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Optimisation of Cereal Farm Strategies for Mitigating Externalities Associated with Intensive Production

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Abstract: Intensive cereal farming results in various unintended consequences for the environment including water pollution. Current uptake of on-farm best management measures in the UK is delivering limited benefits and alternative management futures need to be modelled to make informed decisions. The Farmscoper (FARMSCale Optimization of Pollutant Emission Reductions) tool was used to examine two management scenarios for intensive cereal farms in eastern England. The first was based on increased uptake of those measures currently recommended by advisory visits and following walkover surveys. The second was founded on mechanistic understanding of on-farm pollutant sources embedded in the Farmscoper tool. Optimization of measure selection used a multi-objective genetic algorithm. The technically possible reductions (e.g., 10 to 21% for sediment and 12 to 18% for total phosphorus) of current pollutant emissions to water due to uptake of the mechanistic scenario exceeded those resulting from the current advice scenario ($\leq$5%), but with mixed impacts on costs ranging from a saving of £34.8/ha/yr to an increase of £19.0/ha/yr, relative to current best management costs. The current advice scenario generated corresponding cost savings of between £30.4/ha/yr and £73.40/ha/yr. Neither scenario is sufficiently impactful on unintended consequences, pointing to the need for structural change in land cover.

Keywords: cereal farming; pollution; mitigation; trade-offs

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1. Introduction

Modern cereal production is characterised by high resource inputs, energy consumption and capital investments. While it has provided food security and boosts rural economies, it has also contributed significantly to various environmental issues challenging the world today, including the degradation of water quality via eutrophication, elevated greenhouse gas (GHG) emissions, damage to soil physical structure and biodiversity loss [1]. As a result, intensive cereal farming has the potential to disrupt ecosystem services at large scale and result in long-term risks to food security and human health [2]. A paradigm shift is therefore required to ensure that the world's food production is sustainable by mitigating social and environmental externalities [3]. This shift requires restructured cropping systems [4–6] and effective management practices to mitigate the associated negative environmental impacts.

The externalities associated with intensive cereal farming involve multiple pollutants from different sources including elevated losses of nutrients, sediment, pesticides and GHGs. A range of on-farm mitigation measures for improving both water and air quality have been assessed with respect to their efficacies, applicability and costs [7–10]. These on-farm measures can be used to target different pollutants and the key pathways for their emissions into freshwater and the atmosphere. On the basis of existing empirical evidence and, the elicitation of expert opinion, knowledge based decision support tools have been developed to assist the selection of on-farm mitigation measures based on the intersection of farm-specific cropping or livestock types and management and their corresponding physical environment [11,12]. These tools can be used to explore different scenarios for alternative management futures, including combinations of measures shortlisted to reflect farmer preferences [13] to target different stages in the nutrient delivery continuum [14] or on the basis of improved mechanistic understanding of the key controls on pollutant emissions [15].

For evaluation of agricultural emissions to water, the hydrological catchment is the logical management unit. In Europe, surface waterbodies have been delineated for the implementation of the EU Water Framework Directive (WFD). These have become the base spatial unit for the monitoring and assessment of aquatic chemical, biological and ecological status over time in conjunction with the implementation of programmes of measures (POMs). The evidence used for the shortlisting and recommendation of on-farm mitigation measures usually comprises routine water quality monitoring, agri-environment scheme related farm visits and catchment reconnaissance surveys to identify catchment-specific water pollutant pressures and corresponding problematic land management activities. This current approach has its limitations, however, since it frequently relies on single and commonly snapshot visual observations and appraisals of on-farm problems potentially driving water pollution. Clearly, such activities can easily miss the significant role of key controls on water pollution from agriculture, including, for example, the subsurface delivery pathway via assisted drainage and features controlling the delivery of mobilised pollutants between on-farm sources such as fields or yards, and freshwater receptors. An added complexity is that the outcomes of conventional visual audits of on-farm problems for water quality can also be time sensitive.

Given the above background, and the limited benefits arising from the current uptake of best management practices in agriculture, we compared two alternative management futures for cereal farms using the Farmscoper decision support tool [11,12]. The work focused on waterbodies in eastern England, where cereal farming is the dominant farm type. Water quality is one key local catchment management issue because it impacts on aquatic ecological status and the quality of potable water supplies. Water quantity is also important as the study area is also a water stressed area [16]. More generally, however, post-Brexit agricultural policy in England is implementing a 'public goods for public money' incentivisation framework wherein whole farm system planning informed by understanding of the co-benefits and trade-offs of management strategies will be increasingly important. Given this shift in policy, decision support tools like Farmscoper can play a critical role in generating estimates of farm system impacts for multiple outcomes relevant to the provision of public goods and services in the UK and beyond.

2. Study Area and Selection of Waterbodies for Investigation

The study area for the work herein is part of the Broadland Rivers operational catchment which lies inside the Anglian river basin district (Figure 1). With a total catchment area of ~3178 km^2, its rivers flow into the unique wetland habitats of the Norfolk Broads national park which includes over 90 Sites of Special Scientific Interest (SSSIs) that are home to rare and endangered species. Long-term annual rainfall varies between 606 to 765 mm and mean annual temperatures range between 9.5 to 10.6 degrees Celsius. The study area is classified as having a temperate oceanic climate, or Cfb in the Köppen climate classification system. Soils are represented by slowly permeable, seasonally waterlogged, fine loamy over clayey soils (Beccles series) in the south west parts of the study area and deep well-drained coarse loamy soils (Wick series) in the north east.

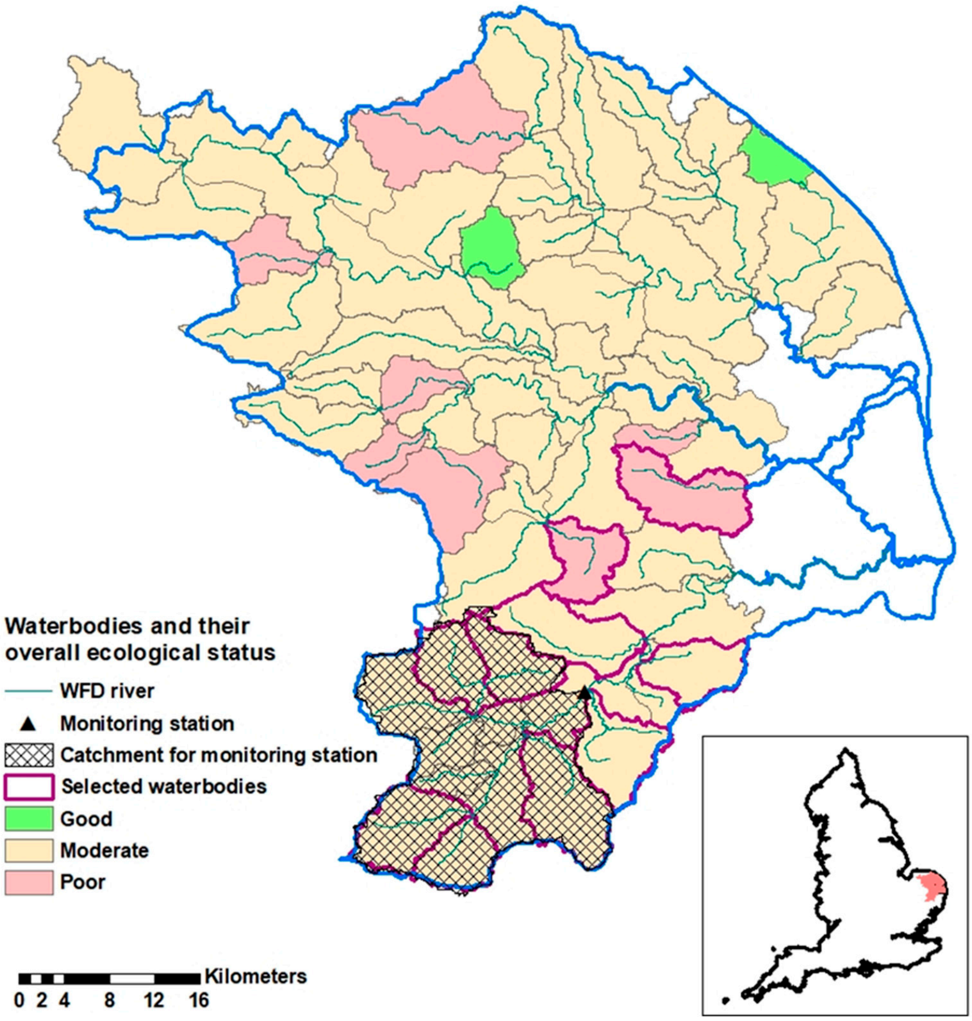


Figure 1. Waterbody WFD ecological status in the study area. Inset shows the location of the study area in eastern England.

