Testing FARMSCOPER and LESA-NP against lake sediment-inferred phosphorus budgets

Crose Mere, Hatch Mere & White Mere

February 2024

NEER131

NATURAL ENGLAND

Project rationale

Many of the West Midland meres are in a ‘hyper’ eutrophic state due to diffuse and point source pollution from their surrounding hydrological catchments. Excessive phosphorus (P) loading means many of these sites are failing to meet the water quality targets outlined in their conservation objectives. To restore the meres to good ecological condition the sources of P pollution in the catchments need to be identified and reduced or removed. At present, there is an evidence gap in understanding what is required to successfully identify, quantify, and manage the sources of high P loading in their catchments.

To bridge this evidence gap, and to evaluate the benefits of specific land use interventions, P source apportionment models can be used. These models permit quantitative prediction of landscape P fluxes based on information about land use and hydrology. Such models are developed and calibrated using a combination of empirical field observations and simple theoretical assumptions. However, the appropriateness of these observations and the validity of these assumptions has not been tested at the lake catchment scale, as the models are currently designed for larger river catchments. To implement successful nutrient management strategies based on source apportionment modelling, it is critical that these models are tested and validated at the lake catchment scale.

Two specific source apportionment models are used by Natural England to inform decisions on lake management, LESA-NP and FARMSCOPER. These two models have not been linked before, but it is clear that their strengths and weaknesses are complementary, and that linking them will produce a better decision support tool for lake management. At present, FARMSCOPER is calibrated at farm scale, but has not been applied to lake catchments in the West Midland Area, and does not contain all elements needed for catchment P loading estimation. LESA-NP contains the elements needed for application to catchment P loading estimation, however it handles agricultural diffuse fluxes in a very generalised way, using very few agricultural land cover categories, and fixed export coefficient regardless of crop types, levels of fertilizer application, climate, and land drainage. While the generalised export coefficients in LESA-NP should work for average sites, they will not represent specific local situations. More importantly, the simplistic land use specification of LESA-NP means that real-world farm management decisions cannot be represented, and thus scenarios for reducing diffuse P exports cannot be predicted. Therefore, linking LESA-NP and FARMSCOPER will provide both the framework needed to represent lake P budgets, and to provide prediction of agricultural diffuse P loads from specific farming regimes.

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Keywords

Lake, Sediment, Phosphorus, Internal loading, FARMSCOPER, LESA-NP, SI-TP

Acknowledgements

We would like to thank Daisy Burris of Natural England for commissioning the project, and coordinating discussion and critical feedback on the content.

Citation

Moyle, M. & Boyle, J. F. 2024. Testing FARMSCOPER and LESA-NP against lake sediment-inferred phosphorus budgets: Crose Mere, Hatch Mere & White Mere. Natural England Evidence Reports, Number NEER131, Natural England.

Contents

1 Executive summary

1.1 Problem

Many of the West Midland meres are in a 'hyper' eutrophic state due to rural diffuse and point nutrient pollution sources leading to phosphorus (P) concentrations that exceed water quality targets outlined in their conservation objectives. To restore the meres to good ecological condition the sources of P pollution in the catchments need to be identified and reduced or removed. For this purpose, Natural England separately apply two models, LESA-NP and FARMSCOPER, the former to predict the impact of P supply on lake water TP concentrations, and the latter to evaluate the impact farm management decisions on landscape P exports. However, while FARMSCOPER is well tested at river catchment scale, neither model has been critically tested at lake catchment scale, and nor have the two models been linked before.

1.2 Objectives and scope

This project is a proof-of-concept for testing and validating a novel coupled model approach for estimating P budgets for the West Midland Meres, using a recently developed method for quantifying historical lake P budgets from lake sediment records. The project will also produce a range of land use scenarios for delivering lake water P concentration reductions to work towards meeting water quality objectives.

1.3 Approach

We focus on three case study sites across the West Midland meres: Crose Mere, White Mere and Hatch Mere. These sites were identified because of the prior existence of dated sediment records needed for testing the coupled model approach. For this study we have compiled current and historic land use data to drive FARMSCOPER and LESA-NP and used the lake sediment records together with hydrochemical monitoring data to examine and validate the model outputs.

1.4 Findings

- LESA-NP and FARMSCOPER both provide reasonable estimates of diffuse P loads at Crose Mere and Hatch Mere.
- At White Mere a very substantial but unknown past P source must have existed.
- Current information is insufficient for determining the hydrological catchment boundaries, placing limits on the reliability of P loadings to the lakes, and on their responses to those loadings.

- Arable dominance of agricultural diffuse P loadings, a feature of both LESA-NP and FARMSCOPER, is not consistent with the historical sediment analysis at Crose Mere. This instead shows broadly similar P yields for arable and grassland categories when livestock are allowed for.
- Drainage category selection in FAMSCOPER has a greater impact on predicted diffuse P exports than does either crop type or fertilizer application rates (within realistic bounds). An objective protocol for drainage category selection is needed.
- Cattle impacts on landscape P export, both diffuse and direct to water courses, appear underestimated by FARMSCOPER.
- LESA-NP internal P loading estimation is unreliable, yielding order of magnitude errors in some cases.

1.5 Recommendations

We make a series of recommendations in three areas: targeted **data gathering** to improve the reliability of interventions; **model development** to allow prediction of lake response to changing land use and climate; and **action** to implement target P reduction at specific sites.

Crucially, we have identified that current hydrochemical data collection and modelling approaches are not sufficient to reliably quantify P budgets across the West Midland meres, and this current approach may lead to ineffective management options being implemented.

1.6 Next steps

This report is a starting point for improved quantification and understanding of P budgets in the West Midland meres. We have identified a number of critical data gaps that need to be addressed in order to continue this process, and highlight that an improved modelling approach is required to predict future lake response to changes in catchment land use. This document also sets out a series of management strategy options that could be implemented at the case study lakes and more widely across the West Midland meres.

The next step is for Natural England to identify priority areas for future data collection, and to work collaboratively with land owners to identify which, if any, recommendations should be implemented based on priorities for lake restoration.

2 Introduction

This project is a proof-of-concept for testing and validating a novel coupled model approach for estimating P budgets for the West Midland Meres, using a recently developed method for quantifying historical lake P budgets from lake sediment records.

The project uses Crose Mere as a primary case study site for testing the linked model approach as there is the necessary hydrochemical and historic P budget information available for validating model outputs (Moyle 2021).

Two additional lakes, White Mere and Hatch Mere, are also used to test the application of the approach at sites where the P budget is less well quantified.

The aims of this project are to:

- Test a linked model approach that uses FARMSCOPER v5 and LESA-NP for predicting catchment P budgets
- Validate the linked model approach Crose Mere using P budgets based on measured lake sediment and catchment data (Moyle 2021)
- Apply the validated linked model to two additional lakes, White Mere and Hatch Mere, and compare the source apportionment outputs with estimated P budget
- Produce a range of land use scenarios for delivering water quality objectives for total phosphorus (TP) at Crose Mere

2.1 Description of the study sites

The Shropshire-Cheshire meres, a group of over 60 lowland lakes of high ecological importance (Fisher et al. 2009; Reynolds 1979), have long been identified as some of the most nutrient rich water bodies in the country (Fisher et al. 2009) and document some of the earliest human impacts on aquatic ecosystems in Europe (Boyle et al. 2015; Dubois et al. 2018). Historic human activity has elevated lake water P concentrations above their natural baselines (Boyle et al. 2015, Moyle 2021). These lakes formed in the extensive Oswestry-Whitchurch-Congleton moraine belt that developed ca. 23 ka during retreat of the Irish Sea glacier (Chiverrell et al. 2021; Pocock and Wray 1925; Poole 1966; Thomas 1989).

