Zhang, Y., Collins, A. L., Jones, J. I., Johnes, P. J., Inman, A. and Freer, J. E. 2017. The potential benefits of on-farm mitigation scenarios for reducing multiple pollutant loadings in prioritised agri-environment areas across England. *Environmental Science & Policy.* 73 (July), pp. 100-114.

Environmental Science and Policy 73 (2017) 100–114

Contents lists available at ScienceDirect

Environmental Science and Policy

journal homepage: www.elsevier.com/locate/envsci

The potential benefits of on-farm mitigation scenarios for reducing multiple pollutant loadings in prioritised agri-environment areas across England

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ARTICLE INFO

Keywords:
Diffuse pollution
Mitigation
Agri-environment
Cost-benefits
Uncertainty

ABSTRACT

Mitigation of diffuse water pollution from agriculture is a key national environmental policy objective in England. With the recent introduction of the new agri-environment scheme, Countryside Stewardship, there is an increased emphasis on the macro-spatial targeting of on-farm mitigation measures to reduce pollutant pressures, and a concomitant need to forecast the technically feasible impacts of on-farm measures detailed in current policy and their associated costs and benefits. This paper reports the results of a modelling application to test these limits in the context of the associated costs and benefits for the reduction of diffuse water pollution from agriculture for each Water Framework Directive (WFD) water management catchment (WMC) and nationally. Four mitigation scenarios were modelled, including pollutant source control measures only (SC), mobilisation control measures only (MC), delivery control measures only (DC) and measures for source, mobilisation and delivery control (SMDC) combined. Projected impacts on nitrate, phosphorus and sediment export to water, ammonia, methane and nitrous oxide emissions to the atmosphere, together with the associated costs to the agricultural sector were estimated for each WFD WMC and nationally. Median WMC-scale reductions (with uncertainty ranges represented by 5th–95th percentiles) in current agricultural emissions, were predicted to be highest for the SMDC scenario; nitrate (18%, 11–23%), phosphorus (28%, 22–37%), sediment (25%, 18–43%), ammonia (26%, 17–32%), methane (13%, 7–18%) and nitrous oxide (18%, 16–20%). The median benefit-to-cost ratios (with uncertainty ranges represented by 5th–95th percentiles) were predicted to be in the following order; DC (0.15, 0.09–0.65), MC (0.19, 0.09–0.95), SMDC (0.31, 0.20–1.39) and SC (0.44, 0.19–2.48). Of the four scenarios simulated, the SC and SMDC suites of measures have the greatest potential to deliver reductions in BAU emissions from agriculture, and the best benefit:cost ratio.

1. Introduction

It has long been recognised that emissions from agriculture result in the excess loadings of multiple pollutants on receiving freshwaters across England (Johnes and Burt, 1991; Heathwaite et al., 1996; Carpenter et al., 1998; McGonigle et al., 2012; Houses of Parliament, 2014a), and on increasing rates of gaseous emission to the atmosphere (Sutton et al., 1995; Skiba et al., 1997; Misselbrook et al., 2000; Houses of Parliament, 2014b). Policy approaches for controlling this pollution in England include the promotion of voluntary codes of good practice, incentivised schemes and regulation. The intention is that these approaches, in combination, alleviate environmental damage by agricultural diffuse pollution and thereby lessen the corresponding external costs to society. Incentivised schemes are best represented by agri-environment initiatives which have increasingly encouraged the uptake of combinations of on-farm measures to tackle significant pollutant pressures and to help deliver multiple policy objectives including the protection of natural resources and the maintenance of ecosystem services (Boatman et al., 2008).

Built on the evolving knowledge of the efficacy and associated costs of on-farm mitigation measures (cf., Cooke and Petch, 2007; Cherry et al., 2008; Balana et al., 2011; Schoumans et al., 2014), integrated modelling approaches have been increasingly applied as a means of combining hydrology and nutrient flux simulations with economic scenarios (Gomann et al., 2005; Mainstone et al., 2008; Moreau et al., 2012; Bouraoui and Grizzetti, 2014), to account for measure depen-

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http://dx.doi.org/10.1016/j.envsci.2017.04.004
Received 9 January 2017; Received in revised form 11 April 2017; Accepted 11 April 2017
Available online 26 April 2017

dency and competition (Gooday et al., 2014, 2015), and the potential for pollutant swapping (Collins and McGonigle, 2008; Stevens and Quinton, 2009) or co-benefits (Johnes et al., 2007; Verspecht et al., 2012; Greene et al., 2015; Collins et al., 2016). Alongside these developments in modelling approaches for agricultural diffuse water pollution, the concept of the water pollutant transfer continuum, i.e. source-mobilisation-delivery-impact (Lemunyon and Gilbert, 1993; Haygarth et al., 2005), has been adopted widely for structuring the assessment of water pollution risk, designing mitigation strategies and targeting monitoring for the estimation of mitigation impacts (Kronvang et al., 2009; Wall et al., 2011; McGonigle et al., 2014; Murphy et al., 2015; Bloodworth et al., 2015; Zhang et al., 2017). The prohibitive costs associated with universal or blanket implementation of numerous on-farm mitigation measures, mean there is a growing trend towards the optimisation of on-farm mitigation measure selection using cost-effectiveness (Haygarth et al., 2007; Gooday et al., 2014) or farmer attitudes (Collins et al., 2016). Much previous work has compared the potential benefits of blanket measure application versus spatial targeting to address critical source areas (Johnes et al., 2007; Strauss et al., 2007; Collins and Davison, 2009; Doody et al., 2012; Shore et al., 2014). It is now widely accepted that the spatial variability of agricultural pollutant pressures has to be considered implicitly in the design of robust and cost-effective mitigation strategies (e.g. Anthony et al., 2012; Zhang et al., 2012; Greene et al., 2015; Jones et al., 2016).

In England, the Department of Environment, Food and Rural Affairs (Defra) has recently (January 2016) introduced the new Countryside Stewardship (CS) scheme which aims to 'protect and enhance the natural environment, in particular the diversity of wildlife (biodiversity) and water quality' (Countryside Stewardship, 2015). This new agri-environment scheme has identified priority areas which require on-farm mitigation to meet the environmental objectives associated with various national and international policy drivers, including the WFD, Bathing Water Directive, Sites of Special Scientific Interest and Natura 2000 designations, and surface or ground water safeguard zone delineations. To provide knowledge-based evidence on the technically feasible impact of new mitigation scenarios in association with the new CS scheme, several theoretical scenarios were constructed using interventions targeting the different stages of the water pollutant transfer continuum and evaluated with a national scale modelling framework. The modelling framework uses Farmscoper (FARM SCale Optimisation of Pollutant Emission Reductions) which was initially developed by ADAS UK Ltd. for the evaluation of mitigation impacts on pollutant reductions at farm scale (Zhang et al., 2012; Gooday et al., 2014). The tool has been scaled up and validated at catchment (Zhang et al., 2012) and national levels (Collins et al., 2016; Collins and Zhang, 2016) and continues to be used extensively in support of UK agri-environmental policy. This paper reports a national scale application for England, with the modelled outputs being summarised at WFD WMC scale (Environment Agency, 2015), to support the ongoing re-design of on-farm mitigation strategies since a mid-term review of CS is scheduled in 2018. Preliminary efforts were also made to explore the uncertainty ranges for the predicted efficacy of the policy scenarios tested here (e.g. Collins et al., 2016).

2. Methodology

The key procedures involved in the quantification of potential mitigation strategy cost-effectiveness at WFD WMC scale using the national scale Farmscoper modelling framework have been described in detail elsewhere (Collins et al., 2014, 2016; Collins and Zhang, 2016). In brief, the framework is underpinned by a number of national layers based on farm survey and census data, process-based modelling of agricultural pollutant losses and IPCC models (Fig. 1). More general background on the Farmscoper tool is provided in Supplementary information (SI).

2.1. Mapping agricultural pollutant pressures for the CS priority areas

The areas of high and moderate priority for CS options across England related to water quality for the period 2015–2021 were provided by the Environment Agency (Countryside Stewardship Water Quality Priority Areas v5, October 2014, Chris Burgess, pers. comm., 16 March 2015). For each designated priority area, the presence of pollutant pressures, including nutrient, sediment, pesticides, FIO and dissolved oxygen concentrations together with, river hydrology and morphology, was assessed at the site level. In the study presented here, the focus was on nitrate, phosphorus and sediment flux to waters, since agriculture is considered to be a significant contributor to these pressures (cf. Zhang et al., 2014), together with nitrous oxide, methane and ammonia emissions as key emissions from agriculture to the atmosphere. The model is currently set up for nitrate rather than nitrogen, owing to existing policy drivers for the control of nitrate pollution in waters, such as the EU Nitrates Directive (91/676/EEC). This does mean that the total impact of nitrogenous pollution from agriculture on freshwater ecosystems (see Durand et al., 2011 for a review of these other forms and impacts) is not included in this analysis.

Using ArcGIS software, the CS designated priority areas (Fig. 2) and corresponding key pollutant pressures were intersected with the WFD WMC boundaries to generate new spatial data layers of pollutant emissions. The agricultural land areas comprising CS priority zones within each WFD WMC were also determined. Although the modelling scenarios focussed on CS priority areas for water quality protection, the modelling framework simultaneously computes the costs and benefits of on-farm interventions for multiple pollutants of water and air, since many measures impact simultaneously on both receptors, enabling the potential for pollution swapping to be taken into account explicitly.

2.2. Selection of on-farm mitigation measure combinations

The concept of the diffuse pollution transfer continuum from land to water suggests that the translocation of pollutants from agricultural sources to receiving aquatic environments involves mobilisation and delivery along multiple pathways: for water these include natural flow pathways to and in conjunction with groundwater, as overland or quickflow, or via artificial (e.g. tile) drainage. In many locations, a combination of these flow paths exists. In the Farmscoper simulation tool, delivery pathways to water are characterised as leaching to groundwater, runoff (surface or shallow quickflow), preferential flow (e.g. via macropores/cracks) or direct (e.g. incidental losses). All pollutant loadings and mitigation impacts are evaluated on an annual basis. Though some monthly variations are implicitly represented, there is no explicit characterisation of event-based dynamics, i.e. storm processes for either pollutant emissions or mitigation efficacy.