Based on remote sensing data for 2019, 61% of the study area is represented by arable and horticulture and 20% by improved grassland. The remaining important land uses include broadleaved woodland (7%) and urban areas (8%). Small areas of grazing meadows and semi-natural fens are found in river valleys and around the broads. Cereal farms dominate, growing winter cereals, peas and beans, potatoes and sugar beans. Apart from food production, agriculture and associated land management also makes a significant contribution to landscape character and tourism within the study area. As one of Europe's most popular inland waterways, boating, walking, angling and birdwatching are popular local activities. As a result, large stretches of the local river channels have been heavily modified for historical flood defence, land drainage and grain milling activities. Large scale and intensive arable farming operations have also been associated with the high nitrate and pesticide concentrations in drinking water source areas and groundwater, storm-related surface runoff and soil erosion, elevated nutrient levels in protected habitats and failure to achieve 'good' ecological status in WFD designated surface waterbodies [17]. Part of the catchment is also highly susceptible to the degradation of soil physical quality from soil compaction (European Commission–Joint Research Centre, 2008). Considering the urgency surrounding climate change, the agricultural sector must endeavour to deliver its expected contribution at the catchment scale in support of achieving the national target of net zero GHG emissions by 2050.

For the purposes of the implementation of the EU WFD, the study area was previously divided into 59 surface waterbodies and 3 coastal waterbodies (latter not shown in Figure 1, nor used in this work). As illustrated in Figure 1, the majority of the former are reported as being in 'moderate' ecological status. To avoid any potential complications from non-agricultural sectors or activities, our work focused on those waterbodies that are not heavily modified. Additionally, the remaining waterbodies were filtered further based on waterbody specific data from the 'catchment data explorer' data portal (https://environment.data.gov.uk/catchment-planning/), which is managed by the UK Environment Agency, England (hereafter referred to as the 'EA data portal'). Here, the following criteria were used to finalise the selection of the target waterbodies: a waterbody must be larger than 25 km^2 to ensure information on agricultural activities and the natural environment data are reliable; its current ecological status must be either 'moderate' or 'poor', and agricultural and rural land management has been cited as one of the reasons for not achieving 'good' ecological status. Using these criteria, 10 waterbodies were selected and their spatial distribution is shown in Figure 1. It is worth noting here that most of the shortlisted waterbodies are in the southwest portion of the study area in the Waveney catchment, where heavy soil with a high risk of soil compaction [18] is widespread.

3. Data Collection and Modelling

A schematic presentation of the workflow is shown in Figure 2.

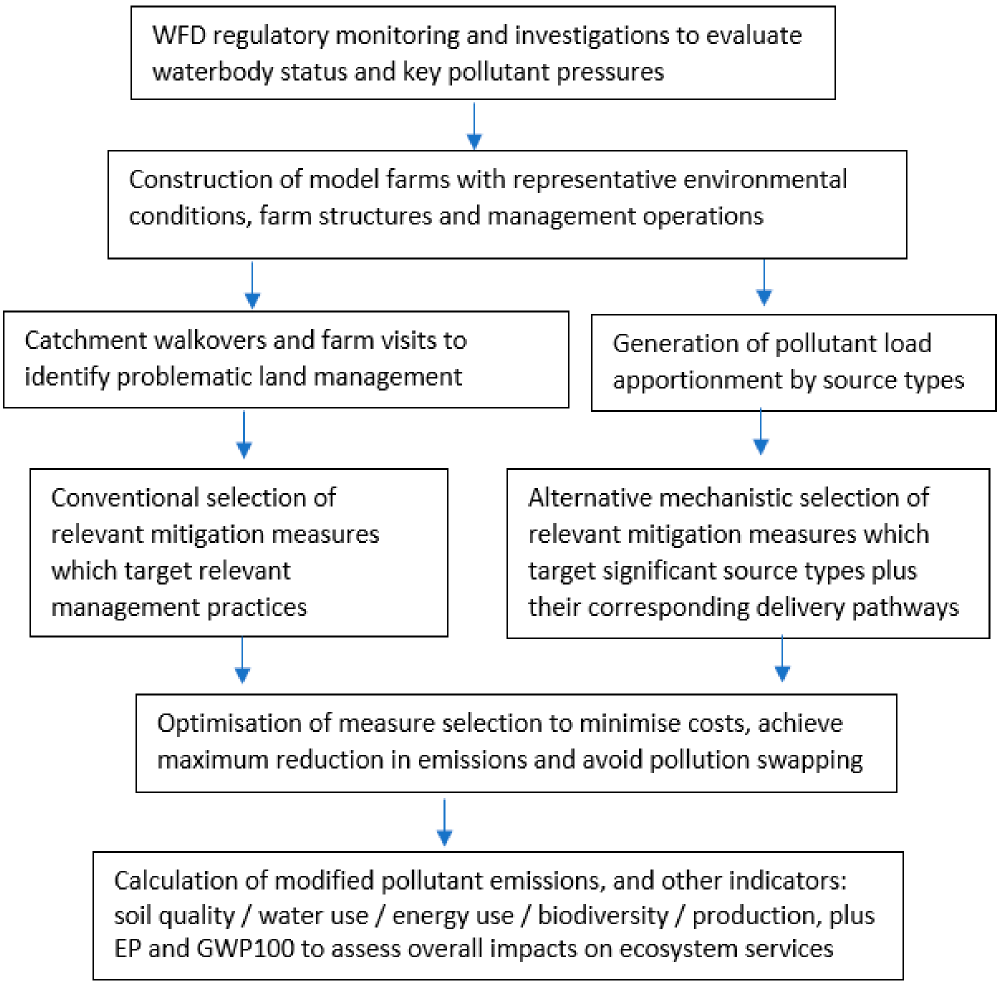


Figure 2. Schematic of the workflow.

3.1. Development of Model Farms and On-Farm Mitigation Scenarios

Representative model farms were generated using waterbody-specific data, where available, to characterise the natural environment, farm structure and land management practices. Long-term average (1981–2010) annual rainfall was estimated based on HadUK-Grid data which provides a collection of gridded climate variables derived from the network of UK land surface observations at 1 km^2 resolution. Soil drainage status was inferred from NatMap 1000 (National Soil Resources Institute, Cranfield, UK) which lists relevant soil series for each 1 km^2 cell nationally. Pedo-transfer functions [14] were used to determine Hydrology of Soil Type (HOST; [19]) and soil drainage status for the relevant soil series. To characterise farming activities in the different waterbodies, aggregated June Agriculture Survey (JAS) data for 2019 for each waterbody was sourced from the UK Department for Environment, Food and Rural Affairs (Defra) to quantify farm types, crop areas and livestock types, counts and ages. Only data for the 'Cereals', 'General cropping' and 'Mixed' Robust Farm Types (RFT; Defra 2020) were used for the modelling. Multiple-year (2015–2019) average fertiliser application rates for the same RFTs were calculated based on published data from the British Survey of Fertiliser Practice (BSFP; https://www.gov.uk/government/collections/fertiliser-usage (accessed on 20 October 2020). Based on manure dressing data available in the BSFP, there was no significant change in manure spreading practices since 2010 for spring or autumn crops and grassland. Accordingly, the manure allocation schemes available in the 'Upscale' workbook of Farmscoper were adopted without modification. Business-as-usual (BAU) impacts of farming in the target waterbodies were based on both farm structure (i.e., crop areas, livestock types, counts and ages) and current farm management practices resulting from the combination of regulation, incentivization and advice.