2.1.1 Crose Mere

Crose Mere is one of the Ellesmere group of Shropshire-Cheshire Meres (Reynolds 1979) and is located in Shropshire. The lake surface outflow flows east into Sweat Mere and the two meres and their surrounds form a SSSI complex. Crose Mere has no natural surface inflows but is fed by three sub-surface field drains that enter the lake on the south-west side. Two of the field drains are between approximately 200 and 500 m in length and are sub-surface for their entire length, the southernmost of which can be accessed by two manholes. The third field drain (Crose Mere Brook) extends the length of the catchment and branches into a network of smaller drains. This longer field drain becomes a surface inflow approximately 300 m before it enters the lake and has been diverted into an artificial swale to reduce the sediment supply to the lake.

The Walters (2022) site audit of Sweat Mere and Crose Mere SSSI contains a summary of evidence for the site to date.

2.1.2 White Mere

White Mere is one of the Ellesmere group of Shropshire-Cheshire Meres (Reynolds 1979) and is located in Shropshire. White Mere has one surface inflow that enters the lake on the north shore, and an outflow that leaves the lake on the western edge.

2.1.3 Hatch Mere

Hatch Mere is one of the Delamere group of Shropshire-Cheshire Meres (Reynolds 1979) and is located in Cheshire. The lake has surface inflows that enter the lake through the northern willow carr and an outflow that drains south towards Blakemere Moss.

2.2 Study site details

The study sites details are summarised in Tables 1 to 4.

Table 1. A summary of lake statistics for the three sites

	Crose Mere	White Mere	Hatch Mere
Latitude and longitude	52.8687°N, -2.8462°E	52.8907°N, -2.8719°E	52.2446°N, -2.6711°E
Lake area km^2	0.154	0.243	0.0347
Topographical catchment area km^2	1.727	0.914	2.221
Altitude (m a.s.l)	88	96	76
z-max (m)	9.3	13.8	3.8
z-mean (m)	6.13*	5.6	1.5

*Our measurement – the rest are CEH Lakes Portal values. Note Crose Mere z-mean is therefore an updated estimate.

2.3 Lake water quality summary

Tables 2-4 summarise the water quality data for the three lakes.

Table 2. Current water quality

	Crose Mere	White Mere	Hatch Mere
Alkalinity (μEq L^{-1})	3139	2058	2133
Annual mean TP (μg L^{-1})*	87	247	98
Mean TP (μg L^{-1}) for 2018-2023, showing N and standard error	N = 28 65.8 ± 3.0 2018-2023	N = 25 258 ± 17 2018-2020	N = 19 93.5 ± 6.7 2021-2023

*Most up to date complete year. For Crose Mere and Hatch Mere this is 2019. For White Mere this is 2022.

Table 3. A summary of the WFD lake typologies information for the three sites

Typology	Crose Mere	White Mere	Hatch Mere
Elevation type	Low elevation	Low elevation	Low elevation
Size type	Small	Small	Very small
Depth type	Shallow	Shallow	Very shallow
Geology type	High alkalinity	High alkalinity	High alkalinity

The below targets are informed by the Common Standards Monitoring Guidance for Freshwater Lakes (Version March 2015) and The Water Framework Directive (Standards and Classification) Directions (England and Wales) (2015).

Table 4. Water quality TP targets for the three sites. Calculations for WFD targets are based on the information in Tables 1-3, providing updated WFD GES and HES targets for Crose Mere

Lake	CSM Maximum Annual Mean TP (µg L^{-1})	WFD GES (µg L^{-1})	WFD HES (µg L^{-1})
Crose Mere	35	36	26
White Mere	35	34	24
Hatch Mere	50	47	36

- For Crose Mere the CSM target is more stringent than WFD GES, despite the recalculated GES target being lower than the previous GES target of 40 ug/L (Walters 2022).
- For White Mere and Hatch Mere the WFD GES target is more stringent than the CSM target.

3 Overview of methods and data

This section is an overview of methods used in this study, described here for Crose Mere. A similar methodology was applied at White Mere and Hatch Mere. The full methodological details and data descriptions for all three sites can be found in Section 11. A graphical summary of the methods can be found in Figure 3.

3.1 Crose Mere topographic catchment

Crose Mere's topographical catchment is well-defined (Figure 1), but the hydrological balance of the lake is uncertain because we find a mismatch between the measured outflow and the topographic catchment (Moyle 2021). This leads to two different estimates of the total P budget because of the differing estimates for the outflow P load. The estimates are described in Section 6.

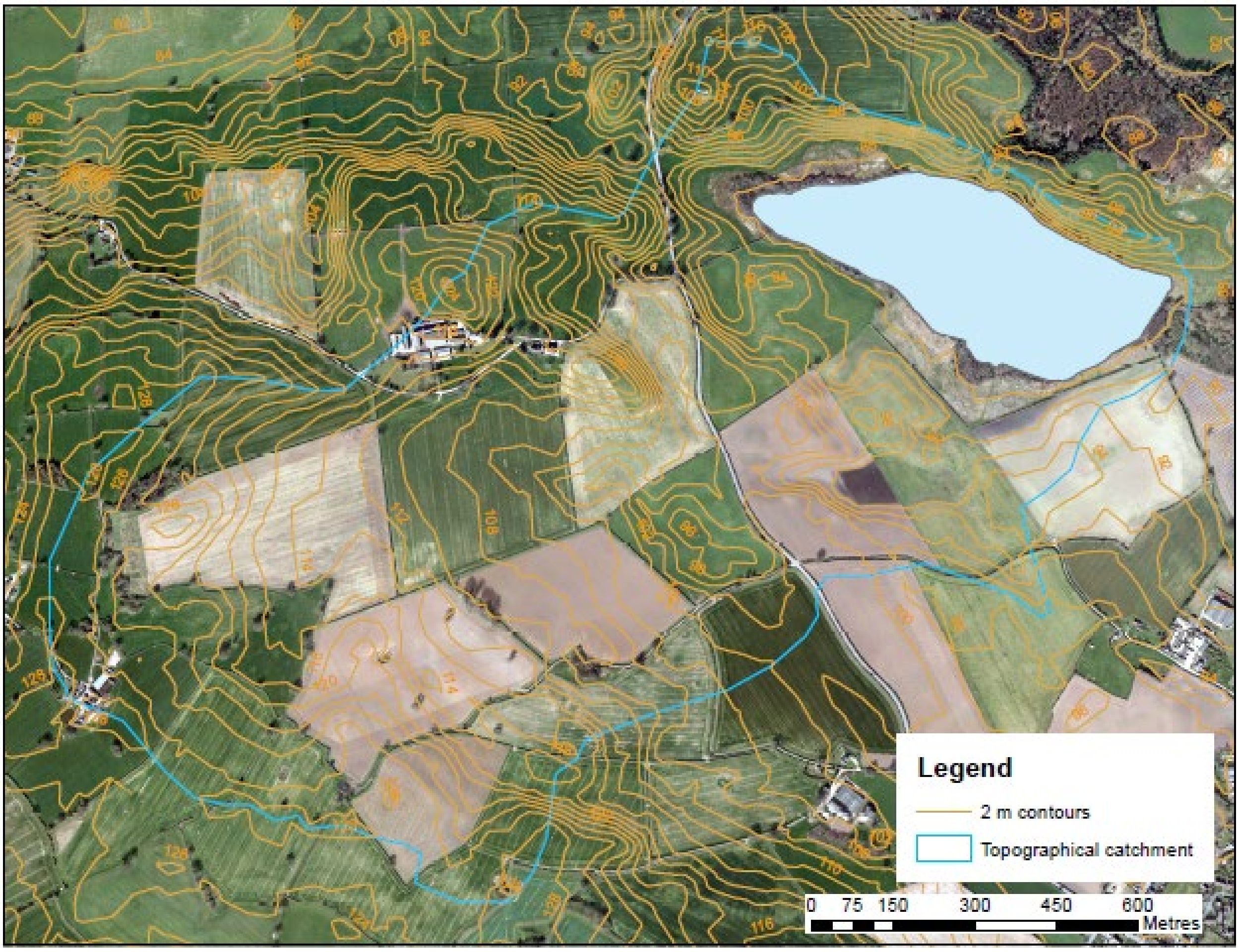

Figure 1. Topographical map of Crose Mere, with 2 m LIDAR-based contours superimposed on the 2012 Air photograph

3.2 Crose Mere catchment land use data

3.2.1 Land cover

Land cover was found using CEH Land cover maps available via DIGIMAP Download as vector shapefiles for 1990, 2000, 2007, 2015, and then 2017 to 2021. Data for 1935 was taken from a non-georectified image (The Dudley-Stamp survey) in DIGIMAP ROAM (map viewer).

