 2. These estimates should be interpreted with some caution since no historical sediment yield data are currently available for large areas of England and Wales. The variability between target and maximum MBSDR is relatively narrow, with the majority of pre-war reconstructed sediment yields falling within 5-10 t km^{-2} yr^{-1}. Exceptions are associated with steeper upland areas in the Lake District, Pennines and some areas of Wales and Dartmoor. However, these rather limited areas result largely from the absence of data, as does the interpretation of the southeast and east of England.

Table 10: Soil association risk typology (Evans, 1990).

Erosion risk	Soil associations
Very small risk of erosion by water	22, 342d, 346, 411a, 411c, 421a, 421b, 511b, 511i, 512a, 512b, 512c, 512d, 512e, 512f, 532a, 532b, 541a, 541i, 541v, 541w, 541B, 561b, 561c, 572t, 573a, 581a, 581b, 581c, 581d, 612a, 641b, 643a, 643c, 643d, 711b, 711c, 711d, 711f, 711g, 711h, 711k, 711m, 711p, 711r, 711s, 711t, 712a, 712b, 712c, 712d, 712e, 712f, 712g, 712h, 712i, 713b, 713c, 713d, 713e, 713f, 713g, 714a, 714b, 714c, 714d, 811a, 811b, 811c, 811d, 811e, 812a, 812b, 812c, 813a, 813b, 813c, 813d, 813e, 813f, 813g, 813h, 814a, 814b, 814c, 831a, 831c, 832, 841b, 841c, 841d, 871b, 871c, 92a, 1011a
Small risk of erosion by water	313b, 341, 342a, 342c, 343c, 343e, 343i, 411b, 411d, 431, 511a, 511c, 511d, 511f, 511j, 541d, 541f, 541g, 541h, 541j, 541k, 541l, 541n, 541o, 541p, 541q, 541u, 541x, 541y, 541z, 541C, 541D, 542, 543, 555, 571a, 571g, 571l, 571m, 571n, 571p, 571r, 571s, 571t, 571u, 571v, 571w, 571z, 571A, 572a, 572b, 572d, 572f, 572g, 572h, 572i, 572j, 572l, 572n, 572o, 572q, 572r, 581e, 581f, 581g, 582a, 582b, 582c, 582d, 611b, 631b, 631c, 631e, 631f, 634, 643b, 711a, 711e, 711i, 711j, 711l, 711n, 711o, 711q, 711u, 711v, 711w, 713a, 841a, 841e, 92b
Moderate risk of erosion by water	342b, 343a, 343b, 343d, 343g, 343h, 511b, 511e, 511g, 513, 541c, 541e, 541r, 541t, 544, 551g, 552b, 561a, 561d, 571b, 571c, 571f, 571h, 571i, 571j, 571k, 571o, 571q, 571x, 571y, 572c, 572e, 572k, 572m, 572p, 572s, 573b, 582e, 631d, 641a, 92c
High risk of erosion by water	541b, 541m, 541s, 551c, 551e, 554a, 571d, 571e
Very high risk of erosion by water	541A, 551a, 551b, 551d

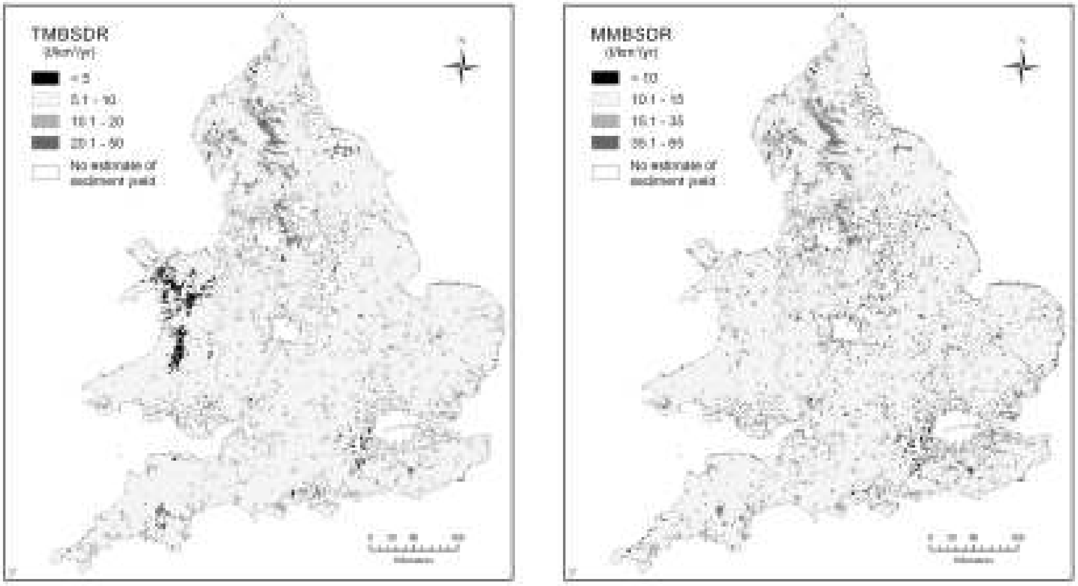

Figure 2: Nationally extrapolated estimates of TMBSDR and MMBSDR.