Two strategies were adopted to frame mitigation scenarios for each target waterbody. Given the known pollutant emission pressures from existing catchment monitoring and investigations, one strategy used the available evidence from the EA data portal to shortlist relevant on-farm mitigation methods. Here, the available records of Reasons For Failure, responsible sectors (e.g., agriculture–arable, agriculture–livestock, wastewater treatment) and problematic activities (e.g., poor soil management, poor nutrient management,) were identified for each waterbody. The listed activities associated with arable farming were then used to shortlist relevant on-farm mitigation measures as given in the Farmscoper mitigation library and relevant management bundles are detailed in Table S1 [20]. If multiple activities were identified as driving reasons for failure for any given waterbody, the shortlisted on-farm mitigation measures from the relevant different bundles were combined. This approach was treated as the 'current advice' scenario.

The second approach was based on the quantification of the relative contributions of key on-farm source types to the identified priority water pollutants in each waterbody. Here, on-farm mitigation measures which target the most significant contributing source types (i.e., >10%) and which target the delivery stage of the nutrient transfer continuum [21] were selected. Source type-based apportionment of priority pollutants was obtained from automated runs of the Farmscoper 'Evaluation' workbook. The selection of source type-specific measures was based on the measure descriptions available in the Defra User Guide [7,9]. Lists of the on-farm measures for the relevant source types identified on the model farms in the study area are provided in Table S2 This scenario is referred to as the 'mechanistic scenario' since it was founded on existing mechanistic understanding of the typical on-farm sources and delivery pathways for the pollutants emitted from different farm types, captured in the Farmscoper tool.

3.2. Assessment of Business-as-Usual Pollutant Load Predictions

We evaluated the BAU estimates of pollutant loads from Farmscoper against available data, including regulatory water quality monitoring and annual GHG reporting. For the former, a routine monitoring station was identified in the study area (Figure 1). Continuous daily mean flow data were downloaded from the National River Flow Archive (https://nrfa.

ceh.ac.uk/data/station/meanflow/34006, accessed on 7 May 2021) and then combined with water quality data for the same station from the Water Information Management System (http://environment.data.gov.uk/water-quality/id/sampling-point/AN-WAV050, accessed on 8 May 2021) to estimate annual nitrate, total phosphorus and sediment loads between 2004 and 2014. For GHG emissions, gridded emissions data at 1 km^2 resolution for the study catchments were extracted from national datasets downloaded from the National Atmospheric Emissions Inventory (https://naei.beis.gov.uk/data/map-uk-das, accessed on 3 June 2021). Summary statistics (e.g., median, 5th percentile, 95th percentile) were calculated for comparison with the modelled BAU estimates for the model farms in the study area. For sediment loss to water, empirical data for evaluating the model predictions were represented by suspended sediment yields for the study waterbodies based on the estimates for different catchment types in the UK reported by [22]. These catchment-type estimates are based on the extrapolation of high-quality measurements of suspended sediment yields in research catchments.

3.3. Optimisation of the Selection of On-Farm Mitigation Measures

Because any farm rarely has a single problematic management activity or pollutant source, the number of potentially relevant mitigation measures can easily snowball along with the associated implementation costs, as well as risks of pollutant swapping or other unintended impacts on different ecological services. For both alternative future management scenarios, the shortlisted relevant on-farm measures were optimised using the functions available in the 'Evaluation' workbook of Farmscoper. Here, a multiple objective optimisation procedure can be executed using the NSGA-II algorithm [23] to identify a family of solutions lying on the pareto-optimal front of non-dominated solutions. The algorithm is based on an elitists selection method wherein elites of a population are given the opportunity to be carried to the next generation, an explicit diversity preserving mechanism (Crowding distance) and an emphasis on the non-dominated solutions. It is currently used as an industry standard in comparisons of genetic algorithms. More details about the algorithm and its implementation in Farmscoper can be found in the relevant literature [12,23]. Similar algorithms have already been employed to inform the management of diffuse pollution problems at catchment scale [6,24]. A solution or, in this case, a combination of on-farm mitigation methods dominates another solution if it is superior or equal in all objectives (i.e., water pollutant reductions) but at least superior in one objective. Each solution is a unique combination of on-farm mitigation measures from the shortlist for the scenario in question and seeks to minimise both annual cost and maximise the corresponding reduction of the chosen pollutants simultaneously. Emissions to both water (nitrate, phosphorus, sediment, pesticides, FIOs) and air (methane, nitrous oxide, ammonia) were specified. Based on the known waterbody-specific pollutant pressures, a minimum reduction of 5% was specified for the target pollutant(s), whilst at the same time permitting no elevation in the emissions of other pollutants, to ensure minimization of trade-offs. The settings for the optimisation algorithm were: population size (50) and number of generations (100). With the optimisation runs, a collection of valid solutions meeting all the constraints was generated along with expected annual costs to farms, as well as modified pollutant emissions and qualitative scores for additional outcomes represented by soil physical quality, biodiversity, water use, energy use and farm production.

3.4. Estimation of Eutrophication and Global Warming Potential

Farmscoper predicts a suite of quantitative data for pollutant emissions to water and air. In turn, these were used to undertake mid-point impact quantification for two widely used categories in life cycle assessment (LCA): eutrophication potential (EP) and global warming potential (GWP100). Both were calculated by multiplying pollutant specific characterisation factors and their corresponding annual emissions to aggregate the effects from different pollutants into common units for environmental impacts [25]. More specifically for this study, EP and GWP100 are expressed in kg PO_4 eq. and kg CO_2 eq., respectively.

For the former impact category, emissions of nitrate, phosphorus, nitrous oxide and ammonia are aggregated. In so doing, the characterisation factors were 0.44, 3.06, 0.13 and 0.44 kg PO_4 eq. kg^{-1}, respectively. For the latter, methane, ammonia and nitrous oxide emissions were aggregated, using corresponding respective characterisation factors of 26, 1.6 and 296 kg CO_2 eq. kg^{-1}.

4. Results

4.1. Characteristics of Model Cereal Farms and Their BAU Emissions to the Environment

Mapped spatial patterns of rainfall and soil distributions suggest that most areas (97%) in the selected waterbodies have long term annual rainfall of 600–700 mm, whilst ~78% of the agricultural land has very heavy soils requiring artificial drainage for both arable and grassland use and 15% for arable use, i.e., intensive cereal production. 2019 JAS data indicates that cereal farms are the dominant RFT (covering 39.5 % to 84.5% of agricultural land with a median of 58%) in all of the waterbodies. Mixed farms and general cropping farms are the second most important but rarely exceed 30% of the agricultural land in any individual target waterbody.

In total, 17 model cereal farms were generated using Farmscoper. For most waterbodies, two model farms were constructed to reflect the presence of two distinctive types of soil drainage conditions. All the model farms comprised a large proportion of winter cereals. In descending proportions, these comprised winter wheat (34–57%), winter oilseed rape (11–20%) and winter barley (3–15%). More variability was observed for other crops, such as maize and sugar beet. These are insignificant in area by comparison (<7%) but could generate high externalities since both, for instance, are highly susceptible to soil erosion [26,27]. Table 1 compares the relative proportions of key crops within each waterbody in 2019 and their corresponding proportions for the model cereal farms. In total, arable land use accounts for between 74% to 94% of the farmed area on the model cereal farms. The remaining area is used for cattle, sheep and lambs and poultry.

Table 1. Relative proportions (%) of major crops at waterbody scale in 2019* and corresponding information for the model farms.

	Winter Wheat		Oilseed Rape		Winter Barley		Maize		Sugar Beet	
Water-	**2019**	**Model**	**2019**	**Model**	**2019**	**Model**	**2019**	**Model**	**2019**	**Model**
**Body ** **	**Data**	**Farm**	**Data**	**Farm**	**Data**	**Farm**	**Data**	**Farm**	**Data**	**Farm**
45,650	46.1	54.1	13.6	19.5	10.2	6.2	1.6	0.1	12.8	5.9
45,660	43.5	49.7	14.2	10.6	9.8	7.1	0.5	0.0	11.6	6.2
45,690	43.0	48.5	18.2	14.9	7.2	8.9	1.9	0.4	10.2	5.7
45,720	38.0	34.4	13.5	12.2	12.8	14.5	1.9	1.5		2.7
45,741	50.0	52.9	19.5	13.6	9.3	14.0	4.5	1.6		2.0
45,830	50.1	56.6	17.4	20.3	6.8	2.7	0.9	0.0		3.4
45,840	38.8	48.7	11.2	20.6	13.8	8.7	4.4	1.2	8.7	4.2
45,850	39.3	45.8	16.3	11.2	12.3	14.6	3.4	0.7		0.3
45,880	40.3	49.3	19.0	15.8	11.6	6.9	2.9	2.1		0.5
51,190	35.9	39.7	13.9	15.1	7.7	8.6	6.3	0.8		1.0

Note(s): * Land Cover Map 2019: Crop, Scale 1:250,000, Tiles: GB, Updated: 30 June 2020, CEH, Using: EDINA Environment Digimap Service, https://digimap.edina.ac.uk (accessed on 1 February 2021); ** Shared common prefix (i.e., 'GB1050340') was omitted.