3.2.2 CEH crop maps

Crop data was gathered using CEH Land Cover plus crop maps, available via DIGIMAP download as shapefiles for 2016 and 2017 (shown in Figure 2). The crop maps are also available for 2018-2021, but only as images shown in the DIGIMAP ROAM map viewer, therefore land cover classes were manually assigned.

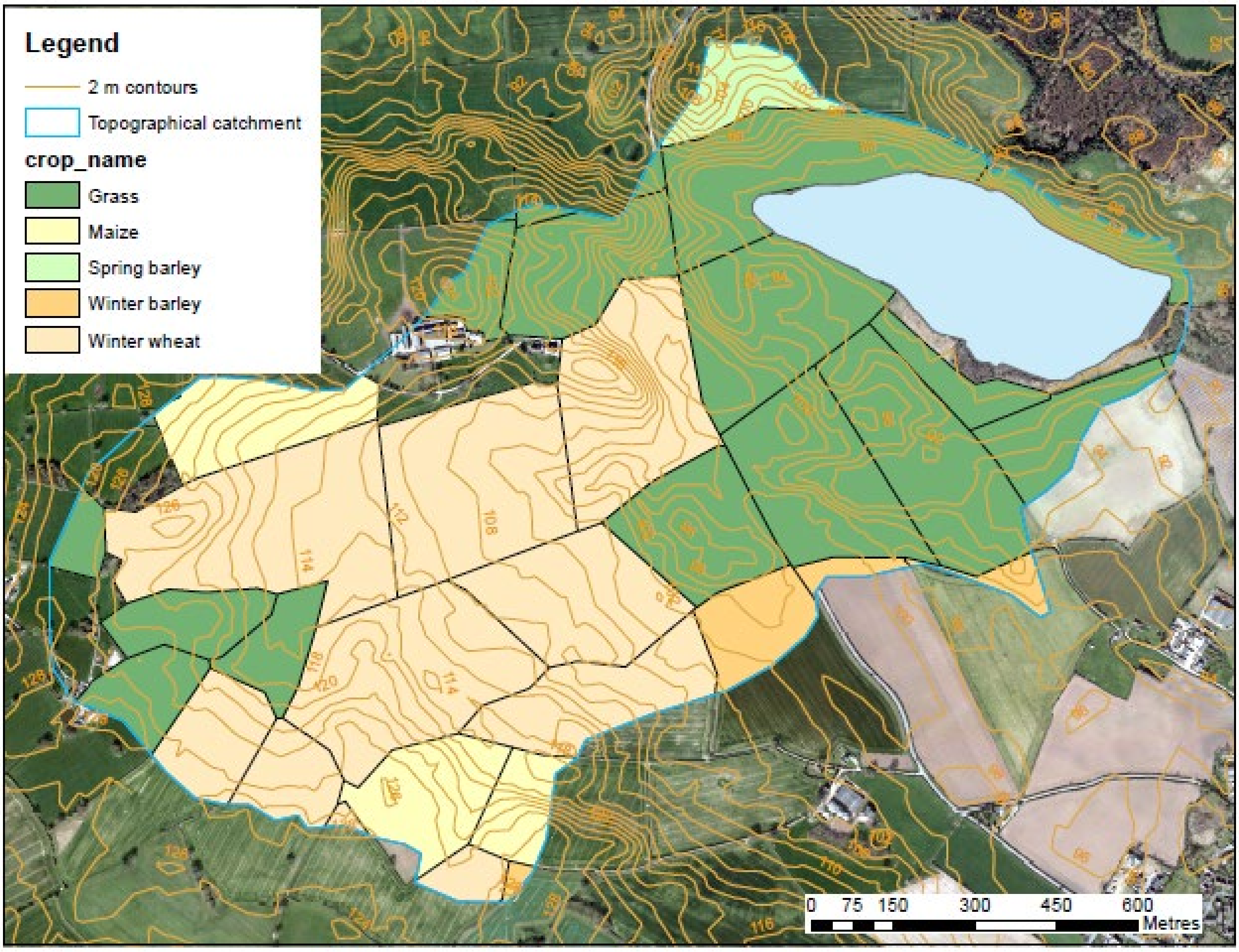

Figure 2. CEH Land Cover plus crop map, illustrated for 2017

For time periods prior to 2016, crop information was inferred from county agricultural. The county data have been compiled in approximately decadal steps from 1905 forwards and adjusted for the Crose Mere catchment.

3.2.3 Livestock

Livestock numbers are based on field data from Moyle (2021) and Google Earth Pro imagery.

3.2.4 Wildfowl

Wildfowl numbers are estimated using data from Moyle (2021) and Atkins (2010).

3.2.5 Population

The number of households in the catchment are taken from the Natural England Site Audit for Crose Mere (Walters 2022) and occupancy is assumed to be at national average rates.

3.2.6 Fertiliser application

Fertilizer application rates are assumed to be at national average rates. These data are obtained from British Survey of Fertilizer Practice reports (DEFRA, 2014) combined with historic total N and P application values (ktons/yr, Withers et al., 2014), scaled to overlapping application rates (kg/ha).

3.3 FARMSCOPER and LESA-NP

The data described above was input into FARMSCOPER and LESA-NP. For FARMSCOPER, further detail was required for the catchment, including data on soil, drainage, and farm management. The decisions made are detailed in Section 11. For LESA-NP the original input data were revised according to the data above. Importantly, the approach for estimating internal P loading was updated to reflect observed data (Moyle, 2021). This is discussed at length in Section 7. The model setup process is summarised in Figure 3.

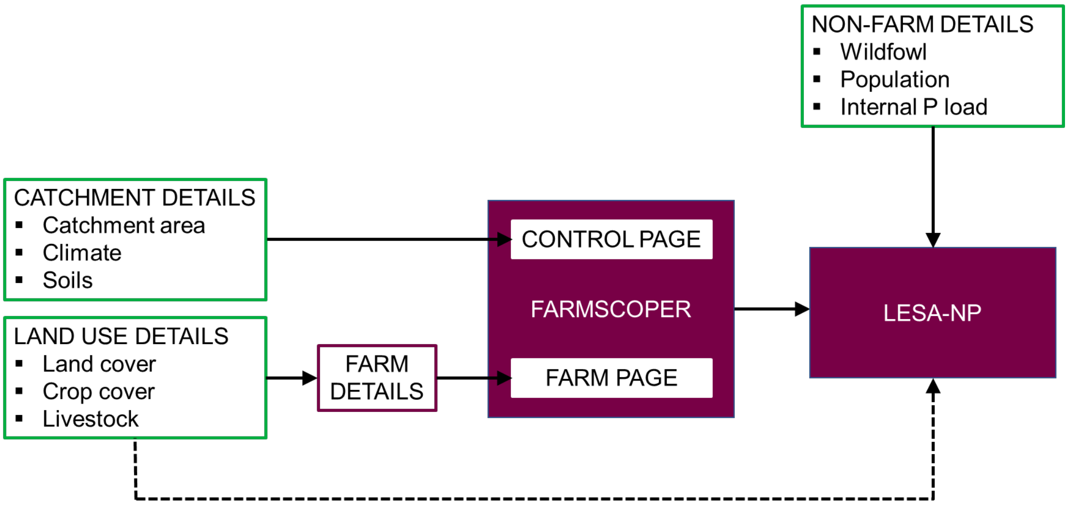

Figure 3. A graphical summary of the methodology used in this study

4 Methods limitations

The detailed methodology can be found in Section 11.