On-farm mitigation measures in Farmscoper (n = 105) were reviewed for their relevance to water quality. Seven measures were considered to have insignificant potential benefits at national scale. These were: install air-scrubbers or biotrickling filters in mechanically ventilated pig housing, more frequent manure removal from laying hen housing with manure belt systems, in-house poultry manure drying, irrigate crops to achieve maximum yield, protection of in-field trees, irrigation/water supply equipment is maintained and leaks repaired and use high sugar grasses.

The remaining 98 measures were further assessed and assigned to three mutually exclusive groups targeting the stages of the water pollutant transfer continuum: source control measures (SC), mobilisation control measures (MC) and delivery control measures (DC). Another theoretical combination of measures (SMDC) included all three of the above sets as a means of assessing the maximum potential impacts of combined measures targeting the land to water continuum. The four theoretical mitigation scenarios used for modelling (SC, MC, DC and SMDC) included 59, 18, 21 and 98 on-farm measures, respectively (Table 1). The scenarios assumed measure uptake rates

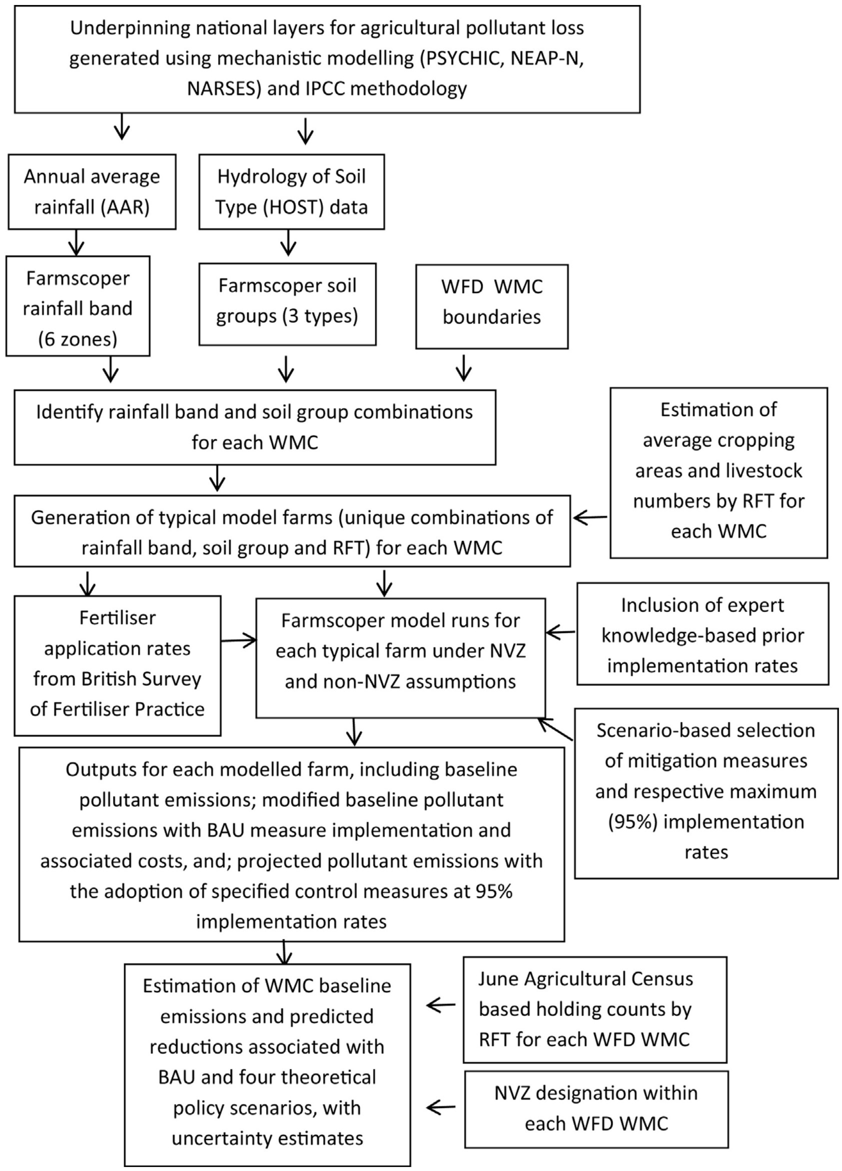

Fig. 1. Fundamental elements of data flow for running the national scale Farmscoper framework. PSYCHIC: Phosphorus and Sediment Yield Characterisation In Catchments model; NEAP-N: National Environment Evaluation of Agricultural Pollution-Nitrate model; NARSES National Ammonia Reduction Strategy Evaluation System; IPCC: Intergovernmental Panel on Climate Change; WFD: EU Water Framework Directive; WMC: Water Management Catchment; RFT: Robust Farm Type; NVZ: Nitrate Vulnerable Zone; BAU: Business-as-usual.

of 95%, i.e. those measures relevant to each robust farm type (RFT; see SI for more information) included in the modelling framework are applied to 95% of the relevant areas on a given model farm. This uptake ceiling is used by UK policy teams in the evaluation of maximum potential policy impacts. For England, the national modelling framework includes 5917 model farms, comprising 685 cereals, 701 general cropping, 699 horticulture, 679 specialist pig, 682 specialist poultry, 685 dairy, 386 LFA grazing, 701 lowland grazing and 699 mixed farms. These model farms have specific attributes (e.g. rainfall, soil, cropping areas and livestock numbers) but their distribution within the relevant WMC is random as opposed to the farms having precise landscape positions therein, i.e. all farms have equal opportunity to be within the available rainfall and soil combinations and any NVZ areas comprising any individual WMC.

For scenario evaluation in the new CS priority areas, the proportions of agricultural land area inside Nitrate Vulnerable Zone (NVZ) designations were mapped (see SI for more detail). Mandatory regulations apply to NVZ areas regarding manure handling and the timing of manure/fertiliser applications to crops and grass, which affects the uptake rates of those mitigation measures covered by the mandatory rules. Hence, different prior implementation rates for NVZ and non-NVZ areas were implemented in the modelling framework.

2.3. Estimation of the efficacy of the four new mitigation scenarios targeting the agricultural water pollution continuum

Each CS priority area has its own environmental pressures arising from agriculture and typically these comprise multiple, rather than lone, pollutants. To account for these variations, the total predicted environmental benefits (*EB*) of the four theoretical policy scenarios were estimated using:

$$EB = \Sigma_{i=1}^{n} P_i BL_i LR_i C_i \quad (1)$$

Where i is the list of pollutants (water quality and gaseous emissions) relevant to a given CS priority zone, and P is the proportion of agricultural land area being affected by the specific agricultural pollutants. While mapped pressures were used to specify the proportions of agricultural land area prioritised for individual water quality-related pollutants, all agricultural land areas were assumed to be generating gaseous emissions, i.e. P is set to 1.0. BL is the total loading

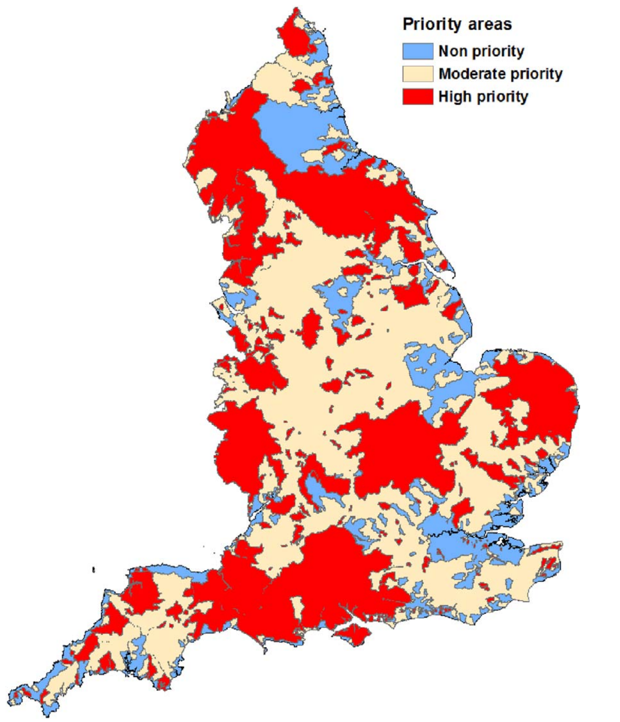

Fig. 2. Mapped priority areas across England for the new Countryside Stewardship (CS) scheme.

with business-as-usual (BAU) implementation levels of on-farm mitigation measures (see SI for more information on assumptions here). *LR* is the predicted reduction in pollutant load (relative to BAU). C_i is the environmental damage cost for each specific pollutant, calculated by multiplying the estimated reduced pollutant emissions under each of the four theoretical policy scenarios by unit (for 2013) damage costs provided by Defra policy teams (0.97 £ kg^{-1} $year^{-1}$, 33.16 £ kg^{-1} $year^{-1}$, 0.39 £ kg^{-1} $year^{-1}$, 2.79 £ kg^{-1} $year^{-1}$,1.20 £ kg^{-1} $year^{-1}$ and 18.60 £ kg^{-1} $year^{-1}$ for nitrate, phosphorus, sediment, ammonia, methane and nitrous oxide, respectively). On the basis of Eq. (1), the environmental benefit is equivalent to the damage costs avoided due to the implementation of the on-farm mitigation measures included in each of the four scenarios. The estimated benefits were all normalised by agricultural land area to derive specific annual benefits. These benefits were then compared against the total costs of each on-farm mitigation scenario to derive indicative benefit-to-cost ratios. As the pre-selected sets of mitigation measures (Table 1) focus primarily on the pollutant delivery continuum to from agricultural land to water and, thereby, the scope for reducing loads delivered to aquatic environments, the proportion of the estimated benefits which are specifically water quality-related (i.e. sediment, phosphorus and nitrate) were also estimated, alongside the combined benefits for mitigating losses to both water and air.