Sediment gap analysis

Table 11 presents the estimated gap (as a ratio) between current agricultural sediment stress and either the TMBSDR or MMBSDR for the study sites. This gap analysis must be interpreted in the context of various limitations and uncertainties including those noted above for the nationally extrapolated estimates of background sediment pressures across England and Wales. The results in Table 11 present a mixed picture with the greatest gaps between current agricultural sediment loss and TMBSDR (ratios of 7.9 and 5.6) being estimated for the River Ehen (Ennerdale Water to Keekle confluence) and River Kent and tributaries study sites, respectively. Both the Bassenthwaite Lake and Slapton Ley study sites are also estimated to have significant gaps (ratio of 4.5) between current agricultural sediment loadings and the TMBSDR (Table 11). The corresponding gaps between current agricultural sediment delivery to river channels and the MMBSDR are 4.8 (River Ehen – Ennerdale to Keekle confluence), 3.7 (River Kent and tributaries), 2.9 (Slapton Ley) and 2.7 (Bassenthwaite Lake). For the dataset presented in Table 11 as a whole, the estimated gap between current agricultural sediment pressure and TMBSDR ranged between 0.2 – 7.0 and for MMBSDR between 0.1 and 4.8. Given that the gap ratios are less than 1.0 in some cases, for both TMBSDR and MMBSDR, it is likely that even with non-agricultural sediment pressures on watercourses taken into account, the background thresholds may not be exceeded. As the study sites were selected as those suffering from sediment stress, further work is required to resolve the relationship between estimated background sediment thresholds and alternative standards or targets for screening pollutant pressures and compliance for designated sites.

Table 11: Estimated gap (as a ratio) between current agricultural sediment delivery to the study sites and TMBSDR or MMBSDR.

SSSI name	Ratio versus TMBSDR	Ratio versus MMBSDR
Ant Broads And Marshes	0.3	0.2
Aqualate Mere	0.6	0.4
Avon Valley (Bickton To Christchurch)	Uncertainty over catchment boundary	
Barnby Broad & Marshes	Uncertainty over catchment boundary	
Bassenthwaite Lake	4.5	2.7
Bure Broads and Marshes	0.3	0.2
Chesil & The Fleet	Uncertainty over catchment boundary	
Hawes Water	Catchment area <10 km^2	
Hornsea Mere	Uncertainty over catchment boundary	
Leighton Moss	Catchment area <10 km^2	
The Mere, Mere	Catchment area <10 km^2	
Marazion Marsh	Uncertainty over catchment boundary	
Ouse Washes	1.1	0.7
Portholme	Uncertainty over catchment boundary	
River Avon System	0.7	0.5
River Axe	3.4	2.2
River Beult	2.0	1.3
River Camel Valley And Tributaries	2.1	1.3
River Dee (England)	Confidence issues with current pressure estimate	
River Derwent	1.2	0.7
River Derwent And Tributaries	4.1	2.5
River Eden And Tributaries	1.5	1.0
River Ehen (Ennerdale Water To Keekle Confluence)	7.9	4.8
River Frome	Uncertainty over catchment boundary	
River Itchen	0.8	0.5
River Kennet	0.8	0.5
River Kent And Tributaries	5.6	3.7
River Lambourn	0.6	0.4
River Lugg	2.5	1.6
River Mease	0.6	0.4
River Nar	0.4	0.2
River Teme (inc R. Clun)	2.2	1.4
River Test	0.7	0.4
River Wensum	0.5	0.4
River Wye	2.5	1.6
Slapton Ley	4.5	2.9
Trinity Broads	0.2	0.1
Tweed Catchment Rivers - England: Lower Tweed And Whiteadder	Uncertainty over catchment boundary	
Tweed Catchment Rivers - England: Till Catchment	Uncertainty over catchment boundary	
Upper Thurne Broads And Marshes	Uncertainty over catchment boundary	

Conclusions

The continued limited success of policy measures to reverse water quality and ecological decline underscores the need for updated tools to support catchment screening and improved targeting of mitigation measures for specific sectors including agriculture. The cross sector modelling framework described herein represents a useful tool for assisting this drive towards improved targeting and will support the newly launched stakeholder-led catchment-based approach (CaBA) for improving the management of the aquatic environment in England (Environment Agency, 2012; Defra, 2013). The gap analysis must be interpreted with due caution given the uncertainties and limitations associated with the nationally extrapolated estimates of TMBSDR and MMBSDR.

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