4.1 Model limitations

- FARMSCOPER is a simplified version of the PSYCHIC (Davison, 2008; Stromqvist, 2008) model, which itself is a simplified representation of processes that are likely to be important. FARMSCOPER does not represent spatial variation in drainage, does not allow for field gradients and flow concentration/dispersion, therefore particular local flow behaviours will not be correctly represented in the model. Also, by representing drainage using broad categories, uncertain choice of category has an exceptionally large impact on outcome.
- A number of other on-farm processes are not properly represented by FARMSCOPER, notably direct excretion of animal waste to water courses.
- LESA-NP uses generalised export coefficient values, therefore model output is highly dependent on the choice of values, which are not tailored to site-specific circumstances.
- LESA-NP does not handle internal load correctly, yielding highly uncertain values.

4.2 Catchment data limitations

- Topographical catchments can be approximated using conventional 5 m digital terrain models, or from higher resolution LIDAR data (Ordnance Survey), but it is not likely that these represent the hydrological catchment. It is therefore not possible to be certain how much water flows through any particular water body and without proper identification of the catchment boundary, it is not possible to determine exactly which P sources lie within the hydrological catchment. These limitations are probably unavoidable, although we have recommended a method for reviewing hydrological catchment areas in the meres.

4.3 Land use data limitations

- FARMSCOPER and LESA-NP are run, as is usual, without fully accurate local details, in that average stocking density, population, bird counts, atmospheric loading, etc., are estimated from general information or incomplete local sources. These provide sensible information but are not fully accurate. Without site-specific information, the true lake P budget cannot be quantified.
- A similar issue applied to the historical dimension for data on past drainage, stocking, crop types, and fertilizer application rates. These are again all reasonably estimated by downscaling national or county trends, but are not correct for specific local cases.

4.4 Sediment record limitations

- The Moyle (2021) lake sediment mass balance modelling methods has strong theoretical foundations, the underlying principles having four decades of empirical validation and calibration. Nevertheless, the lake wide P sedimentation rate history must be measured. Reliable measurement requires two things. First, robust sediment chronologies must be developed for sediment cores from the lake. Second, those cores must be combined to generate lake wide average rates, correcting for sediment focussing. The sediment approach is described in detail in Moyle and Boyle (2021). For Hatch Mere, and particularly for Crose Mere, we have good quality chronologies, and sufficient sediment cores to achieve this. At White Mere we have only generalised chronologies, and no specific information for the last century, such that results are currently highly uncertain.

5 Results

5.1 Sediment record of the annual total P budgets at Crose Mere, Hatch Mere and White Mere

The principle of using lake sediments to calculate total lake P budgets is outlined at length in Moyle (2021) and Moyle and Boyle (2021). In brief, when considering long-term averages (in the order of decadal), the externally supplied P load ($L_{external}$, combining water supplied by surface and subsurface water flows, atmospheric deposition, animal contributions (Birds and direct cattle) is equal to the total P export. The latter comprises two components, both of which can more easily be measured directly than $L_{external}$. These components are: L_{out}, the outflow load, which can be measured by monitoring of lake water TP and outflow discharge; and $L_{sediment}$, the apparent sedimentary P burial load, which can be measured using dated lake sediment cores. A small part of $L_{sediment}$ can be rereleased to the lake water, a process known as internal P loading. In the long run this component is subsumed in estimates of $L_{sediment}$, comprising in effect, reduced capture of P by the lake sediment.

A complication to this simple view occurs at many of the meres in that a proportion of outflowing water does so below the surface. Conceptually, the subsurface component of this export can be regarded simply as part of L_{out}. In practice it cannot be directly measured and must therefore be calculated from a water budget. Fortunately, limiting values for this are easily estimated. The lower limit for L_{out} comprises the measured value. The upper limit is based simply on an estimated value for L_{out} assuming that the whole topographical catchment were contributing to the total discharge. At Crose Mere, the $L_{sediment}$ has been estimated to be 35.2 kg P/year (Moyle 2021), while the lower limiting L_{out} has been measured at 14.9 kg P/yr. The upper limiting value is estimated to be 27.4 kg/yr. Table 5 summarises these two limiting budgets.

Table 5. A summary of the P loading values used for Crose Mere

P load in kg/yr	Lower limiting Lout (Measured)	Upper limiting Lout (estimated from topographical catchment)
Sediment burial load	35.2	35.2
Outflow load	14.9	27.4
External load (sum of previous)	50.1	62.6
Water loading q_s, m/yr	1.408	2.592

We conclude that the external P load to Crose Mere lies between 50 and 63 kg P/yr.

At Hatch Mere, the mean $L_{sediment}$ value has been estimated at 85 kg/yr (Boyle et al., 2015), but no discharge measurements have been made at the flow stream. Based on assumed mean annual runoff of 0.25 m/yr, and the catchment and lake are data from Methods>Catchment land use data at White Mere and Hatch Mere>Hatch Mere (2.221 and 0.0347 km^2, respectively), and mean lake water TP = 79.8 SE 4.4%, the L_{out} = 44.3 kg/yr. Combined, this gives 129 kg/yr estimated external P load.

At White Mere, the mean $L_{sediment}$ is estimated to be in the order of 250 kg/yr (Moyle 2015), though it must be stressed this number is highly uncertain at present owing to the complex nature of the lakebed. As at Hatch Mere, no discharge measurements are available from which to estimate L_{out}. Based on assumed mean annual runoff of 0.25 m/yr, and the catchment and lake are data from Methods>Catchment land use data at White Mere and Hatch Mere>White Mere (0.914 and 0.243 km^2, respectively), and mean lake water TP = 374 SE 4.5%, the Lout = 85 kg/yr. Combined, this gives 459 kg/yr estimated external P load.

5.2 The Vollenweider method

The lake mass balance modelling approach of Vollenweider (1969) offers an alternative approach to estimating the external load, based on predicted values for R_P, the lake phosphorus retention coefficient.

$$R_P = \frac{L_{external} - L_{out}}{L_{external}} = \frac{L_{external} - TP \times q_s}{L_{external}}$$

From which, the external load can be estimated from lake water TP, q_s and R_P.

$$L_{external} = \frac{TP \times q_s}{(1 - R_P)}$$

According to Vollenweider (1969), R_P can be estimated from the water loading, q_s (the water discharge of the lake normalised by like area, $m^3/m^2/yr = m/yr$) and an effective P sedimentation velocity, v (m/yr).

$$R_P = \frac{v}{v + q_s}$$

Knowing the correct value for v is problematic (Moyle 2021, Moyle and Boyle 2021), but it is reasonable to assume that it lies between 5 and 20, which provides us with limiting estimates for $L_{external}$.

For Crose Mere, and using both values of q_s from Table 5, and TP = 45 mg/m^3 (Moyle 2021), this predicts a range of 44 - 157 kg P/yr, of which the high value is the least likely.

At Hatch Mere, with a q_s value of 16 m/yr, and TP = 79.8 mg/m^3, the predicted $L_{external}$ is 58 – 100 kg P/yr. And at White Mere, with a q_s value of 0.94 m/yr, and TP = 258(374) mg/m^3, the predicted $L_{external}$ ranges 373-1313 kg P/yr.