2.4. Quantifying uncertainty associated with the modelled predictions

While the most comprehensive national data for farm business structures (livestock numbers and crop areas from the 2010 June Agricultural Survey) and current (BAU) uptake of mitigation measures were used to construct the model farms in the national framework deployed here, some assumptions and generalisations were made for the national scale simulations. These included, but are not restricted to, a random distribution of model farms in different rainfall and soil bands (see SI for details on these bands), uniform rates of fertiliser application across different areas, the relative proportions of manure spread to land across different areas, and the geo-referencing and allocations of 2010 June Agricultural Survey data at WFD WMC scale. To date, sensitivity

Table 1
Grouping of on-farm mitigation measures by components of the water pollutant transfer continuum.

Measures for source control (SC)
Make use of improved genetic resources in livestock
Use plants with improved nitrogen use efficiency
Fertiliser spreader calibration
Use a fertiliser recommendation system
Integrate fertiliser and manure nutrient supply
Do not apply manufactured fertiliser to high-risk areas
Avoid spreading manufactured fertiliser to fields at high-risk times
Use manufactured fertiliser placement technologies
Use nitrification inhibitors
Replace urea fertiliser to grassland with another form
Replace urea fertiliser to arable land with another form
Incorporate a urease inhibitor into urea fertilisers for grassland
Incorporate a urease inhibitor into urea fertilisers for arable land
Use clover in place of fertiliser nitrogen
Do not apply P fertilisers to high P index soils
Reduce dietary N and P intakes: Dairy
Reduce dietary N and P intakes: Pigs
Reduce dietary N and P intakes: Poultry
Adopt phase feeding of livestock
Reduce the length of the grazing day/grazing season
Extend the grazing season for cattle
Reduce field stocking rates when soils are wet
Move feeders at regular intervals
Construct troughs with concrete base
Increase scraping frequency in dairy cow cubicle housing
Additional targeted bedding for straw-bedded cattle housing
Washing down of dairy cow collecting yards
Frequent removal of slurry from beneath-slat storage in pig housing
Increase the capacity of farm slurry stores to improve timing of slurry applications
Adopt batch storage of slurry
Install covers to slurry stores
Allow cattle slurry stores to develop a natural crust
Anaerobic digestion of livestock manures
Minimise the volume of dirty water produced (sent to dirty water store)
Minimise the volume of dirty water produced (sent to slurry store)
Compost solid manure
Site solid manure heaps away from watercourses/field drains
Store solid manure heaps on an impermeable base and collect effluent
Cover solid manure stores with sheeting
Use liquid/solid manure separation techniques
Use poultry litter additives
Manure Spreader Calibration
Do not apply manure to high-risk areas
Do not spread slurry or poultry manure at high-risk times
Do not spread FYM to fields at high-risk times
Calibration of sprayer
Fill/Mix/Clean sprayer in field
Avoid PPP application at high risk timings
Drift reduction methods
PPP substitution
Construct bunded impermeable PPP filling/mixing/cleaning area
Treatment of PPP washings through disposal, activated carbon or biobeds
Plant areas of farm with wild bird seed/nectar flower mixtures
Uncropped cultivated areas
Unfertilised cereal headlands
Use dry-cleaning techniques to remove solid waste from yards prior to cleaning
Capture of dirty water in a dirty water store
Monitor and amend soil pH status for grassland
Increased use of maize silage
Measures for mobilisation control (MC)
Establish cover crops in the autumn
Early harvesting and establishment of crops in the autumn
Cultivate land for crops in spring rather than autumn
Adopt reduced cultivation systems
Cultivate compacted tillage soils
Leave autumn seedbeds rough
Loosen compacted soil layers in grassland fields
Allow grassland field drainage systems to deteriorate
Use slurry band spreading application techniques
Use slurry injection application techniques

(continued on next page)

Table 1 (continued)

Measures for mobilisation control (MC)
Incorporate manure into the soil
Unharvested cereal headlands
Undersown spring cereals
Leave over winter stubbles
Leave residual levels of non-aggressive weeds in crops
Use correctly-inflated low ground pressure tyres on machinery
Avoid irrigating at high risk times
Use efficient irrigation techniques (boom trickle, self-closing nozzles)

Measures for delivery control (DC)
Cultivate and drill across the slope
Manage over-winter tramlines
Establish in-field grass buffer strips
Establish riparian buffer strips
Ditch management on arable land
Ditch management on grassland
Fence off rivers and streams from livestock
Construct bridges for livestock crossing rivers/streams
Re-site gateways away from high-risk areas
Farm track management
Establish new hedges
Establish and maintain artificial wetlands − steading runoff
Establish tree shelter belts around livestock housing
Management of woodland edges
Management of in-field ponds
Management of arable field corners
Beetle banks
Uncropped cultivated margins
Skylark plots
Management of grassland field corners
Locate out-wintered stock away from watercourses

analysis has not been undertaken.

The following approach (Fig. 3) was adopted to quantify the uncertainty associated with predicted agricultural pollutant load reductions:

a) Within the Farmscoper modelling framework, the efficacy and prior uptake of individual mitigation measures were represented by ranges where minimum and maximum values were specified (see SI for more detail). The mid-points of the specific ranges allocated to individual measures were used to estimate both BAU and mitigated (using the four scenarios) emissions from agriculture.
b) On the basis of a) above, pollutant-specific 5th and 95th percentile reductions were generated for each RFT (9 in total; see SI for detail) included in the modelling framework by treating all model farms for any given RFT as the population. It is recognised that this might not represent the full range of the 5th and 95th percentiles and that the limited number of replicates could also lead to certain averaging effects.
c) The estimated 5th and 95th percentile reductions, relative to BAU, were then assigned to the RFTs present within each WFD WMC to calculate the overall lower and upper uncertainty ranges in percentage reductions for the catchment as a whole.

Using this approach, the uncertainty associated with the predicted impacts of the theoretical scenarios were scaled linearly from mitigation measures (using ranges on the uncertainty for the efficacy of measures − see SI) to farm type (using rules for the applicability of individual mitigation measures to different farming systems) to landscape unit (WFD WMC) scale, using information on the combinations and numbers of RFTs present (Fig. 3).

3. Results and discussion

3.1. Mapping problem areas with excessive agricultural pollutant emissions

The high and moderate priority designated areas under the new CS scheme cover 53, 715 km^2 and 57, 909 km^2 of agricultural land in England, respectively (Fig. 2). These areas account for 41% and 44% of the national land area. Within these targeted areas (high and moderate priorities combined), pollutant pressure mapping by the Environment Agency (Chris Burgess, pers. comm., 16 March 2015) suggested that 33% of agricultural land suffers from nitrate loading in excess of EU Nitrate Directive thresholds (noting that these do not represent ecologically relevant thresholds for freshwaters), 83% from excess phosphorus loadings and 48% from excess sediment loadings. Within the high priority areas alone, excess loadings have been estimated by the Environment Agency to affect 48%, 87% and 73% of agricultural land in the case of nitrate, phosphorus and sediment, respectively. On this basis, phosphorus and sediment pressures are designated as being more widespread than those arising from agricultural nitrate emissions; the latter are most important as an issue in groundwater-dominated areas (Wang et al., 2016). However, the thresholds adopted elsewhere in the EU for nitrogen, including the threshold recommended by Durand et al. (2011) of 2 mg L^{-1} NO_3-N, suggest that current pressure assessment based on nitrate loadings alone has the potential to underestimate the total nitrogenous pressure from agriculture on aquatic environments by up to 50%. It should be noted, therefore, that this analysis focuses purely on nitrate pollution in relation to the EU Nitrate Directive standards, and should not be interpreted as indicating the total diffuse agricultural pressure in relation to ecological impacts.

The national scale CS designations for prioritisation level (high and medium) were mapped onto individual WFD WMC areas and the relative proportions of agricultural land area identified by the Environment Agency as suffering from nitrate, phosphorus and sediment pollution were calculated (Fig. 4). There was significant spatial variability for all three water pollutants at this management unit scale (Fig. 4). For the 93 WFD WMCs comprising England, the median proportions of land under pressure from nitrate, phosphorus and sediment were 8%, 50% and 19%, respectively. These values indicate that water pollutant pressures are generally localised at WMC scale even though pressures from phosphorus are more widely distributed than those arising from excessive agricultural loadings of nitrate and sediment. Using 5% of the agricultural land area producing excess loadings as a minimum threshold for the pressures which need to be tackled using on-farm interventions, it was estimated that there are 52, 88, and 77 WFD WMCs across England which are experiencing high nitrate, phosphorus and sediment pressures, respectively. Nearly half of the WFD WMCs have pressures from all three water pollutants examined here, under a third have pressures from a combination of sediment and phosphorus, and the remaining WMCs are suffering from other combinations or single pressures. To represent these spatial variations and highlight the spatial coincidence of different pressures, pressure layers for agricultural nitrate, phosphorus and sediment were combined into a compound pressure index (CPI) using the following formula:

$$CPI = \Sigma_{i=1}^{n}(HP_i + 0.5\ MP_i)/n \quad (2)$$

Where i is the individual agricultural pollutant, HP is the proportion of land under the individual pollutant pressure within high priority areas and MP is the proportion of land under the individual pollutant pressure within moderate priority areas. In the absence of any detail from the Environment Agency on the criteria for differentiation between high and moderate priority areas, a subjective scaling factor of 0.5 was used to account for the lower level of priority given to the moderate priority areas. CPI can be used to indicate the presence of higher proportions of land in designated high priority areas for the CS scheme which have pressures from more than one agricultural water pollutant (Fig. 5). The