Accounting for the potential impacts of existing on-farm mitigation measures resulting from regulation, incentivisation and advice, the quantified annual emissions of key pollutants to water and air for the model cereal farms in the selected waterbodies are summarised in Table 2. Additional footprint information is summarized in Table S3. Among the pollutants listed in Table 2, sediment and total phosphorus show higher inter-model farm variability with soil drainage status as the main driver. The model farms with soil drained for arable only are characterised by much lower sediment and total P loadings to water, in comparison with those with soils drained for both arable and grassland use. For the former farms, the estimates range between 0.29 to 0.35 kg/ha and 173.4 and 204.6 kg/ha for total P and sediment, respectively. For the latter farms, the corresponding respective estimates are 0.50 to 0.59 kg/ha and 313.6 to 367.3 kg/ha. No significant differences were estimated for other pollutants between the two categories of soil drainage status. The annual ranges in these additional emissions for all model farms were estimated at 18.6 to 24.2 kg/ha for nitrate, 5.1–6.8 kg/ha for ammonia, 0.6–10.5 kg/ha for methane, 2.6 to 3.2 kg/ha for nitrous oxide and 0.07 to 0.12 dose units/ha for pesticides. Comparison of the specific loads estimated for the model farms against other available estimates are provided in Table 3 wherein 5th percentile, 95th percentile and median values for the two sample populations are included. In interpreting these comparisons, it is important to bear in mind that the modelled estimates cover cereal farms only. For nitrate and nitrous oxide, the modelled emissions are relatively close to the monitored values. For ammonia and methane, significant differences are observed because livestock farming is the major source and contributor of these pollutants on arable farms. In the case of the water pollutants, the use of daily mean flow and regular monthly sampling for pollutant concentrations in the assembly of the monitored data could easily miss the infrequent but significant storm events for water pollution transfer. This, for instance, is likely to be a factor explaining why there is a higher discrepancy between the modelled and monitored loads for sediment and total P, relative to the corresponding difference in loads estimated for nitrate.

Table 2. Annual water and air pollutant emissions at farm scale under BAU.

Water-	Drainage	Nitrate	Total Phosphorus	Sediment	Ammonia	Methane	Nitrous oxide	Pesticides
Body *	**Status ****	**kg NO_3-N ha^{-1}**	**kg P ha^{-1}**	**kg ha^{-1}**	**kg NH_3-N ha^{-1}**	**kg CH_4 ha^{-1}**	**kg N_2O ha^{-1}**	**Dose Units ha^{-1}**
45650	A	22.0	0.31	174.0	6.4	1.5	3.1	0.09
45660	A	21.8	0.35	204.6	5.9	1.8	3.1	0.10
45690	A	22.5	0.33	184.8	6.2	1.5	3.2	0.10
45720	A	18.6	0.32	184.0	5.2	3.6	2.7	0.08
45741	A	22.1	0.32	179.2	6.8	10.5	3.1	0.10
45880	A	20.9	0.31	173.4	6	0.6	2.9	0.09
51190	A	19.6	0.29	174.7	5.1	4.8	2.7	0.07
45650	AG	23.7	0.53	314.8	6.4	1.3	3.2	0.10
45660	AG	23.2	0.59	367.3	5.9	1.8	3.1	0.11
45690	AG	24.2	0.55	331.9	6.2	1.2	3.2	0.11
45720	AG	19.7	0.55	335.2	5.2	3.6	2.7	0.09
45741	AG	23.8	0.54	321.9	6.8	10.5	3.2	0.11
45830	AG	24.2	0.54	323.4	6.8	6.9	3.1	0.12
45840	AG	23.5	0.52	313.9	6.2	1.2	3.1	0.09
45850	AG	21.5	0.59	360.5	5.9	1.2	2.6	0.10
45880	AG	22.4	0.52	313.6	6	0.6	3	0.10
51190	AG	20.8	0.50	319.6	5.1	4.8	2.7	0.08

Note(s): * Shared common prefix (i.e., 'GB1050340') was omitted; ** 'A' stands for for drained for arable and 'AG' stand for 'drained for arable and grassland'.

Table 3. Comparison of modelled BAU area-specific pollutant loads (kg/ha/yr) against alternative estimates.

		Nitrate *	**Phosphorus ***	**Sediment ****	**Nitrous Oxide *****	**Ammonia *****	**Methane *****
Modelled	P5	17.2	0.3	173.9	2.7	9.5	0
	P95	21.6	0.6	361.8	3.2	12.6	3.5
	median	19.6	0.5	313.9	3.1	11.2	0.6
Monitored	P5	4.1	0.1	5.3	3.5	16.7	15.8
	P95	17.7	0.3	59.5	4.8	38.1	32.1
	median	11.2	0.1	17.5	4.3	25.3	22.4

Note(s): * Monitored data (2004 to 2014) extracted from the national Water Quality Data Archive (https://environment.data.gov.uk/water-quality/view/landing, accessed on 9 January 2020); ** monitored data based on Natural England report (Cooper et al., 2008); *** 2018 UK gridded GHG inventory data (https://naei.beis.gov.uk/data/mapping-archive, accessed on 3 June 2021).

As highlighted in previous work (cf. [14]), the major externalities associated with cereal farming are the above average emissions of nitrate, sediment and total P in comparison with other major farm types at national scale. As part of our modelling exercise herein, the potential pollutant load reductions associated with the existing uptake of mitigation measures under BAU were also quantified (supplementary Table S4). These estimates suggest that existing best management has greater impacts on nitrate (reductions of baseline emissions by 6.1 to 10.2%) and pesticide (reductions of baseline emissions by 11.9 to 25.4%) emissions, than on sediment and phosphorus (<5%).

4.2. Optimisation Results Using the Current Advice and Mechanistic Scenarios

Corresponding to monitored pollutant pressures in each of the target waterbodies, two broad categories of problematic land management practices are listed in the Environment Agency data portal (https://environment.data.gov.uk/catchment-planning/, accessed on 4 March 2021): soil management and nutrient management. The former is associated with fine sediment pressure and the latter is associated with phosphate pressure. Multiple malpractices are likely to generate these known pollutant pressures. More specifically, the Reasons For Failure assessment for the study waterbodies from the Agency data portal suggests that one waterbody is failing due to fine sediment alone (GB105034045741), three waterbodies due to both fine sediment and phosphate (GB105034045720, GB105034045830 and GB105034045840) and the remaining six (GB105034051190, GB105034045650, GB105034045660, GB105034045690, GB105034045850, GB105034045880) due to phosphate alone. To improve soil and nutrient management on farm, Farmscoper includes in its mitigation library, nine and 16 measures, respectively. These measures are listed in supplementary Table S1.

Using on-farm measures typically recommended for cereal farms by current advice, optimisation runs for the model cereal farms yielded a wide range in the number of viable solutions. For six model farms, only four viable solutions were found, whereas for one cereal farm the optimisation generated 100 solutions. This range is most likely due to individual model farms having more homogeneous land use, single pollutant pressures, or relatively high uptake of mitigation measures under BAU. Median estimated values for pollutant emission reductions and impacts on additional indicators for the model farms are presented in Table 4. The results suggest the measures selected using this approach can offer some savings in annual total costs (£30.4/ha/yr to £73.40/ha/yr), but the effects on the targeted pollutants, i.e., sediment and total P, would be limited. For most model farms, the technically feasible reductions in BAU sediment loads would be only ~4% compared with only ~5% for total P. The modelled estimates in Table 4 suggest that differences between the impacts on model farms with different soil drainage status would be limited. Similar magnitudes of reductions were also predicted for BAU nitrate loadings. More significant impacts were predicted with regards reductions in ammonia and energy use (>15%) on most model farms. Higher variability was observed for improvements in the terrestrial biodiversity scores, with better improvements (>5) in those waterbodies which have sediment listed as one priority Reason For Failure, but very limited ($\leq$1) positive changes were predicted for the remaining waterbodies.