5.3 FARMSCOPER and LESA_NP application to Crose Mere - current status

5.3.1 LESA-NP

To apply LESA-NP to the lakes it was first necessary to correct for some errors in the worksheet, and second to use revised land use and stocking data. A series of trials were run (Table 6) to implement the corrections and revisions.

Table 6. LESA-NP P output (kg P/yr) variation with original export coefficients, correcting for errors. Trial 5 has P export coefficients for Arable and Pasture derive from FARMSCOPER. Note the absence of livestock in Trial 5 is because FARMSCOPER includes the livestock P contribution within the Arable and Pasture categories. WwTW = wastewater treatment works

Source	Trial 1 Original	Trial 2 Repaired links	Trial 3 Internal load emprical	Trial 4 Correct catchment	Trial 5 FARMSCOPER ECs
Septic systems	0.8	21.8	21.8	3.9	3.9
Birds	41.9	61	4.2	4.2	4.2
Anglers	9	0	0	0	0
WwTW	0	0	0	0	0
Atmospheric	4.8	2.3	2.3	2.3	2.3
Sediment	723.2	346.8	7.5	7.5	7.5
Arable	229.6	153.5	153.5	54.3	38.7
Pasture	54.4	41	41	16.2	6.5
Livestock	0	0	0	12.2	NA
Total	1063.7	626.4	230.3	100.6	63.1

Trial 1 is included for completeness. It is the original version, and as received the worksheet had many broken links between the FRONT END tab and the tabs with source data. Consequently, the values used to drive the model are not those for Crose Mere, and the output it therefore incorrect. The exceptionally large sediment P load is in part due to the great over-estimation of the lake area arising from this fault. An error in the December coefficients for seasonal distribution of water flow (a value of 50% instead of 0.5%) adds slightly to the exceptionally high total P load.

Trial 2 has the correct links to lake and catchment data supplied by Atkins (2010), and correct December seasonal distribution coefficient. This reduced the total P load from 1064 kg/y to 626 kg/yr.

Trial 3 has Sediment P load derived from a simple empirical model (see Section 7.4 for explanation), and has the geese count altered to use the average bird abundance rather than peak abundance. This takes just over 400 kg/yr from the total load.

Trial 4 uses the original LESA-NP export coefficients, but replaces the Atkins (2010) values for land cover, stocking, and population density by the values from this report (Table 19, mean for 2015-2021). This greatly reduced the catchment area because Atkins (2010) used a combined catchment for Crose Mere, Whattal Moss and Sweat Mere, the latter two lying down stream of Crose Mere. On the other hand, Atkins (2010) had no data for livestock, so a substantial contribution is added to the budget from this source. This gives a total of 93 kg/yr P supply from the catchment. This value is higher than the lake sediment estimated P budget (50-63 kg/yr), but within the range of the much less certain Vollenweider estimate (44-157 kg/yr).

5.3.2 FARMSCOPER

LESA-NP for Crose Mere, **Trial 5**, is driven by P export coefficients derived from FARMSCOPER. FARMSCOPER was driven using land cover data (illustrated in Figure 2), with Control and Farm settings as described in Section 11.4. Export coefficients for use in LESA-NP were calculated from land cover specific P export values (kg P/yr, Report tab, Total loss) divided by the corresponding landcover area (ha), yielding and export coefficient in kg/ha/yr. The output from LESA-NP based on the FARMSCOPER export coefficients is shown in Table 6 (column, Trial 5).

5.4 FARMSCOPER and LESA_NP application to White Mere and Hatch Mere – current status

Table 7 shows P export by source category for White Mere and Hatch Mere, based on LESA-NP default export coefficients, and for export coefficients derived from FARMSCOPER.

Table 7: P export by source category for White Mere and Hatch Mere. All values kg P/yr

	Hatch Mere LESA-NP default values	White Mere LESA-NP default values	Hatch Mere FARMSCOPER	White Mere FARMSCOPER
Septic systems	64	10.9	64	10.9
Birds	1.8	3.7	1.8	3.7
Anglers	0	0	0	0
WwTW	0	0	0	0
Atmospheric	0.5	3.6	0.5	3.6
Sediment	17.6	-	17.6	-
Arable	21.2	11.9	33.2	11.9
Pasture	39.6	10.2	61.8	17.8
Livestock	28.3	3.1	0	0
Groundwater	0	0	0	0
Total	173	43.4	178.9	47.9

5.5 FARMSCOPER and LESA_NP application to Crose Mere – historical change 1905 to present

Table 8 shows historical P export values for Crose Mere, both total and source-specific, based on LESA-NP output using FARMSCOPER export coefficients, with FARMSCOPER driven by historical parameters from Table 19.

Table 8: Historical P export values for Crose Mere based on LESA-NP output using FARMSCOPER export coefficients

	People	Birds	Atmospheric	Arable	Grassland	Total
2021	3.6	4.2	2.3	37.9	5.0	53.0
2020	3.6	4.2	2.3	37.8	4.9	52.8
2019	3.6	4.2	2.3	38.2	5.4	53.7
2018	3.6	4.2	2.3	45.0	4.8	60.0
2017	3.6	4.2	2.3	45.0	4.8	60.0
2015	3.6	4.2	2.3	45.6	5.2	60.9
2007	3.6	4.2	2.3	47.4	5.7	63.2
2000	3.6	4.2	2.3	46.3	7.3	63.7
1990	3.6	4.2	2.3	45.0	8.7	63.8
1985	3.6	4.2	2.3	63.9	6.4	80.5
1975	3.6	4.2	2.3	62.6	6.4	79.1
1965	3.6	4.2	2.3	65.5	7.3	82.9
1950	3.6	4.2	2.3	61.1	7.3	78.5
1945	3.9	4.2	2.3	57.5	6.3	74.2
1935	3.9	4.2	2.3	24.8	7.4	42.5
1925	4.5	4.2	2.3	28.8	7.2	47.0
1915	4.8	4.2	2.3	29.0	6.9	47.2
1905	4.5	4.2	2.3	31.3	6.2	48.4

5.7 Potential future scenarios for P reduction at Crose Mere

Five scenarios for phosphorus export reduction are tested here for the Crose Mere catchment, and used to predict the impact on lake water TP in the coming decades. An abrupt reduction in external P load will increase the significance of the 7.5 kg/yr internal P load, the magnitude of which will slowly decline over the coming decades. A full dynamic model remains experimental (Boyle et al., 2023), but a simple exponential decay model with an annual rate constant of -0.1 /yr for long term sediment P stores (Penn et al., 1995), and a rate constant of -1 /yr to represent short terms stores (sediment and marginal plants) give a plausible rate of change. These are converted to lake water TP values using the Vollenweider method, anchoring R_P to the current TP of 68 µg/L which gives it a value of 0.57, and q_s = 2.55 m/yr.

$$TP = (L_{external} + L_{internal})\frac{(1 - R_P)}{q_s}$$

Scenario 1

Best practise FARMSCOPER optimization of nutrient export, following the approach of Atkins (2014) River Wye action plan. The Optimise function in FARMSCOPER_evaluate identifies combinations of measures that optimise P export reductions for differing numbers of measures chosen from a list of 115 options. Applied to Crose Mere, Figure 4 shows percentage cumulative P reduction as a function of the number of measures included.