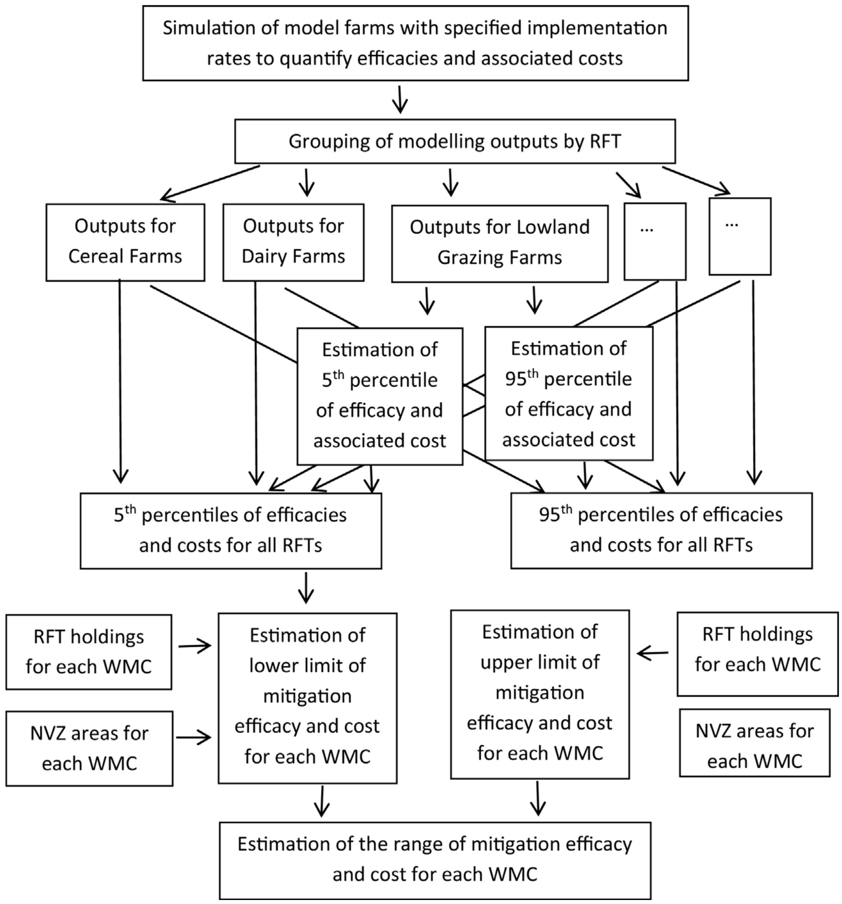

Fig. 3. Schematic data flow for the quantification of the uncertainty ranges for mitigation efficacy and costs at WMC-scale.

presence of pressures from multiple agricultural water pollutants underscores the need for targeted mitigation strategies to consider co-benefits and to avoid pollution swapping issues. The choice of scaling factor (0.5) in Eq. (2) will have impacts on the calculated CPI, especially if a WMC is dominated by moderate priority areas. It will not, however, affect the identification of WMCs dominated by high priority and multiple pollutant pressures.

3.2. Evaluation of the modelled BAU agricultural pollutant emissions

The agricultural sector is the dominant source of sediment and nitrate, and a significant source of phosphorus loading to freshwater across England and Wales (Morse et al., 1993; Hughes et al., 2008; Zhang et al., 2014; Greene et al., 2015). In rural catchments, agriculture is the single dominant source of sediment, nitrate and phosphorus export to waters and of agricultural gaseous emissions to the atmosphere. As part of a similar modelling exercise which evaluated the potential impacts of farmer preferred mitigation measures, Collins et al. (2016) compared the predicted BAU (2010) agricultural loadings of sediment and nitrate for different WFD river basin districts (RBDs) with PARCOM monitoring data (1991–2010). Neal and Davies (2003) provide more background information on the PARCOM data. In the case of agricultural GHG emissions to air, the simulated BAU emissions of methane and nitrous oxide were compared by Collins et al. (2016) with corresponding official UK GHG inventories from agriculture for 2013 at RBD scale (more information can be found at https://www.gov.uk/government/statistics/final-uk-emissions-estimates). These comparisons suggested good agreement, in terms of the spatial patterns across the RBDs, for all pollutants evaluated, but identified a systematic over-prediction by the national scale modelling framework for nitrous oxide emissions.

To assess the predicted BAU emissions further, observed sediment, nitrate and phosphate concentrations for the period 1980–2010 from 33 monitoring stations comprising part of the Harmonised Monitoring Scheme (HMS) were combined with estimated river discharge for the respective catchment areas to derive annual loads. Considering the accessibility and relevant spatial scale of national datasets, the river discharges for the relevant HMS catchments were based on flow estimates for 2010 using the CEH Low Flows software. The flow data took account of both natural flow and external influences, e.g. abstraction. Since Farmscoper only generates long-term mean annual outputs, average monitored concentrations were used to calculate average annual loads. 2010 was characterised by normal seasonal contrasts in flow conditions but without any extreme flood events (Marsh and Sanderson, 2011). It is therefore considered to be an average year for river flow. The 33 HMS catchments (Fig. 6) all met the following selection criteria: a) at least 10 estimates of water quality parameter concentrations were available, b) the monitoring stations had close proximity to a specified catchment outlet which has measured flow data available, c) the absence of significant point source contributions (e.g. from large sewage treatment works), and d) catchment areas of > 25 km^2 (reflecting lower confidence in modelled pollutant pressures for smaller areas; see Zhang et al., 2014). Modelled BAU loads for sediment, nitrate and phosphate were compared with HMS data, indicating good agreement for nitrate and phosphate and strong correlation but a systematic bias for sediment (Fig. 7) as indicated by the deviation of predicted from measured loads and the divergence from the corresponding 1:1 line. Using simulated and measured data without transformation, Kling-Gupta efficiencies (Gupta et al., 2009) were estimated at 0.65, 0.3 and 0.1 for nitrate, phosphate and sediment, respectively. The low efficiency for sediment was mainly caused by significant bias error. Similar results have been reported

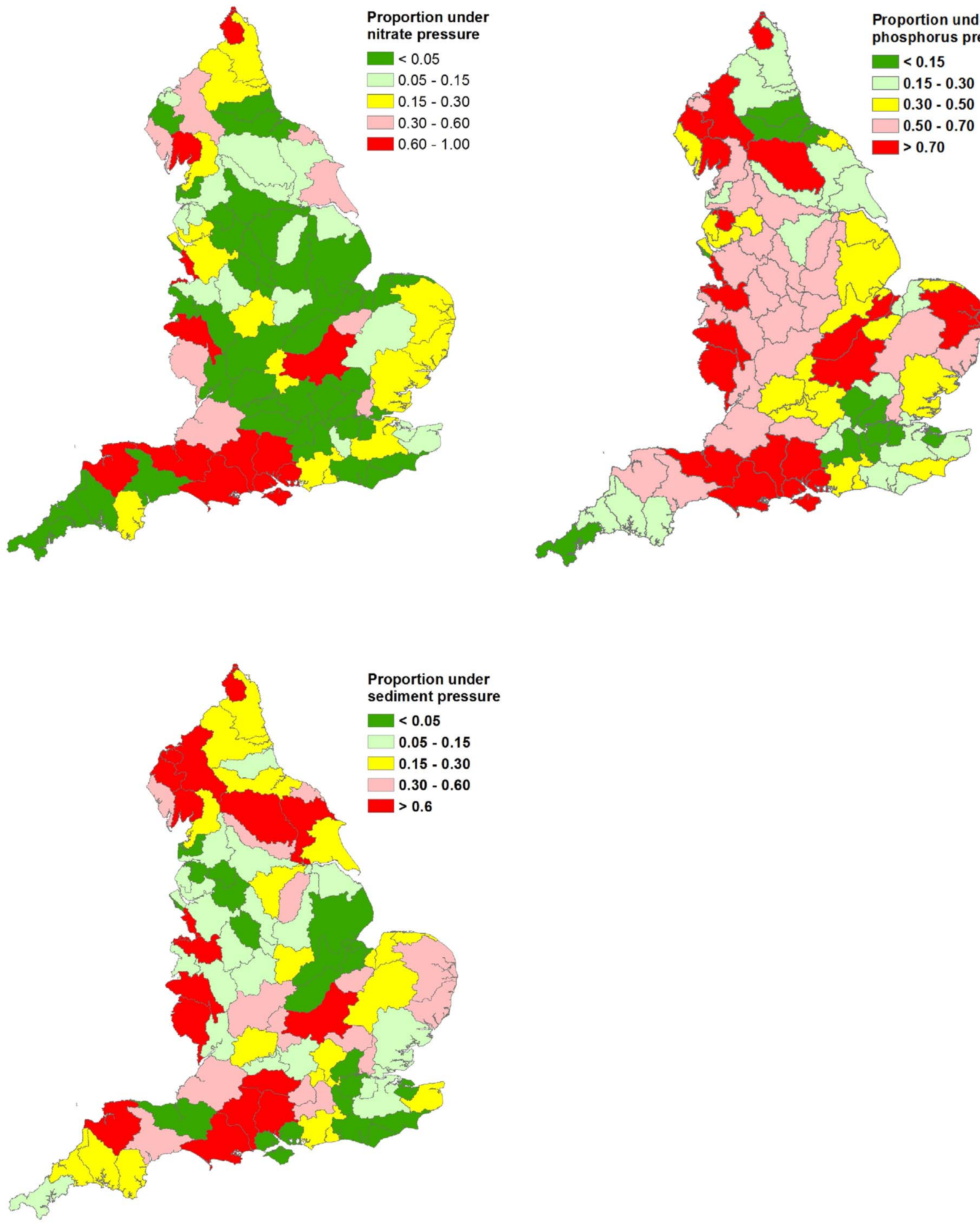

Fig. 4. Environment Agency defined proportions of each WFD WMC across England experiencing specific agricultural pollutant pressures to be tackled by the new CS scheme.

previously for an evaluation exercise at RBD scale (Collins et al., 2016) for sediment and nitrate. The comparison of modelled BAU and HMS sediment loads suggests that over-prediction was more severe in catchments with low measured annual loads (Fig. 7), corresponding to lowland parts of the country and HMS catchments with areas > 40 km^2. Here, it is important to note that the modelling framework does not currently take into account landscape long-term sediment retention associated with floodplain deposition and, therefore, the modelled BAU loads would be expected to over-predict the corresponding HMS estimates, which are net of such landscape storage. Uncertainties associated with the HMS water quality data which are based on routine low temporal resolution sampling (Walling et al., 1992) and the scale of the catchments for which HMS data were used to estimates loads should also be borne in mind here (Phillips et al., 1999), especially for pollutants like sediment, the delivery of which is rainfall dependent and thereby dominated by a few major storm events during each hydrological year (Horowitz, 2008). The comparison of phosphorus loads was further complicated by the fact that measured and modelled loads are not necessarily the same fraction. Due to the data availability issue, no efforts were made to account for potential uncertainty associated with the flow estimates (Preston et al., 1992; Lloyd et al., 2016) which is also important in addition to uncertainties associated