Farmscoper-based evaluation of the relative contributions of pollutants from various on-farm source types suggests that soils and fertilisers are the most important contributors to sediment and phosphorus loss on all the model farms. For all model farms, >90% of particulate P losses are from this source area. There are, however, marked differences in terms of the relative magnitudes between the farms with different soil drainage status for dissolved P. While the fertiliser source is more important on all model farms, its contributions on farms with heavy soils are much more pronounced (>70%) as shown in Figure 3. Other on-farm sources, including FYM, slurry and voids are estimated to contribute <1% of the annual dissolved P load at farm scale for nearly all model farms considered. This contrasts with more contributing sources and the wide ranges of variations shown for the mixed farms in the study area. The latter RFT was included here for comparison only and no optimisation work were undertaken for this farm system. Mitigation measures for these two main on-farm source types (i.e., soils and fertilisers), plus measures targeting pollutant delivery pathways associated with these on-farm sources, resulted in a pool of 46 potential measures for inclusion in the optimisation runs for the alternative management scenario based on mechanistic (i.e., source contributions) understanding. Inclusion of an increased number of measures resulted in more valid solutions for the model farms, with the number of viable solutions varying between 93 to 150. Median values of reductions in the annual emissions of pollutants to water and air along with impacts on additional key indicators are presented in Table 5. All model farms can technically achieve greater annual pollutant load reductions using this alternative management scenario. The ranges of predicted reductions, relative to BAU, are 10 to 21% for sediment and 12 to 18% for total P. Importantly, these solutions still maintain comparable co-benefits for the reductions of nitrate, ammonia and energy use for all model farms, together with slightly higher reductions in nitrous oxide (9.7 to 12.3%) and similar improvements to soil physical quality scores (8.2 to 19.3). It is clear, therefore that the alternative management scenario generally produces better results albeit with some increase in annual costs and the number of on-farm measures involved.

Table 4. Impacts of optimised on-farm measure selection using measures recommended by current on-farm advice.

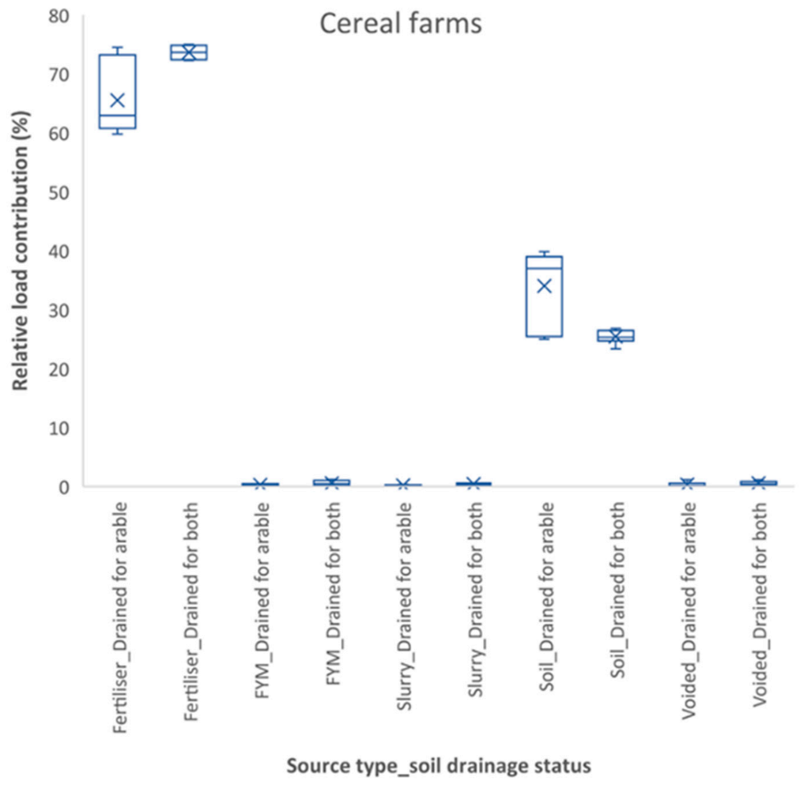
Water-Body *	Drainage Status **	Nitrate (%)	Phosphorus (%)	Sediment (%)	Ammonia (%)	Nitrous Oxide (%)	Energy Use (%)	Soil Quality (Score)	Total Cost (£/ha)
45,650	A	−5.6	−5	0	−17.9	−5.5	−14.5	0.9	−42.9
45,650	AG	−6.2	−5.3	0	−17.9	−5.6	−14.5	0.9	−42.8
45,660	A	−5.8	−4.4	0	−24.4	−5.9	−18.8	1	−44.5
45,660	AG	−6.4	−4.6	0	−24.3	−6	−18.7	1	−45
45,690	A	−6.3	−5.1	0	−25.3	−6.2	−19.2	1	−34.8
45,690	AG	−7	−5.4	0	−25.3	−6.4	−19.2	0.9	−34.7
45,720	A	−11.8	−6.5	-3.7	−24	−9.2	−22.9	6.6	−66.8
45,720	AG	−17.3	−13.6	-14.9	−23.9	−9.9	−22.2	6.8	−59.4
45,741	A	−6	−2.4	-4.2	0	−3.1	−3.9	5.7	−38.7
45,741	AG	−5.5	−1.9	-2.2	0	−3.1	−3.8	5.8	−36.6
45,830	AG	−12.5	−6.9	-2.5	−23.5	−9.8	−22.9	6.8	−73.4
45,840	AG	−12.5	−9.7	-8.8	−23.2	−8.3	−21.6	6.6	−62.1
45,850	AG	−5.9	−4.6	0	−10.7	−6.1	−18.7	0.9	−48.1
45,880	A	−6.5	−5	0	−24.8	−6.6	−19.1	0.9	−30.6
45,880	AG	−7.2	−5.3	0	−24.8	−6.7	−19.1	0.9	−30.4
51,190	A	−5.9	−4.3	0	−23.9	−6.2	−19.1	1	−41.5
51,190	AG	−6.6	−4.6	0	−23.9	−6.3	−19.2	0.9	−41.2

Note(s): * Shared common prefix (i.e., 'GB1050340') was omitted; ** 'A' stands for drained for arable and 'AG' stands for 'drained for arable and grassland'; negative values indicate a reduction relative to BAU or a cost saving.

Table 5. Impacts of optimised on-farm measure selection using measures selected on the basis of modelled pollutant source apportionment at farm scale.

Water-Body *	Drainage Status **	Nitrate (%)	Phosphorus (%)	Sediment (%)	Ammonia (%)	Nitrous Oxide (%)	Energy Use (%)	Soil Quality (score)	Total Cost (£/ha)
45,650	A	−14.5	−13.4	−13.2	−24.4	−10.5	−17.9	11.2	5.0
45,650	AG	−14.4	−12	−10.4	−24.4	−10.4	−17.9	9.9	11.2
45,660	A	−19.4	−17.5	−19.5	−15.5	−12.2	−18.7	15	−1.2
45,660	AG	−18.8	−15.8	−16.4	−25.6	12.1	−18.4	14.7	−9.7
45,690	A	−19.3	−17.1	−19.2	−25.9	−12.5	−19	16.3	−7.7
45,690	AG	−18.8	−15.4	−15.7	−14	−12.1	−19.1	11.5	−0.8
45,720	A	−20.1	−18.1	−20.8	−15.3	−12	−18.9	12.1	−12.2
45,720	AG	−19.6	−16.3	−17.5	−15.3	−11.8	−18.4	12.4	−9.3
45,741	A	−16.8	−13.9	−13.7	−24.5	−11.8	−19.2	14.7	−9.6
45,741	AG	−16.9	−11.9	-10.9	−24.5	−11.8	−19.5	13.4	19.0
45,830	AG	−17	−12.3	−10.5	−24.6	−12.6	−18.6	14.1	−25.0
45,840	AG	−14.8	−12.5	−11.9	−24.4	−9.7	−17.8	8.5	17.3
45,850	AG	−16.7	−15.1	−15.7	−21.3	−10.5	−16.8	8.2	−34.8
45,880	A	−18.6	−15.7	−16.6	−25.5	−13.1	−19.1	16.3	−8.5
45,880	AG	−18.1	−13.5	−12.9	−25.5	−12.6	−18.7	12.8	−12.9
51,190	A	−18.3	−16	−17	−13.6	−11.9	−19.2	13.7	−14.6
51,190	AG	−18.1	−14.1	−13.8	−15.2	−11.8	−19.2	11.5	−4.0

Note(s): * Shared common prefix (i.e., 'GB1050340') was omitted; ** 'A' stands for drained for arable and 'AG' stands for 'drained for arable and grassland'; negative values indicate a reduction relative to BAU or a cost saving.