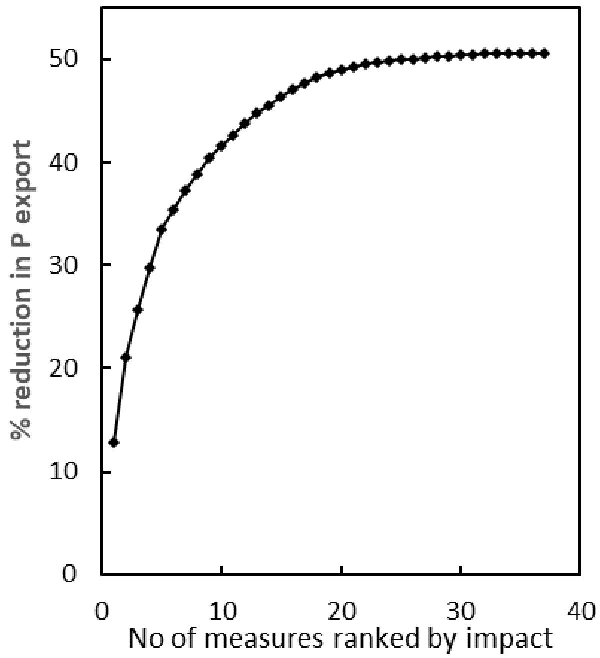

Figure 4. Percentage cumulative P reduction as a function of the number of measures included (in ranked order).

Figure 4 shows that 37 of the 115 measures reduced P export, leading to an Optimiser maximum P export reduction of 51%, with the top 5 most impactful measures leading to a reduction of 33%. These measures were, in decreasing order of impact, Establish cover crops in the autumn 12.8%, Uncropped cultivated areas 8.3%, Establish riparian buffer strips 4.6%, Adopt reduced cultivation systems 4.0%, Plant areas of farm with wild bird seed / nectar flower mixtures 3.7%. For comparison, the River Wye catchment plan (Atkins 2014) found on average a P export reductions of 44% and 30%, for optimiser maximum and Top 5, respectively. Given the uncertainty in these predicted reductions, the level of prior P reduction measures being set at national average values rather than at site specific observed values, the Scenario 1 reduction assumed a rounded average of 40%.

Scenario 2

50% conversion of arable to woodland. Half of the 81.8 ha of arable land at Crose Mere (FARMSCOPER EC = 46 $mg/m^2/yr$) are replaced by woodland (LESA-NP EC = 2 $mg/m^2/yr$).

Scenario 3

50% conversion of farmland to woodland based on sediment inferred P budget data. The sediment P budget tightly constrains the total agricultural diffuse supply. The sediment record does not directly tell us what kind of agricultural sources contribute to this, but the historical sediment record reconstruction analysis (Section 7.3) suggests that arable and pastural sources produce similar contributions. Here we replace half of the total land area by woodland (EC – 2 $mg/m^2/yr$), assuming an average EC of 29.7 $mg/m^2/yr$ for farmland based on sediment budget corrected for birds, atmospheric and septic system contributions based on LESA-NP.

Scenario 4

Cattle exclusion from riparian land.

Scenario 5

100% conversion of farmland to woodland based on sediment inferred P budget, and septic systems replaced by mains sewer connections.

The lake water TP predicted for these scenarios are shown in Figure 15.

Discussion

5.8 Topographical versus hydrological catchment area

Moyle (2021) shows that the outflow discharge from Crose Mere is lower than expected given local area river discharges and the size of the topographical catchment area (Figure 1). This shows either that the hydrological catchment area is poorly represented by topographical catchment area, or that a substantial part of the water discharge from Crose Mere is via subsurface pathways. If the former is correct, then a problem arises in that while we can know the area of the hydrological catchment we cannot know its mapped boundaries. The latter situation is certainly the case for lakes such as Betton Pool and Bomere Pool, where no surface outflow is present, and all outflowing water is subsurface.

Hydrological monitoring of the inflow and outflow at Crose Mere (Moyle 2021) means upper and lower estimates for the hydrological catchment can be produced, whereas for many of the other meres the lack of any hydrological data means the situation is less certain. The hummocky landscapes at meres such as White Mere and Betton Pool mean that the topographical catchment is ambiguous. In the absence of reliable outflow data, or the total absence of surface outflow as at Betton Pool and Bomere, and knowledge that much water must be escaping below ground level, we can have very low certainty about catchment area. This situation leads to two problems. First, it means that we cannot know which point sources within a landscape should be included in the catchment to a particular lake, leading to uncertainty over calculated P loading. Second, if hydrological catchment is unknown, then the water load to the lake is also unknown, and it is then not possible to reliably estimate phosphorus retention by the lake. The expected quantitative impact of an uncertain hydrological catchment area can be illustrated using the Vollenweider (1968) model:

$$TP = L_{external} \frac{(1 - R_P)}{q_s}$$

Where TP is the lake water TP concentration, $L_{external}$ is the external P load to the lake, R_P is the phosphorous retention coefficient, q_s is the areal water loading to the lake and is therefore reliant on hydrological catchment area (see Moyle 2021, and Moyle & Boyle, 2021 for more explanation and context).

The impact of varying the catchment to lake area ratio can be seen in Figure 5A. If we assume a fixed rainfall rate and P export coefficient (where an export coefficient is a landscape P yield expressed in $mg/m^2/yr$), it is apparent that lake TP will vary strongly with catchment to lake area ratio, directly impacting any forecasting of mitigation benefits. However, uncertainty in catchment area also impacts estimation of external P loads, as rearranging the previous equation gives:

$$L_{external} = TP \frac{q_s}{(1 - R_P)}$$

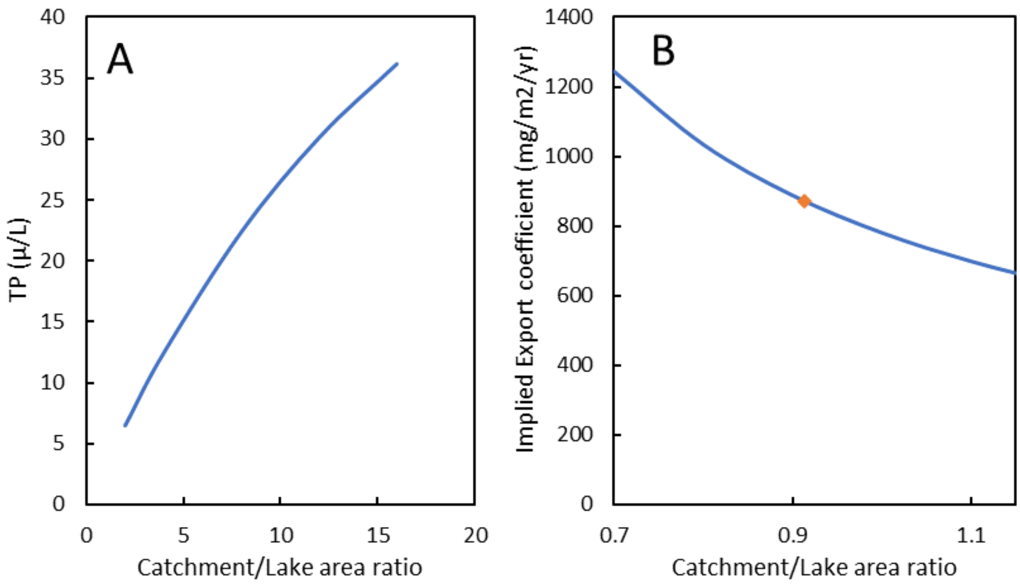

Figure 5. The impact of catchment to lake area ratio on: A) lake water TP. B) Estimated diffuse P loads (expressed as an export coefficient) calculated from measured TP. The orange diamond represents our best estimate of the catchment to lake area ratio at White Mere

We illustrate this effect in Figure 5B. Here we have applied the above equation to White Mere, assuming a current TP value of 258 µg/L, mean annual discharge of 0.25 m/yr, R_P based on Kirchner and Dillon (1975), and assuming there is no internal P load. Expressing the external load as a catchment yield (equivalent to an export coefficient), we find a factor of two variation in estimated value arising from a realistic range of catchment areas. This shows that in the absence of reliable catchment area data, P budgets are also uncertain. The impact of this at Crose Mere is shown in the next section.