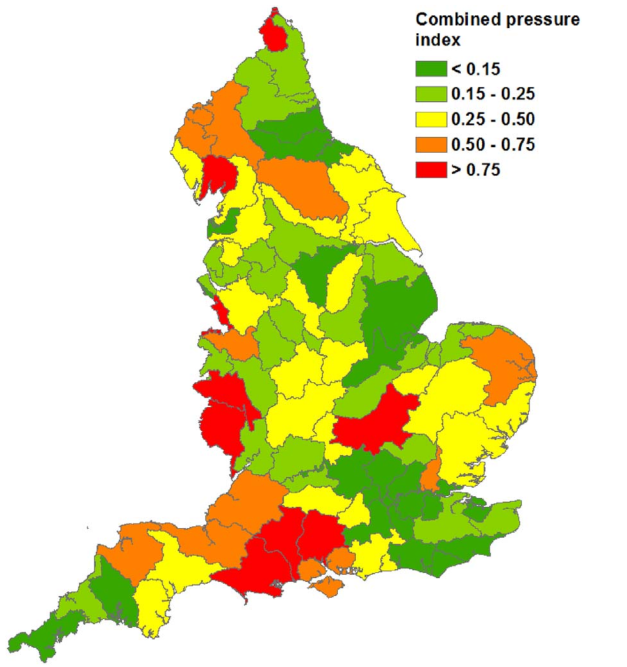

Fig. 5. The spatial distribution of the combined agricultural pollutant pressure index (CPI).

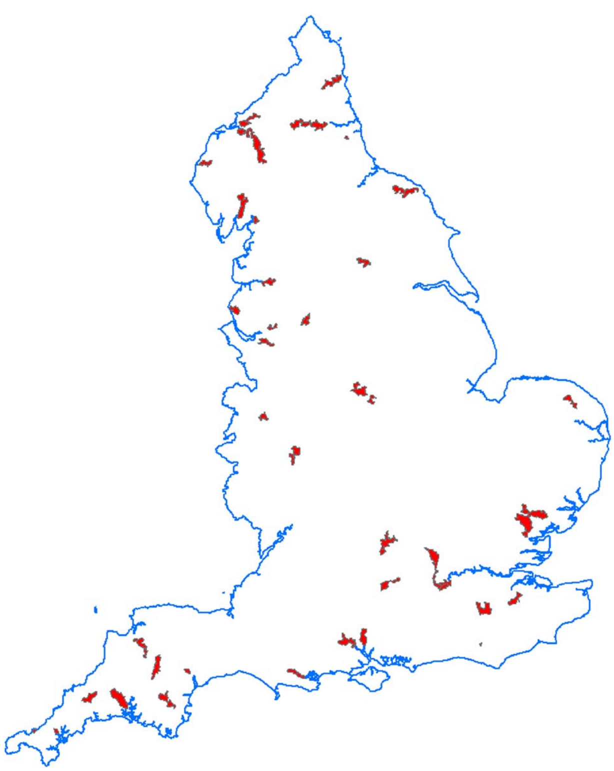

Fig. 6. The spatial distribution of the HMS catchments used in the evaluation of the modelled BAU emissions to water.

with low frequency pollutant concentration data. The discrepancy between the flow and pollutant concentrations in term of temporal spans (estimated flow for 2010 and average concentrations for the period between 1981 and 2010) also complicates the comparisons.

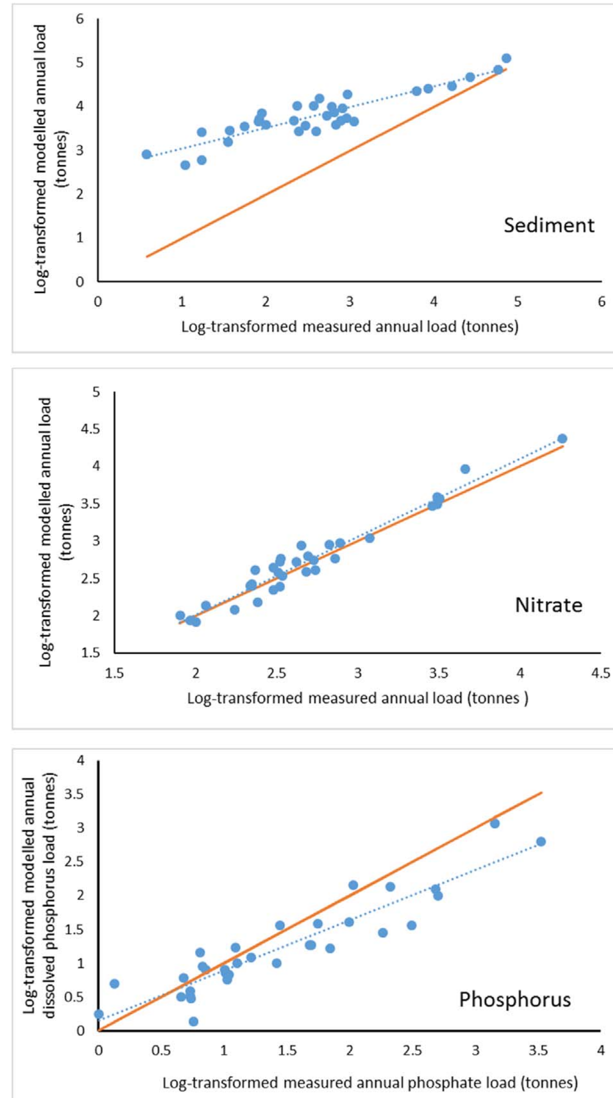

Fig. 7. Comparison of modelled BAU and measured HMS loads of sediment, nitrate, and phosphorus using the catchments shown in Fig. 6 where the solid line in red is the 1:1 line for modelled and measured loads. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

3.3. Predicted WMC-scale agricultural pollutant load reductions relative to BAU

Summary statistics for the agricultural pollutant load reductions predicted for each scenario for the 99 WMCs were determined at the national scale (i.e. across all WFD WMCs: Table 2). Non-parametric descriptive statistics (median and percentiles) were used for the data summaries to lessen the influence of extreme values and violation of normality assumptions. The distributions of the modelled predictions used to generate the summary statistics were based on > 90 samples, reflecting the presence of most RFTs (n = 9; see SI) in each of the WMCs. A full list of the predicted pollutant reductions associated with each of the four scenarios for each individual WFD WMC can be found in the online SI.

The modelled outputs show that the SC scenario is much more effective in reducing BAU agricultural pollutant loadings to water and air compared with either the MC or DC scenarios (Table 2). For the SC scenario, the median loading reductions across all pollutants were approximately 15–20%. For the MC and DC scenarios, the corresponding median pollutant load reductions were mostly < 10%. The most substantial differences between the predicted impacts of the theoretical

Table 2
Summary statistics for the predicted percentage reductions in BAU pollutant loads at WFD WMC-scale.

	Nitrate	Phosphorus	Sediment	Ammonia	Methane	Nitrous Oxide
Source control (SC)						
Minimum	8.2	15.8	9.4	1.4	1.7	11.2
Maximum	21.9	31.8	38.6	44.5	26.7	22.2
Median	16.2	19.5	15.7	17.8	15.4	17.9
P5*	11.8	16.4	11.2	5.2	7.6	15.8
P25*	14.7	18.1	14.4	10.6	13.3	17.1
P75*	17.9	21.9	18.7	27.5	18.7	18.7
P95*	21.4	27.9	26.3	37.6	22.6	19.5
Variability**	59.4	58.7	96.5	181.4	97.6	20.4
Mobilisation control (MC)						
Minimum	-1.5	3.9	4.5	0.7	0.0	-2.6
Maximum	7.2	21.1	25.8	18.4	0.0	4.6
Median	3.3	9.9	10.9	7.3	0.0	0.7
P5	0.3	5.5	5.8	2.0	0.0	-1.2
P25	2.4	7.8	8.1	4.4	0.0	-0.1
P75	4.6	12.9	13.1	10.8	0.0	1.8
P95	5.8	16.3	18.3	15.7	0.0	3.1
Variability	171.2	109.2	115.1	189.0	NA	577.9
Delivery control (DC)						
Minimum	1.0	3.2	4.9	0.2	0.0	1.3
Maximum	3.7	23.9	44.0	1.4	0.0	4.3
Median	2.2	7.3	11.1	0.5	0.0	2.9
P5	1.5	4.1	6.1	0.3	0.0	1.8
P25	1.8	6.1	8.2	0.4	0.0	2.6
P75	2.5	10.2	16.8	0.7	0.0	3.3
P95	3.2	15.0	25.6	1.1	0.0	3.8
Variability	77.5	148.4	175.1	150.7	NA	69.0
All measures (SMDC)						
Minimum	9.3	20.1	15.9	6.0	1.7	10.7
Maximum	24.7	45.6	58.0	35.0	21.1	22.1
Median	17.9	27.7	25.1	26.3	13.3	18.1
P5	11.2	21.5	18.4	16.6	7.1	16.4
P25	16.1	24.8	21.6	23.9	11.8	17.4
P75	20.2	31.3	31.2	28.9	15.7	19.0
P95	22.9	37.3	42.8	31.8	18.4	20.4
Variability	65.4	57.3	97.0	57.6	85.3	21.7

* P5, P25, P75 and P95 are calculated percentiles at 5%, 25%, 75% and 95%, respectively.