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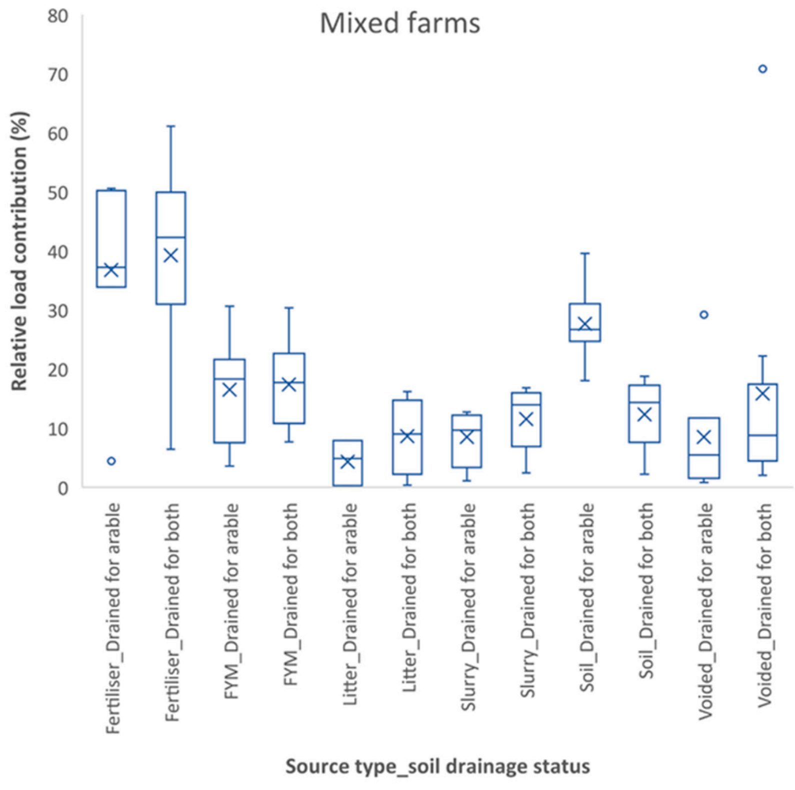


(**b**)

Figure 3. Relative contributions of key source types to dissolved P loss on the model cereal farms (**a**) and, on mixed farms (**b**), for comparative purposes.

Further examination of the on-farm measures selected by the optimisation for the two management scenarios suggests that the reasons for the improved efficacies for reducing multiple pollutants using the mechanistic scenario could be mainly due to the inclusion of measures that target pollutant delivery pathways. To illustrate this, the probability of a measure being selected in the optimised measure sets for two model farms in one specific waterbody (GB105034045720) with different soil drainage status are shown in Table 6. The optimised solutions for the two scenarios share some common on-farm mitigation methods but the key differences are the inclusion of measures 13 ('Establish in-field grass buffer strips') and 14 ('Establish riparian buffer strips') in the mechanistic scenario. Both measures are known to be effective in intercepting waterborne pollutants [28–30].

Table 6. Comparison (ratios) of on-farm measure selection probability using current advice and the alternative management scenario based on mechanistic understanding on two model farms.

Drainage Status for Model Farms	Drained Soil for Arable		Drained Soil for Arable and Grassland	
	Current Advice	Mechanistic Scenario	Current Advice	Mechanistic Scenario
Establish cover crops in the autumn	0.89	1.00	1.00	1.00
Early harvesting and establishment of crops in the autumn	0.49	0.16	0.57	0.50
Adopt reduced cultivation systems	1.00	1.00	1.00	1.00
Cultivate compacted tillage soils	0.57	0.45	0.54	0.62
Cultivate and drill across the slope	0.00	0.12	0.00	0.51
Leave autumn seedbeds rough	0.16	0.00	0.10	0.03
Manage over-winter tramlines	0.00	0.37	0.00	0.22
Establish in-field grass buffer strips	0.00	0.72	0.00	0.97
Establish riparian buffer strips	0.00	0.93	0.00	0.86
Loosen compacted soil layers in grassland fields	0.68	0.70	0.65	0.57
Allow grassland field drainage systems to deteriorate	0.00	0.00	0.00	0.21
Use plants with improved nitrogen use efficiency	1.00	1.00	1.00	1.00
Fertiliser spreader calibration	1.00	1.00	1.00	0.99
Use a fertiliser recommendation system	1.00	0.99	1.00	1.00
Integrate fertiliser and manure nutrient supply	0.72	0.98	0.91	0.99
Do not apply manufactured fertiliser to high-risk areas	0.48	0.34	0.19	0.00
Avoid spreading manufactured fertiliser to fields at high-risk times	0.47	0.39	0.48	0.34
Use manufactured fertiliser placement technologies	1.00	0.99	1.00	1.00
Use nitrification inhibitors	0.03	0.00	0.00	0.00
Use clover in place of fertiliser nitrogen	1.00	1.00	1.00	1.00
Do not apply P fertilisers to high P index soils	1.00	1.00	1.00	0.99
Move feeders at regular intervals	0.00	0.62	0.00	0.10
Construct troughs with concrete base	0.00	0.86	0.00	0.50
Increase the capacity of farm slurry stores	0.63	0.00	0.70	0.00
Adopt batch storage of slurry	0.96	0.00	0.09	0.00
Install covers to slurry stores	0.02	0.00	0.02	0.00
Install covers to slurry stores	0.02	0.00	0.02	0.00
Allow cattle slurry stores to develop a natural crust	0.58	0.00	0.02	0.00
Anaerobic digestion of livestock manures	1.00	0.00	1.00	0.00
Compost solid manure	0.00	0.00	0.09	0.00
Site solid manure heaps away from watercourses/field drains	0.87	0.00	0.99	0.00

Table 6. *Cont.*

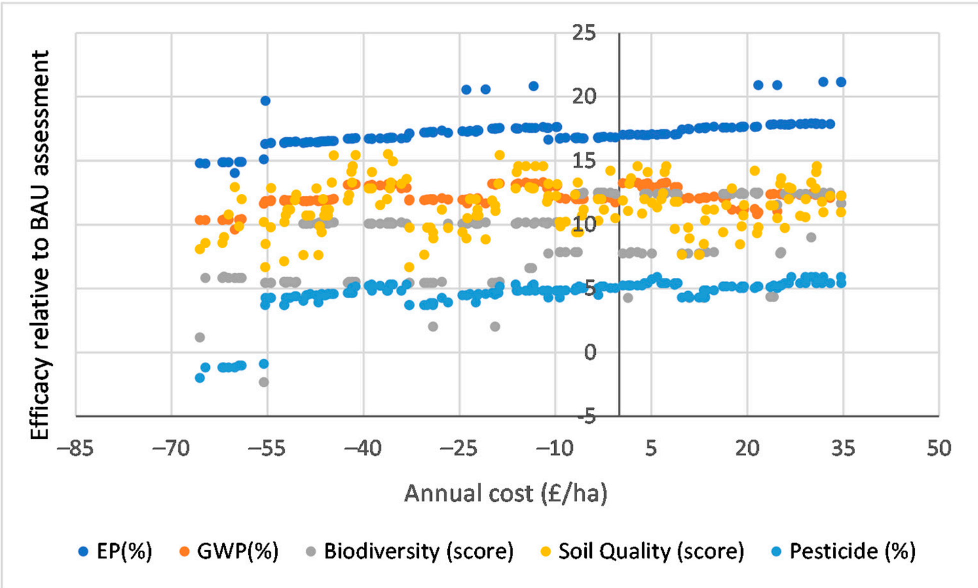
Drainage Status for Model Farms	Drained Soil for Arable		Drained Soil for Arable and Grassland	
	Current Advice	Mechanistic Scenario	Current Advice	Mechanistic Scenario
Store solid manure heaps on an impermeable base and collect effluent	0.89	0.00	0.20	0.00
Cover solid manure stores with sheeting	0.45	0.00	0.58	0.00
Use liquid/solid manure separation techniques	0.55	0.00	0.98	0.00
Manure Spreader Calibration	0.04	0.00	0.01	0.00
Do not apply manure to high-risk areas	0.95	0.00	0.16	0.00
Do not spread slurry or poultry manure at high-risk times	0.97	0.00	1.00	0.00
Use slurry band spreading application techniques	0.85	0.00	0.59	0.00
Use slurry injection application techniques	0.95	0.00	0.86	0.00
Do not spread FYM to fields at high-risk times	0.96	0.00	1.00	0.00
Incorporate manure into the soil	0.04	0.00	0.00	0.00
Re-site gateways away from high-risk areas	0.00	0.66	0.00	0.76
Farm track management	0.00	0.01	0.00	0.07
Establish new hedges	0.00	0.00	0.00	0.06
Irrigate crops to achieve maximum yield	0.00	1.00	0.00	1.00
Unfertilised cereal headlands	0.00	0.44	0.00	0.42
Undersown spring cereals	0.00	0.12	0.00	0.64
Use correctly inflated low ground pressure tyres on machinery	1.00	1.00	1.00	1.00
Avoid irrigating at high risk times	0.00	0.00	0.00	0.22
Monitor and amend soil pH status for grassland	0.98	0.92	1.00	0.97
Replace urea fertiliser to grassland with another form	0.00	0.08	1.00	0.22
Incorporate a urease inhibitor into urea fertilisers for grassland	0.95	0.92	0.00	0.65
Incorporate a urease inhibitor into urea fertilisers for arable land	0.99	0.23	1.00	0.28