5.8.1 Summary

Without the true hydrological catchment area, we cannot reliably calculate the expected lake water TP concentration from diffuse loads, and conversely we cannot reliably calculate catchment diffuse P yields from lake water TP measurements. For meres with large enough catchments that substantial outflow streams are present, and where topographical catchments are well-defined, we can constrain the phosphorus budgets within reasonable limits. Where topographical catchments are poorly defined, and where outflow streams are absent, the impact of diffuse loads on lake water TP, and prediction of diffuse P inputs to lakes, can be highly uncertain.

5.9 Diffuse agricultural P load estimates

The P load calculated from the lake sediment record represents the total catchment P load. It is therefore directly comparable with the output from LESA-NP because the model includes a complete range of both point and diffuse P sources to produce an estimate of the total catchment P load. FARMSCOPER, on the other hand, estimates only the diffuse agricultural load and allowance must be made for P sources that are not included in the model before comparison with the other two estimates is possible.

To compare outputs of diffuse agricultural load for the three case study sites, the LESA-NP and sediment-inferred P load estimates have been adjusted to represent only the diffuse agricultural load. For LESA-NP this was achieved by using only diffuse agricultural load components from the model. For the sediment-inferred output, the LESA-NP estimate of the P load supplied by non-agricultural diffuse sources (i.e. the P load from septic tank, bird, and atmospheric deposition contributions) was subtracted from the total P load value. A comparison of the diffuse agricultural P load estimates from FARMSCOPER, LESA-NP, and the lake sediment records from the three case study sites can be seen in Figure 6, which also shows calculated P load estimates using export coefficient values from White and Hammond (2009). Table 9 shows compares the export coefficients used in this comparison. The values at White Mere and Hatch Mere are greater because of the choice of drainage category in FARMSCOPER; in contrast to Crose Mere, grassland exceeds arable, so drainage status was set as Drained for Grassland Use.

Table 9. Export coefficient for diffuse loads from arable and improved grass (mg/m^2/yr). Non-FARMSCOPER values are from White and Hammond (2009)

Source	Arable	Improved grassland
LESA-NP	65	22
FARMSCOPER Crose Mere	46.3	8.1
FARMSCOPER White Mere	65.1	43.9
FARMSCOPER Hatch Mere	102.1	45.8
Upland and moorland	60	30
Intensive mixed farming	90	80
South Devon	60	40
Limestone and chalk	65	10
Eastern England	22	3
North West England lowland	60	10

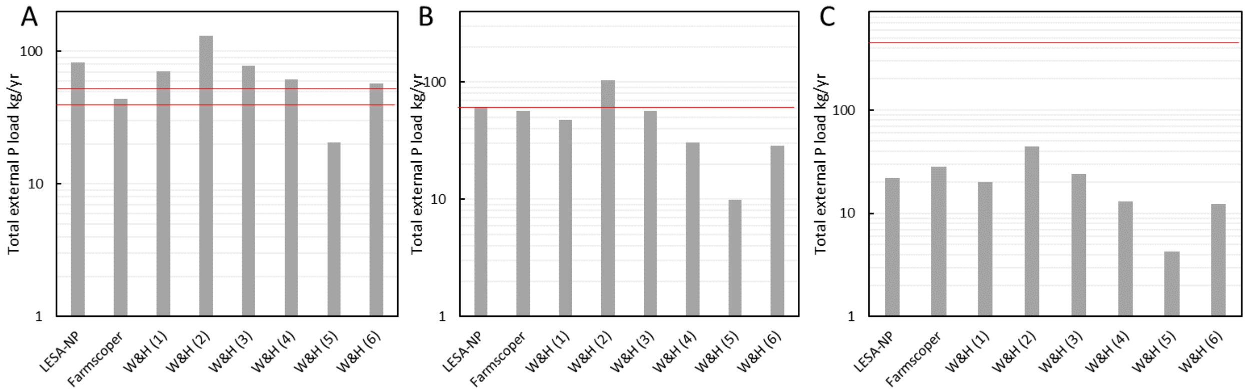

Figure 6. A comparison of LESA-NP, FARMSCOPER, and White and Hammond (2009) UK regional export coefficient estimates of diffuse P load from combined arable and improved grassland with the sediment-inferred estimate of total diffuse agricultural P load (red line) for A, Crose Mere; B, Hatch Mere; C, White Mere. The White and Hammond (2009) export coefficients used are (1) Upland and moorland, (2) Intensive mixed farming, (3) South Devon, (4) Limestone and chalk, (5) Eastern England, (6) Northwest England lowland

For Crose Mere, two red lines are shown, one for each assumption about the size of the hydrological catchment area. FARMSCOPER and LESA-NP are applied using the larger of these two, the topographical catchment area.

At Crose Mere (Figure 6A) we see that LESA-NP, run with export coefficients of 65 and 22 $mg/m^2/yr$ (arable and improved grassland, respectively), overestimates the total diffuse agricultural load. This is likely due to the relatively high P export coefficient for arable land (Table 9). FARMSCOPER agrees closely with the sediment inferred values (Figure 6A). For comparison, Figure 6A also shows arable, improved grassland, and combined loads calculated for Crose Mere based on UK regional export coefficients (White and Hammond, 2009). The range encompasses the LESA-NP and FARMSCOPER values, and on average the values are similar to the sediment inferred value.

A very similar finding is seen at Hatch Mere, though with the LESA-NP and FARMSCOPER values falling much closer to each other (Figure 6B). The sediment inferred diffuse load is less reliable at this site owing to need to correct for the high contribution of septic tank P contributions. Nevertheless, the agreement with the findings at Crose Mere is encouraging.

At White Mere, the situation is wholly different. While the LESA-NP and FARMSCOPER values still agree, both are more than an order of magnitude lower than the sediment inferred diffuse P load (Figure 6C). While the currently available sediment inferred values are of low reliability at White Mere, the Vollenweider method also predicts a very high P load, so the mismatch is evidently real. The two simplest explanations for this mismatch are that the catchment is incorrectly identified, or that an important source of phosphorus has been disregarded. While it is certainly the case that the catchment area is highly uncertain in this hummocky landscape, it is equally clear that the morphology of the land does not permit a catchment of double the current estimate, still less 10 times the size. We must therefore conclude that a substantial phosphorus source is not currently known. A large part of this source is, at present, probably internal phosphorus loading, with phosphorus leaking from the sediment to the water column. This possibility is discussed further in the internal phosphorus load section below. However, internal P loads derive from past high external loads, and it is this that remains unknown.

5.9.1 Historical P loads

An important point to note is that while the combined arable and grassland contributions have been compared successfully with the sediment inferred load at Crose Mere and Hatch Mere, this approach does not test whether the separate arable and grassland contributions are reliably measured. An opportunity to test this is offered by the historical perspective at Crose Mere from the sediment record (Figure 7).