** Variability is calculated as the difference between P95 and P5 divided by the corresponding median value.

scenarios concerned technically feasible reductions for gaseous emissions. Both the MC and DC scenarios were predicted to have no impact on agricultural methane emissions, minor impact on ammonia emissions (< 1% for the MC, 7% for DC with an uncertainty of 2–16%) and mixed impact on nitrous oxide losses. In the case of the MC scenario, there is a possibility that emissions of nitrous oxide could increase for some WMCs, i.e. pollution swapping could occur (see online SI for results for individual WMCs). MC measures generally encourage retention of pollutants in the source area, e.g. 'Allow grassland field drainage systems to deteriorate'. While this might decrease downstream delivery, it has the potential to increase the likelihood of pollutant transformation and gaseous emissions. Taken together, the adoption of the theoretical SC, MC or DC scenarios is predicted to result in median WMC-scale reductions of BAU sediment loadings of 11% (uncertainty range 6–18%) to 16% (uncertainty range 11–26%). Similar reductions were predicted for agricultural phosphorus loads, ranging from a minimum of 7% (uncertainty range 4–15%) to 20% (uncertainty range 16–28%). The modelling generally suggested very low changes in BAU nitrate loads for the MC and DC scenarios (< 4%, uncertainty range 0–6%), whereas the SC scenario was predicted to deliver a greater load reduction (median 16%, uncertainty range 12–21%). These results reaffirm the importance of on-farm pollutant source control options for delivering co-benefits (both water quality improvement and gaseous emission reduction). The relatively high variability (Table 2) predicted for both the MC and DC scenarios suggests that more attention should be given to detailed site-specific assessments on farms to target the individual measures relevant for these components of the pollutant cascade to water.

For the SMDC scenario, the predicted reductions in BAU agricultural pollutant loads, were generally higher than those for the three theoretical policies discussed above (Table 2), as the SMDC scenario involved the implementation of the most comprehensive set of on-farm measures. Here, it is noteworthy that the most substantial reductions were predicted for sediment, phosphorus and ammonia relative to the other scenarios discussed above. This suggests that the increased implementation of on-farm interventions along the whole transfer continuum, i.e. a 'treatment train' approach (McGonigle et al., 2014), will be more beneficial for the mitigation of excessive water quality pressures.

There was considerable spatial variation in the predicted median emission reductions for the different agricultural pollutants associated with the implementation of SC measures (Fig. 8: see Supplementary information Fig. S1–S3 for corresponding maps for the other theoretical scenarios). In general, a higher effect of mitigation on nitrate and ammonia was predicted in the eastern part of England where arable farming systems dominate the agricultural landscape. The spatial patterns of the predicted effect of mitigation were consistent for sediment and phosphorus (Fig. 8), reflecting the close association of these pollutants in agricultural settings (Foster et al., 2003; Neal et al., 2006): the highest predicted effects of mitigation on sediment and phosphorus were in the western parts of England where higher rainfall, steeper slopes and the predominance of livestock farming systems mean that these agricultural water pollutants are more important (Collins and Anthony, 2008; Zhang et al., 2014).

3.4. Predicted WMC-scale uncertainty ranges for agricultural pollutant load reductions relative to BAU

The work flow in Fig. 3 also provided an opportunity for summary statistical analysis of the uncertainty ranges for the predicted reductions in BAU pollutant loads to water and air (Table 3). These results suggest that, at national scale, and across the four mitigation scenarios, the predicted reductions for the mitigation of sediment and phosphorus, and especially the former, are the most uncertain. Uncertainty in the predicted reductions in BAU sediment loads, was not evenly distributed across the country (see Fig. 9 for sediment and Figs. S4–S9 for the other agricultural pollutants). The relatively lower national scale variation in the uncertainties associated with the predicted impacts of the four scenarios for the mitigation of gaseous emissions (Table 3) can be attributed to the fact that the spatial heterogeneity of local physical environmental conditions is less likely to be reflected in the models for gaseous emissions underpinning the Farmscoper tool (Zhang et al., 2012). In contrast, the underpinning process-based models used for predictions of water quality pollutants, and especially sediment and phosphorus, do capture such variations in environmental controls which, in turn, play an important role in driving the more accentuated variations across the uncertainties for the technically feasible impacts of the four theoretical mitigation scenarios (Table 3).

3.5. Mitigation scenario costs to farmers and environmental benefits

The Farmscoper framework estimates two types of costs associated with the implementation of suite of measures by farmers, namely fixed and variable costs, which are combined to give total costs. The annual costs and uncertainty ranges for each theoretical mitigation scenario are presented in Table 4. Costs for the MC and DC scenarios were typically lower than those for the SC and SMDC scenarios, since the latter two involve many more measures (18 to 21 vs 59 to 98, respectively). The estimated median total annual costs to farmers were predicted to be in the following ascending order: £34 ha^{-1} yr^{-1} for

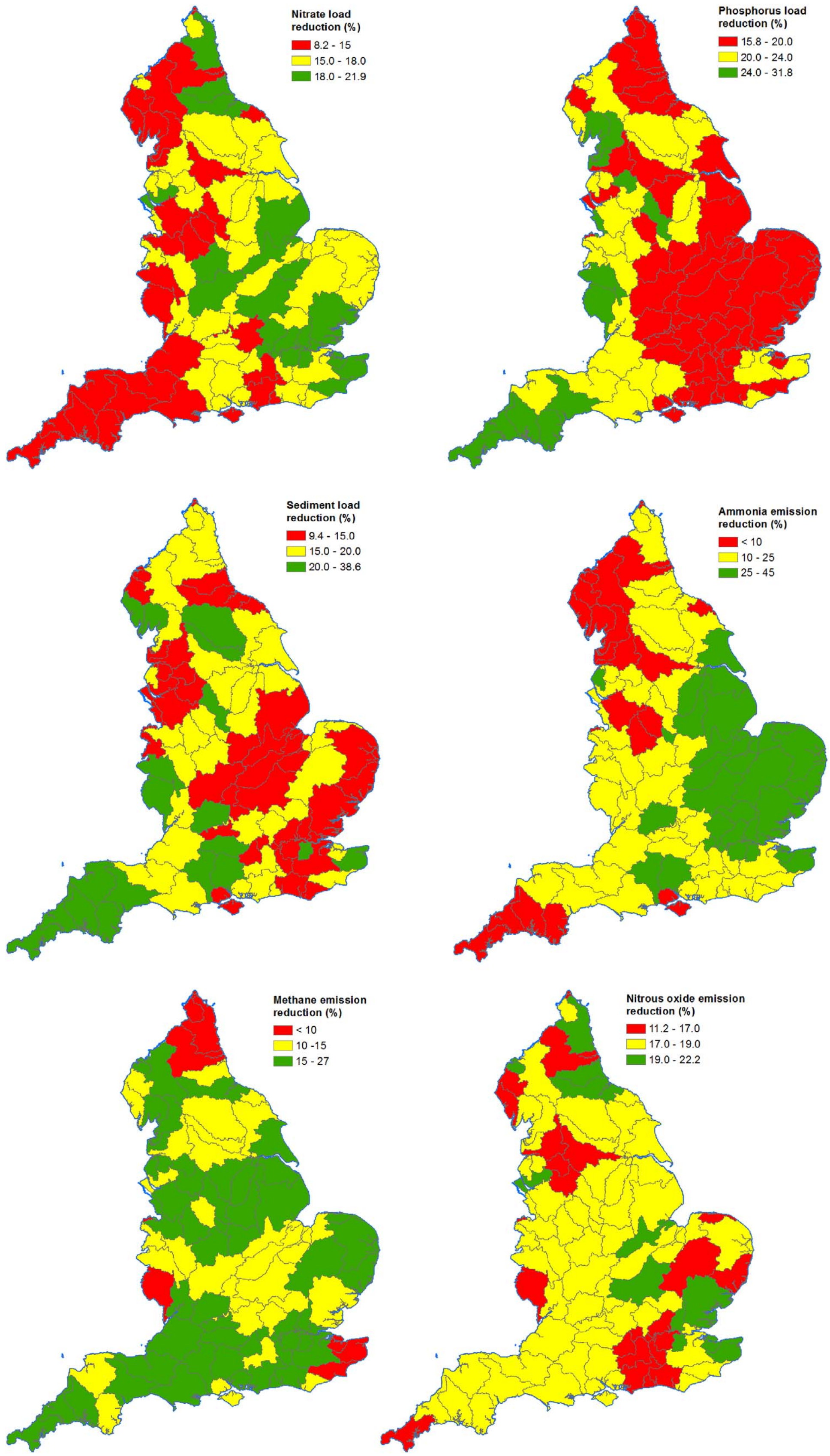

Fig. 8. Estimated median reductions in BAU annual agricultural pollutant loads, predicted for the SC measure scenario.

Table 3
Summary statistics for the predicted uncertainty ranges associated with the percentage reductions in BAU pollutant loads at WFD WMC-scale.

Scenario	Summary statistics	Nitrate	Phosphorus	Sediment	Ammonia	Methane	Nitrous Oxide
Source control (SC)	Median	5.4	9.3	12.3	7.9	4.5	4.4
	P5*	5.2	7.7	10.0	6.3	2.7	3.3
	P95*	5.6	11.6	15.2	10.0	10.9	7.7
Mobilisation control (MC)	Median	4.6	10.1	13.3	6.2	0.0	3.4
	P5	3.8	7.4	11.8	3.9	0.0	2.8
	P95	6.2	10.8	14.8	8.3	0.0	4.2
Delivery control (DC)	Median	1.6	11.1	16.9	0.7	0.0	2.1
	P5	1.2	8.5	11.8	0.5	0.0	1.8
	P95	2.1	12.3	18.9	1.4	0.0	2.6
All measures (SMDC)	Median	7.9	19.3	25.6	7.5	4.5	4.8
	P5	7.2	15.5	24.9	6.5	2.7	3.5
	P95	9.6	22.3	26.2	8.8	10.9	8.4

* P5 and P95 are calculated percentiles at 5% and 95%, respectively.

DC < £38 ha^{-1} yr^{-1} for MC < £66 ha^{-1} yr^{-1} for SC < £129 ha^{-1} yr^{-1} for SMDC. Most of the estimated annual costs were fixed in the case of the SC scenario, whereas variable costs were more important for the MC scenario, reflecting the types of measures involved (Table 1).