With an increasing number of viable optimised solutions, the variations in emission reductions and additional farm indicator scores among the optimized populations increase. The ranges between the highest and lowest values are as high as 8% for total P and 10% for sediment. Each optimised solution also has its associated impacts on other pollutants and indicators which interact and ultimately affects the provision of ecosystem services in each waterbody. Here, for example, ranges in the predicted impacts on pesticide emissions, soil physical quality and biodiversity are all >10%. These results underscore the importance of considering co-benefits and trade-offs carefully.

4.3. Refining Selection of Optimised On-Farm Measure Sets to Account for Co-Benefits and Pollution Swapping

For the mechanistic management scenario, a large number of solutions were found which could all achieve the specified minimum reduction of both priority, and indeed non-priority, pollutants. Implementation of the measure sets selected by the optimization could incur a wide range of economic costs, ranging from cost-savings (i.e., reduced annual costs) to significant expenditure, (e.g., £40/ha/yr).

To further examine the variability of key environmental impacts, the relative change of the estimated EP, GWP100, biodiversity scores, pesticide use and soil physical quality scores with increased annual costs for each model farm were estimated. Some example outputs are shown in Figure 4 which represents model farms with different soil drainage status in two selected water bodies: GB105034045690 and GB105034045720. These two farms were chosen to illustrate the patterns across the farm sizes in the study area. The farms have an agricultural area of 97.9 ha and 210.2 ha, respectively. Clearly, the relationship between annual costs and estimated efficacies are farm specific and impact category dependent. There are higher variabilities for EP than for GWP100. The limited variation associated with GWP100 could be attributed to the dominance of N_2O emissions on its estimation and its different loadings from the targeted source types. The highest variability tends to be associated with the effects on soil physical quality and large differences can be seen in terms of the cost range. For some measure sets with lowest costs, their impacts on biodiversity and pesticide use could be negligible or even negative in some extreme cases. Importantly, a small increase in annual costs could result in step changes in the enhancement of soil physical quality and biodiversity on the target farms.



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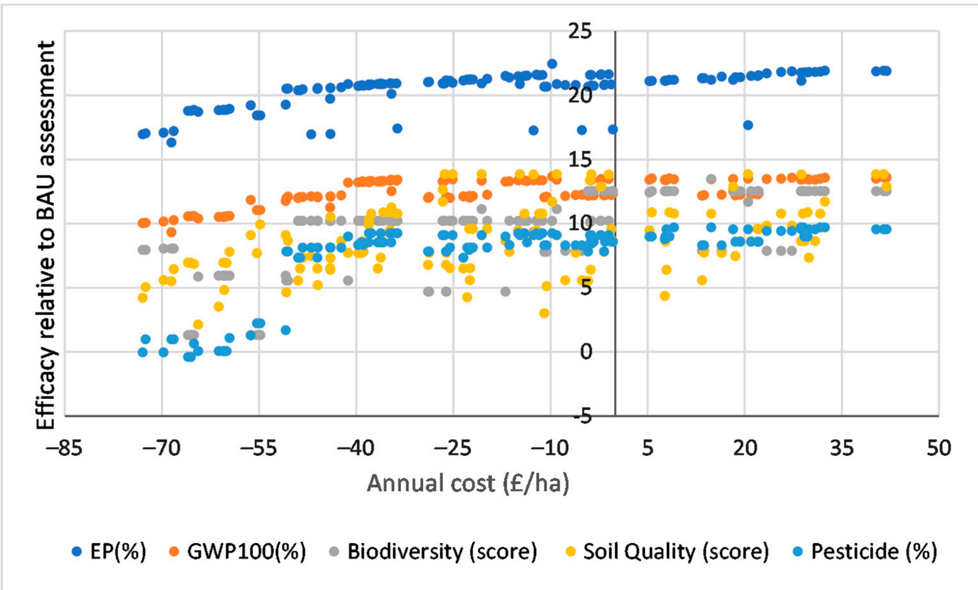

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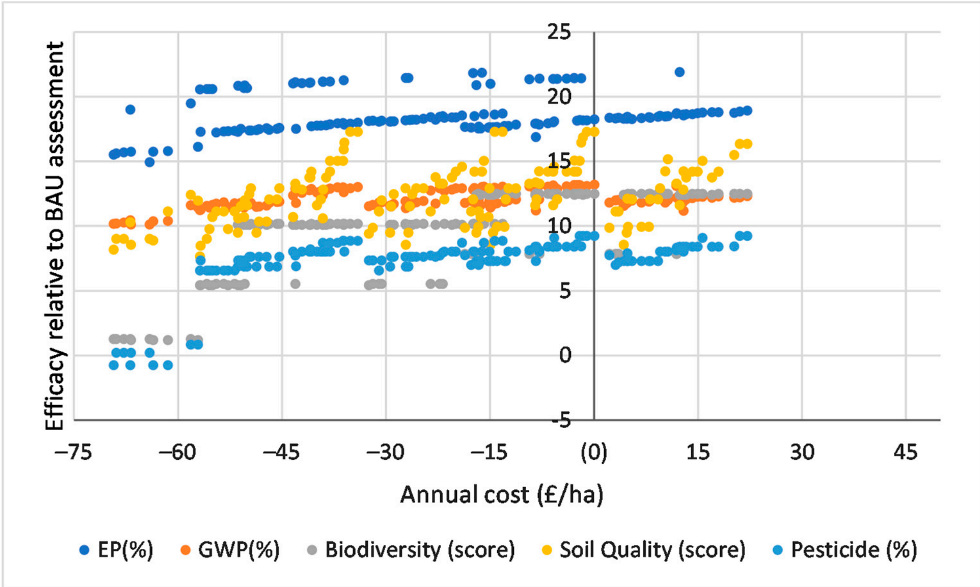