To contextualise the predicted annual costs to farmers further, they were examined relative to the expected economic value of the environmental benefits that could be anticipated to arise from increased deployment of the on-farm mitigation measures (*c.f.* BAU). Environmental damage costs were used to assign lumped monetary values to the non-market outcomes of agricultural pollution with respect to a range of ecosystem goods and services associated with provision of drinking water, biodiversity and public amenity (e.g. bathing waters). For each WMC, environmental costs were calculated using Eq. (1) and then divided by corresponding total costs (fixed and variable combined) to derive a benefit-to-cost ratio (Table 5). The benefit-to-cost ratios were found to vary considerably among the WFD WMCs for the different theoretical scenarios under scrutiny. The spatial distribution of benefit-to-cost ratios for the SC scenario are shown in Fig. 10, where the ratios estimated using both costs for the relevant RFTs within an individual WMC and the corresponding 95th percentile costs for the relevant RFTs at national scale were taken into account. The latter could be treated as the worst case scenario. If only the 5th percentile costs (the best case scenario) were used, the benefit-to-cost ratios would all be > 1 for all WMCs. Benefit-to-cost ratios were highest for the SC scenario followed by the SMDC scenario, with estimated median values of 0.44 and 0.31, respectively (Table 5). The MC and DC scenarios had much lower benefit-to-cost ratios (Table 5). Using the fixed pollutant costs (2013 prices), wide uncertainty ranges suggested by the 5th − 95th percentiles, highlight the necessity for further spatial targeting of on-farm measures within the identified risk areas for the new CS scheme. The 95th percentile predictions for environmental benefit-to-costs for the SC and SMDC scenarios (Table 5) suggest it is possible to achieve desirable outcomes, but the scope for such outcomes is clearly predicted to be highly uncertain. Among the expected environmental benefits, predicted median values for the water quality benefit proportion are less than 0.5 for all scenarios, i.e. more than half of the environmental benefit will be attributable to the co-benefits arising from the reduction of modelled emissions of ammonia, methane and nitrous oxide. The highest proportions of water quality benefit are predicted for the MC (0.35, 0.11-0.37) and DC (0.37, 0.05–0.80) scenarios (Table 5). Mobilisation control options (Table 1) are typically spatially expansive in that they potentially need to cover multiple fields on any given farm. Farmers will therefore need appropriate skills and tools to help with such targeting. Delivery control options are frequently easier to target, as these intervene at critical pollution delivery points to the river system which can be easily identified by walkover surveys.

While benefit-to-cost ratios have been tabulated to compare the different mitigation scenarios, it is important to acknowledge that damage costs have their own uncertainty and are still subject to debate and revision. Although the costs and benefits have been reported here in absolute terms, they are still only indicative and best suited for comparing the different mitigation scenarios in relative terms. Accordingly, only national scale summary statistics for the benefit-to-cost ratios are provided herein, rather than estimates for individual WMCs.

3.6. Limitations of the modelling work

The above results should be interpreted in the context of a number of limitations and uncertainties. Some of these limitations are discussed in Collins et al. (2016) and Collins and Zhang (2016). One limitation of this approach is that it simulates nitrate rather than total nitrogen loading, which underestimates the total diffuse nitrogenous impact on waters in livestock farming regions by up to 50% (Durand et al., 2011). The specific diffuse N forms not, then, included in this assessment include the particulate and dissolved organic N delivered to waters from livestock wastes which generate both N enrichment and organic pollution impacts in streams, together with ammoniacal and nitrite, both of which are toxic at low concentrations to aquatic organisms (Durand et al., 2011). Second, the reliance on the assessment of pressures by the Environment Agency means that the nitrate threshold applied is relevant for drinking water, and not for ecological relevance. The results of this analysis should therefore be interpreted in this context. Another limitation relates to the quality of the datasets underpinning this analysis. The modelling framework is founded on numerous national-scale spatial datasets, each of which has different degrees of spatial accuracy and temporal coverage. The mitigation measures for the different modelling scenarios may not reflect the increasingly exhaustive list of measures being implemented on the ground, since only those measures which have been characterised in terms of efficacy, applicability and cost are included. The current mitigation costs in the modelling framework are based on values discussed and agreed with UK Government policy teams: these costs should be viewed as highly generalised due to the assumption of nationally representative uniform values for the measures concerned. The costs of measure implementation have the potential to vary both spatially and temporally. Additionally, the administration costs of policy instruments are not included in the cost summaries which instead, reflect only the costs to farmers. The discrepancy between modelled BAU and monitored sediment loads at selected sites, due to the non-representation of catchment processes including floodplain deposition in the modelling framework, could have implications for the estimated environmental benefits, i.e. the estimated benefits could be inflated for areas with extensive active floodplains and corresponding low measured downstream suspended sediment yields. Some prelimin-

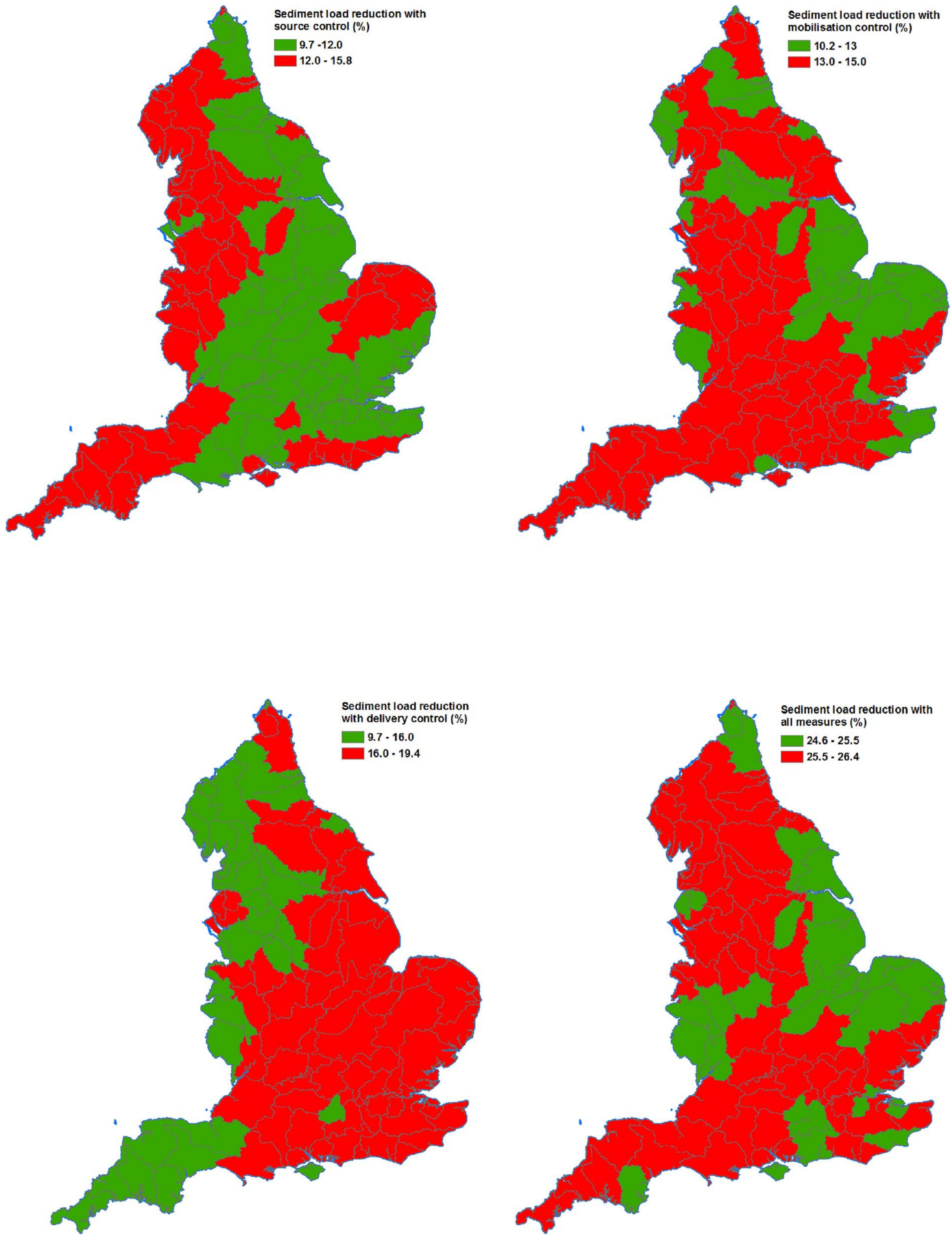

Fig. 9. Estimated uncertainty ranges (5th-95th percentiles) for the predicted percentage reductions in BAU agricultural sediment loads, under the different mitigation scenarios.

ary uncertainty analysis has been undertaken, albeit using a limited number of replicated farms (> 90) per farm type to generate frequency distributions of mitigation impacts. On this basis, the uncertainty ranges presented are unlikely to capture the full uncertainty of mitigation impact and further work is required to improve the representation of uncertainty associated with this modelling approach, e.g. by including the uncertainty associated with the quantification of costs and benefits and the incorporation of a greater number of replicates per farm type to avoid compensation effects.

Whilst the modelling scenarios suggest that SC measures will deliver more impact on pollutant emissions than either MC or DC options, the former are likely to be more difficult to get farmers to implement given

Table 4
Summary statistics for the estimated annual costs (£/ha/year) to farmers at WMC- scale. Note that statistics were calculated across each cost independently.

	Fixed cost	Variable cost	Total cost
Source control (SC)			
Minimum	5.4	−199.1	−4.5
Maximum	258.3	150.0	162.9
Median	45.1	17.9	65.7
P5*	11.3	-113.2	4.3
P25*	25.0	-32.6	42.3
P75*	73.3	65.1	104.7
P95*	158.5	114.6	134.4
Variability**	326.5	1275.7	197.9
Mobilisation control (MC)			
Minimum	-0.3	−27.4	−2.5
Maximum	24.9	53.8	55.9
Median	0.6	31.9	33.6
P5	-0.2	9.3	9.2
P25	0.1	22.1	23.2
P75	1.7	41.4	42.4
P95	4.0	48.4	50.6
Variability	743.8	122.8	123.2
Delivery control (DC)			
Minimum	5.3	1.9	7.8
Maximum	35.9	42.5	50.8
Median	14.6	19.0	37.8
P5	6.8	4.8	14.0
P25	9.3	13.4	30.4
P75	19.7	27.5	41.7
P95	27.3	36.0	47.0
Variability	140.3	164.3	87.2
All measures (SMDC)			
Minimum	15.7	-155.9	13.4
Maximum	291.4	211.1	239.9
Median	65.3	61.7	129.2
P5	22.3	−65.4	41.1
P25	40.0	−4.5	104.4
P75	91.1	107.8	170.8
P95	184.5	173.0	208.5
Variability	248.16	386.29	129.47

* P5, P25, P75 and P95 are calculated percentiles at 5%, 25%, 75% and 95%, respectively.
** Variability is calculated as the difference between P95 and P5 divided by the corresponding median value.

Table 5
Summary statistics for the predicted benefits of the mitigation scenarios.

Environmental benefits (£/ha)	P5*	Median	P95*
Source control scenario (SC)	22.3	33.8	64.4
Mobilisation control scenario (MC)	2.9	6.8	20.8
Delivery control scenario (DC)	3.5	5.6	16.8
All measures control scenario (SMDC)	30.9	43.9	90.0
Benefit-to-cost ratio	**P5**	**Median**	**P95**
Source control scenario (SC)	0.19	0.44	2.48
Mobilisation control scenario (MC)	0.09	0.19	0.95
Delivery control scenario (DC)	0.09	0.15	0.65
All measures control scenario (SMDC)	0.20	0.31	1.39
Water quality benefit proportion	**P5**	**Median**	**P95**
Source control scenario (SC)	0.02	0.11	0.35
Mobilisation control scenario (MC)	0.08	0.35	0.80
Delivery control scenario (DC)	0.05	0.37	0.80
All measures control scenario (SMDC)	0.02	0.14	0.45

* P5 and P95 are calculated percentiles at 5% and 95%, respectively.

that they frequently require fundamental change to farming systems and involve higher capital outlays. Here, taking better account of farmer attitudes towards managing the diffuse pollution problem will be important (Blackstock et al., 2010; Buckley, 2012; Collins et al., 2016). The MC and DC scenarios represent more of a 'sticking plaster' approach to dealing with agricultural pollutant emissions, as opposed to dealing with the problem at source. In the context of statistically significant changes to rainfall intensities in some locations in the UK, including higher mean totals per rain day and more back to back rain days with totals in excess of 30 mm (Burt et al., 2016), the efficacy of interventions targeting pollutant mobilisation and delivery will be at risk of being undermined by changing rainfall regimes, again, underscoring the need for preventative measures at pollutant source. Here the 'treatment-train' approach is also likely to be increasingly relevant by providing multiple lines of defence with respect to the transfer of agricultural pollutants to water.

3.7. Policy implications

Of the four scenarios simulated, the SC and SMDC suites of measures have the greatest potential to deliver reductions in BAU emissions from agriculture, and the best benefit:cost ratios in the context of combined agricultural pollutant emissions to water and air. Given the decision of the UK to leave the EU, there is much ongoing debate concerning what kind of agricultural support policies will best serve private and public good post-Brexit. Various options are being discussed. A new policy framework centred on risk management is one option, with a view to helping farmers manage risk and associated business impacts via affordable insurance schemes, contingency fund planning and a mixture of advice, training and decision-support tools. The results herein suggest that a risk management framework encouraging increased uptake of SMDC 'treatment-trains' could deliver acceptable benefit:cost outcomes. An alternative option focuses on ecosystem services and public good but with more flexibility than current incentivised schemes to deliver the mix of services valued by stakeholders at both larger and more local scales. The continuation of private contributions in the form of payment for ecosystem services is likely in association with this particular option. Here, the modelling results suggest that SC and SMDC scenarios will deliver the best value for public good at landscape (WMC) scale. A third option acknowledges the need for direct area-based payments to cease as a means of supporting more timely structural evolution in the farming industry. Here, the direct payments might be phased out gradually over time or ceased abruptly, but with a compensation payment or bond to deliver a capital injection whilst new market conditions emerge. Such an approach might provide the capital injection necessary to support improved delivery of SC options for diffuse pollution.

Median values for the proportions of public benefit attributable to reductions in agricultural emissions to water are less than 0.5 for all modelled scenarios. This suggests that in the context of any debate focussing solely on the need to mitigate agricultural pollutant emissions to water, it is essential to bear in mind that greater environmental gains will actually be associated with co-benefits arising from predicted reductions in gaseous emissions, thereby supporting the design of intervention strategies for delivering multiple benefits. The highest proportions of water quality benefit are predicted for the MC and DC scenarios and these should therefore feature in any new policy designed specifically to tackle agricultural water pollution at the scales discussed herein, albeit in recognition that designing policies for maximising co-benefits is likely to be priority. Delivery of the MC scenario in particular, will require improved advice, training and decision support tools given that the spatially diffuse nature of pollutant mobilisation processes will hamper easy identification of optimal targeting plans for the relevant on-farm interventions. The risk management policy option is therefore most likely best suited to ensuring successful delivery of the predicted benefit:cost associated with the MC scenario for water quality

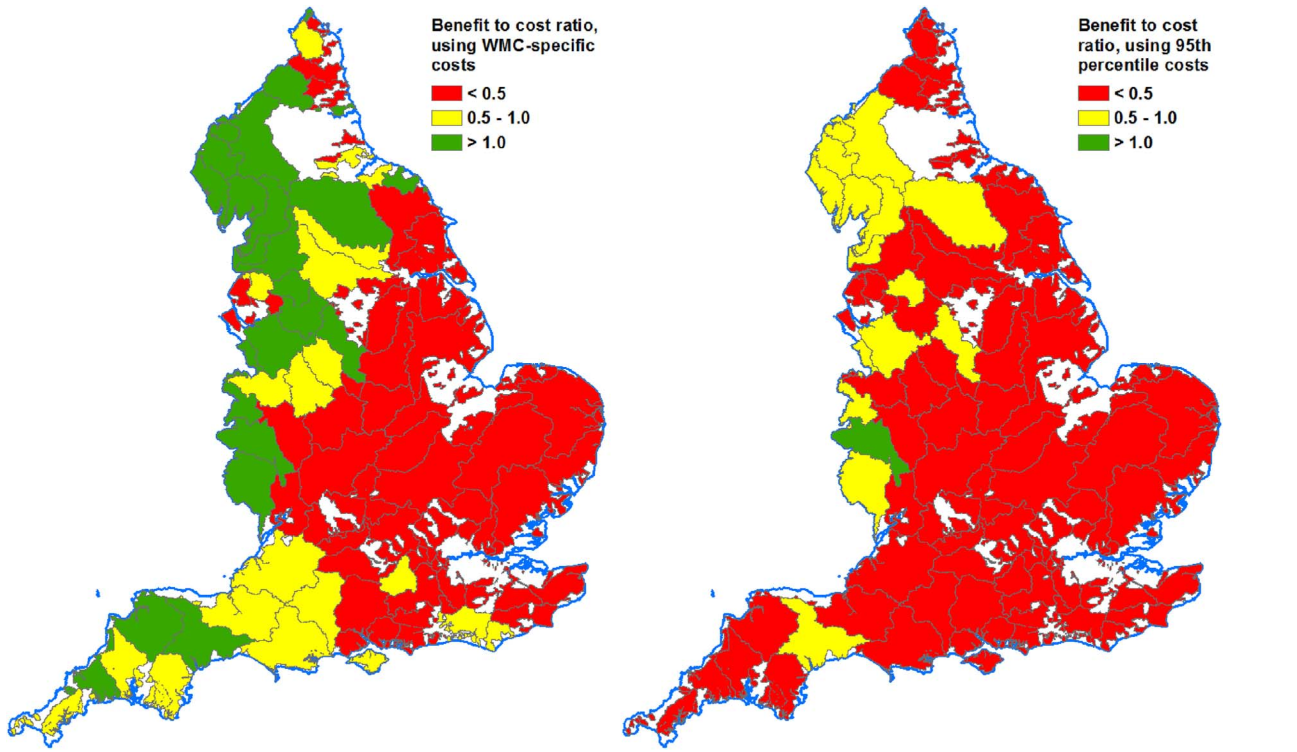

Fig. 10. The estimated ratio of environment benefit to on-farm implementation costs for the SC scenario within CS priority areas at WMC-scale across England.

protection.

4. Conclusions

In the context of the above limitations and uncertainties of the modelling work reported here, the predictions of the strategic policy-relevant modelling framework suggest that, of the four theoretical scenarios considered, the SC and SMDC suites of measures have the greatest potential to deliver reductions in BAU emissions from agriculture. One element common to both scenarios is the inclusion of source control (SC) measures. Furthermore, these scenarios include a greater number of individual on-farm mitigation measures to tackle the agricultural diffuse pollution cascade. Whilst an implicit assumption of the modelling work is that the combined efficacy of a given suite of measures is the multiplicative product of the impact of individual on-farm options, larger suites of measures will typically deliver greater overall reductions in pollutant loadings. The benefit-to-cost ratios of the four scenarios also indicate that the SC and SMDC scenarios would perform better. However, the high fixed and total costs associated with these two scenarios, imply that implementation costs could pose a potential barrier to uptake, especially in the current context of highly volatile commodity prices being paid to farmers. The dominance of environment benefits attributable to gaseous emission reductions suggests that a holistic review of the ecosystem services provided by different mitigation scenarios, including water quality, climate change and biodiversity, is required for an informed evaluation of environmental benefits associated with the deployment of on-farm mitigation measures (Anthony et al., 2012).

Acknowledgements

The authors gratefully acknowledge the funding provided by the Department for Environment, Food and Rural Affairs (Defra) via project LM0304. The provision of shape files and related attribute data for the Countryside Stewardship priority areas by Chris Burgess at the Environment Agency is also gratefully acknowledged.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.envsci.2017.04.004.

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