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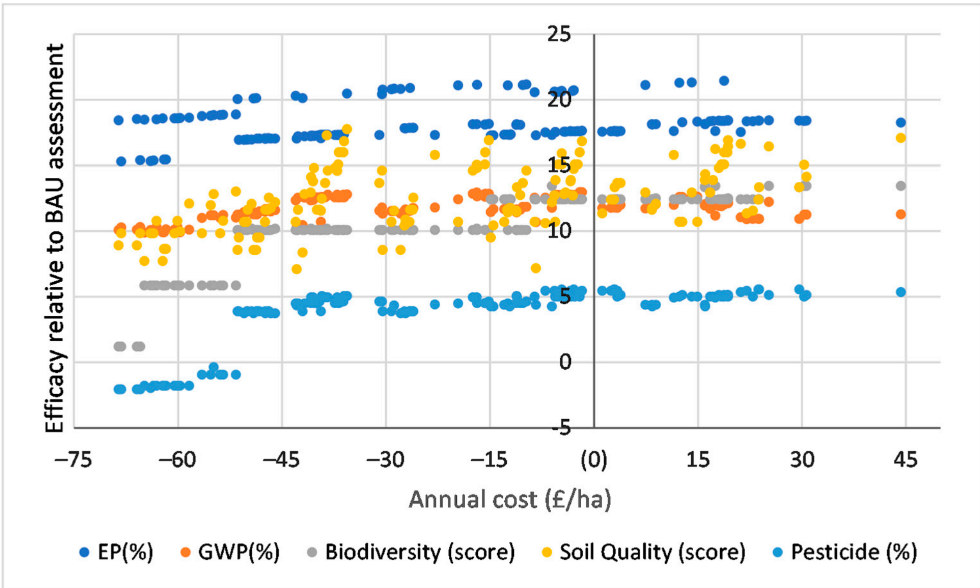

(**d**)

Figure 4. Changes in the management efficacies, relative to BAU, associated with increased mitigation costs in selected waterbodies: (**a**) model farm with soils drained for arable use in waterbody GB105034045690; (**b**) model farm with soils drained for arable and grassland use in waterbody GB105034045690; (**c**) model farm with soils drained for arable use in waterbody GB105034045720, and; (**d**) model farm with soils drained for arable and grassland use in waterbody GB105034045720.

To further narrow down the selection of measure sets for implementation requires the normalization of the impacts from different categories, such as eutrophication, global warming potential and biodiversity. This usually involves the specification of different weightings or Relative Importance Values [31]. Currently, these estimates vary in data types (qualitative vs. quantitative) and uncertainties. An improved evaluation of waterbody specific baseline conditions and their projected trajectories of development under BAU are required before any meaningful weightings can be developed. For individual farms, the practicality of various measures will also be a very important factor in the likelihood of farmer uptake [32].

5. Discussion

Modelled farm scale pollutant loadings under the BAU scenario indicate that cereal farms in the study area emit sediment and total P loads, which are major stressors to local freshwater environments, thereby undermining multiple ecosystem services. This is especially the case where farmland has been drained for arable and grassland use since this improves connectivity and the efficiency of pollutant delivery from farmland to aquatic receptor [33,34]. Considering its dominance in terms of catchment coverage, the mitigation of diffuse water pollution from agriculture is therefore a key catchment management issue in the study area. Despite this, the estimated impacts of existing mitigation measure uptake on cereal farms in the study are limited. This points to the need to consider alternative management scenarios based on mechanistic understanding of pollutant sources, as opposed to increasing implementation rates of those mitigation measures recommended by current advice. Our results, clearly demonstrate that such mechanistically defined scenarios can improve water pollution mitigation, but by incurring a range of annual costs to farms and delivering variable co-benefits.

All the modelled scenarios and mitigation options for this paper focus on in-field and on-farm management under existing farming structures where no major changes in land use or cropping, farm machinery, infrastructure or farmer skillsets are considered. Unfortunately, intensive farming activities carry intrinsic risks even with implementation of those best management practices currently recommended. Here, for instance, waterbody scale average modern background suspended sediment yields have been estimated to range between 8.9 to 9.6 t/km^2 [35]. The estimated lowest technically feasible sediment loadings with current land uses plus all available mitigation measures implemented in the Farmscoper for the study area are 13.3 t/km^2, with an average of 21.6 t/km^2. This illustrates that currently recommended best management options cannot close the sediment gap generated by modern intensive farming in the study area, even when implemented in large numbers.

Cross sector assessment by the Environmental Agency on the opportunity for the improvement of WFD status for P suggests that a reduction in catchment loadings of >50% is required, and this has been judged to be very challenging to achieve without large scale structural land use change [36]. Against this wider context, our results herein also suggest that currently recommended infield/on farm mitigation measures, even when optimized on the basis of mechanistic understanding of on-farm pollutant sources, are not able to deliver the necessary improvements in the environmental footprints of intensive cereal farming, again pointing to the need for land cover change. In order to support the rural economy, while achieving the goals of the 25 Year Environment Plan (https://www.gov.uk/government/publications/25-year-environment-plan, accessed on 9 March 2022) and a legal commitment to net zero emissions by 2050 (https://www.gov.uk/government/publications/net-zero-strategy, accessed on 9 March 2022), the new Environmental Land Management (ELM) schemes are being phased in at national scale in England. Land cover change at farm and landscape scale have been included as options, especially in the 'landscape recovery' tier where re-wilding and tree-planting have been advocated (https://www.gov.uk/government/publications/environmental-land-management-schemes-overview/, accessed on 12 April 2022).

The modelling exercise detailed herein has several limitations. A limited number of farm business surveys were undertaken in the targeted catchments to obtain farm-specific field and farm management information, such as fertilizer application rates, manure spreading, actual mitigation measure uptake rates or information on field drain distribution and maintenance. As a consequence, the true diversity of local cereal farms was not captured in full, but this limitation is common to studies using models to forecast scenarios on multiple farms. The use of the Farmscoper tool alone, rather than a model ensemble, for the quantification of farm scale footprints also ignores the potential model structural shortcomings associated with process representation. Here, ensemble modelling using a suite of models could improve the robustness of the outputs including associated uncertainty ranges. Farmscoper calculates the impacts of management scenarios by assuming multiplicative interactions between on-farm measures. This has the potential to underestimate the impacts of improved management, but equally, avoids over-forecasting outcomes. The empirical evidence base on the interactions between best management interventions continues to be limited, meaning that decision support tools must use simple generic rules to simulate such things. A detailed strategic programme at least assessing the impacts of those best management measures supported by key policy instruments is clearly warranted to improve the forecasts by decision support tools. The discrepancy between modelled emissions from cereal cropping and monitored emissions at catchment scale highlights the contributions from non-agricultural sources. On this basis, the modelled impacts of the future management scenarios are sector (i.e., agriculture) specific. The scenarios detailed herein focused on the use of long-term mean climate data and thereby did not take into account the challenges facing farming a s a result of forecast climate change in the very near or near futures.

6. Conclusions

Targeted modelling work has been undertaken using Farmscoper, in an area with intensive cereal farming. Efforts were made to improve the selection of appropriate mitigation measures by evaluating source contributions, targeting delivery pathway and optimizing on-farm measure combinations. It is clear that the mechanistically based scenario has the potential to reduce local environmental pressures more (reductions of 10 to 21% for sediment and from 12 to 18% for total phosphorus) than current farm advice (<5% for both pollutants). Importantly, however, both scenarios, are not able to deliver the substantial pollutant reductions required to achieve various policy goals. Here, most cost-neutral and cost-saving measures are already being adopted by many farmers and land managers, meaning that further mitigation of environmental impacts associated with intensive farming is likely to be costly. Critically, land cover changes need to be recommended as a potential delivery pathway for reducing externalities. The modelled outcomes and issues raised are not restricted to waterbodies in England alone. Despite continued efforts targeting the mitigation of agricultural diffuse pollution over recent decades, environmental challenges due to unintended consequences persist in areas with intensive agricultural production. It is therefore clear that a new and drastic intervention approach is required at global scale.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/w15010169/s1, Table S1: Recommended on-farm management bundles in Farmscoper; Table S2: On-farm pollutant source types and relevant mitigation methods targeting those sources plus measures targeting; Table S3: Additional annual cereal farm footprint data under BAU; Table S4: Estimated annual pollutant load reductions associated with the existing uptake of on-farm mitigation measures.

Author Contributions: Conceptualization, Y.Z. and A.L.C.; methodology, Y.Z. and A.L.C.; formal analysis, Y.Z.; investigation, Y.Z.; data curation, Y.Z.; writing—original draft preparation, Y.Z.; writing—review and editing, A.L.C.; visualization, Y.Z.; project administration, A.L.C.; funding acquisition, A.L.C. All authors have read and agreed to the published version of the manuscript.

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Data Availability Statement: The data presented in this study are available on request from the corresponding author. The data are not publicly available due to the confidentiality associated with waterbody scale June Agricultural Census data.

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Conflicts of Interest: The authors declare no conflict of interest.

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