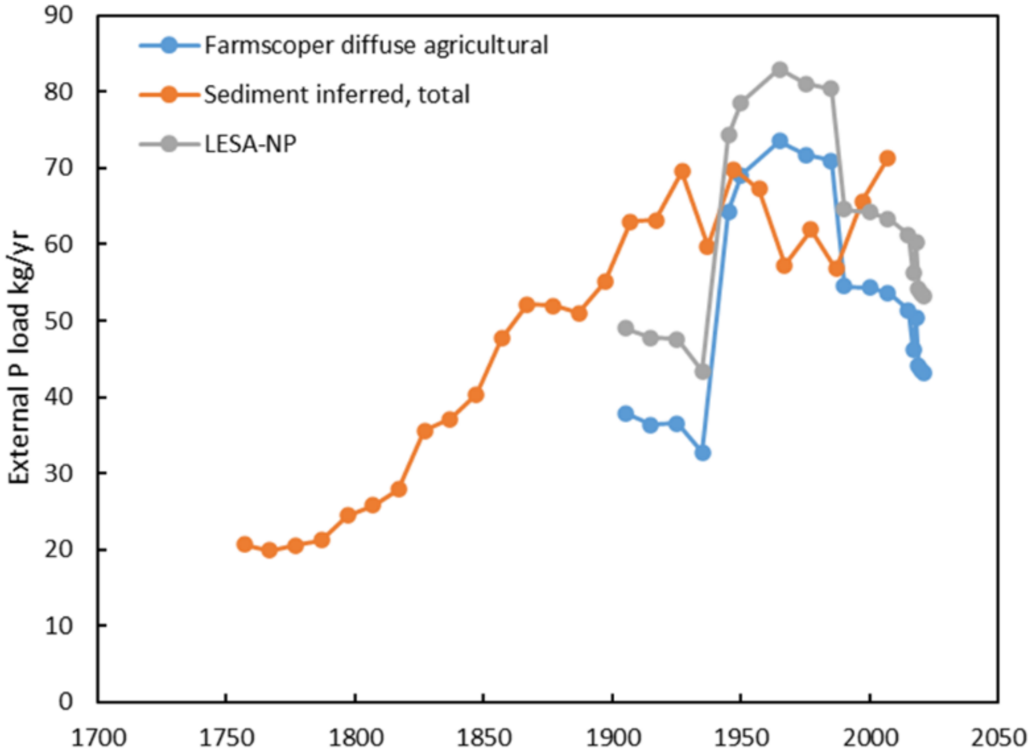

Figure 7. External P load calculated using FARMSCOPER/LESA-NP compared with sediment inferred total P export

Figure 7 reveals broadly similar total magnitudes for the sediment-inferred and modelled P load histories, but shows distinctly different patterns. Driven primarily by the proportion of arable land, FARMSCOPER, and consequently LESA-NP, both show a period of high diffuse P loading from the late 1940s until the 1990s. This pattern is not present in the lake sediment record. In contrast, the lake sediment record shows a steady rise through the

19th century followed by gentle declines from the early 20th century. The recent increase since the late 1990s is an artefact of sediment diagenesis and should be ignored (see Boyle et al. 2023). The most striking discrepancy between the two records relates to the increase in arable land brought about by the drive for increased food production related to the Second World War. Over this period, the models predict a sharp increase in diffuse nutrient exports as a result of the switch to arable, whereas the sediment record shows no such effect.

The implication of this finding is that, at least in the 1940s and 50s, grassland with cattle produced a similar diffuse phosphorus export to arable land. In Section 7.3, we address the question of whether this is still the case. In other words, are current models either overestimating diffuse loads from arable land or underestimating diffuse loads from grazed land.

5.9.2 Drainage and diffuse P loads in FARMSCOPER

FARMSCOPER offers a choice of soil drainage types and additional options for presence of field drainage in the modelled catchment. Deciding what drainage category to apply is not simple, however. There are two issues, first, information from the British Geological Survey, UK Soil Observatory maps, is in 1 km^2 grid cells leaving uncertainty as the correct values for a small catchment. Second, different parts of the landscape have different drainage, with arable likely located on the most freely-draining soils, and grass where the land is less well drained, yet a single drainage category must be selected. At Crose Mere we know that tile drain networks are present (Atkins 2010), but do not know which fields are drained. We selected the drainage category “Drained for arable use” in our modelled catchments based on arguments presented in Atkins (2014) that modern machinery has compacted soils requiring tile drains to be put in place. Nevertheless, for our study sites it is not possible to determine the farm-scale drainage practices from mapped information, although we do know field drains are present, and this is likely to be the case across most catchments.

Given that assignment of soil drainage category in FARMSCOPER is essentially arbitrary, it seems prudent to test the consequences of decisions about the presence of field drainage. Figure 8A illustrates the effect of changing drainage category while maintaining a consistent land use and stocking density. We can see that adding field drainage into the modelled catchment has a large impact on diffuse P export. Indeed, sensitivity testing of FARMSCOPER suggest that selection of soil drainage category has a greater impact on diffuse P exports than choice over land cover or stocking density.

In FARMSCOPER drainage efficiency is set by a drain connectivity coefficient, Drainage flow, a parameter set by default at 90%. We have no catchment-specific data on drainage efficiency for any of our study sites, however we have tested the choice of Drainage flow value on controlling P export. Figure 8B shows a linear dependence of P export on Drainage flow, for both drained categories. This shows that for the drained soils represented in FARMSCOPER, the efficiency of the field drains is as important as the choice of drainage category.

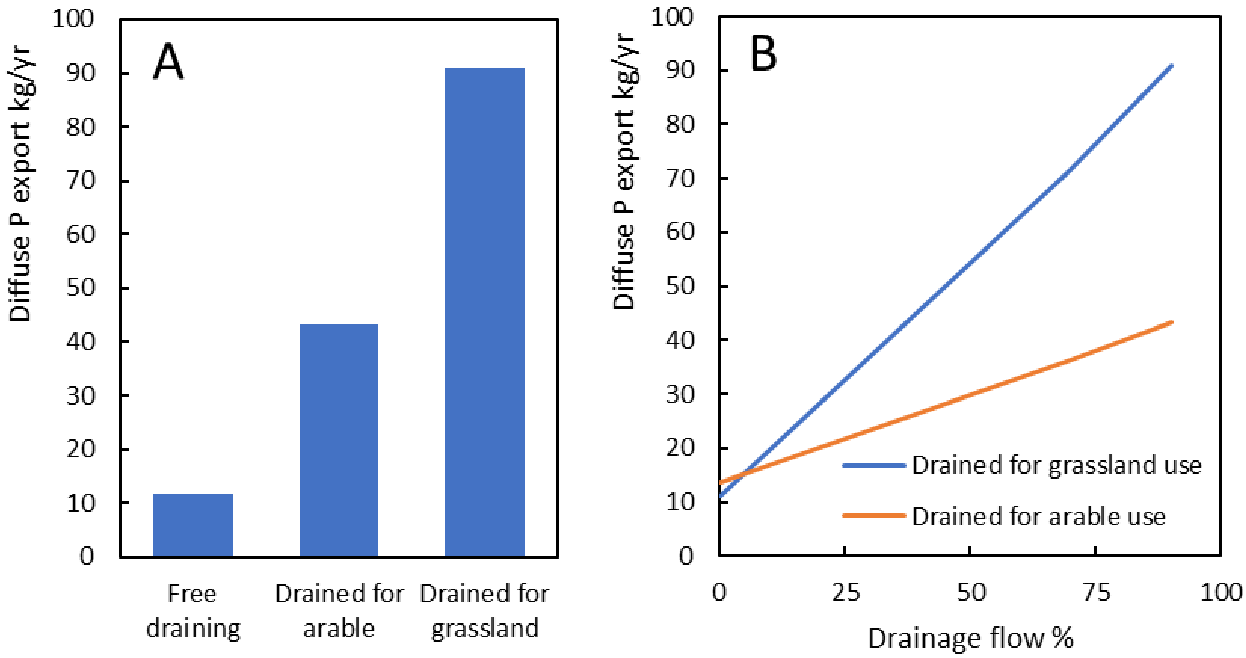

Figure 8. Soil drainage categories (A), and drainage flow (B), impacts on predicted diffuse P exports

An additional complexity arises from spatial heterogeneity in drainage. At Crose Mere, we know that tile drains are found in the grassland, but have no specific information about the arable land. In treating the catchment as a single farm, no mechanism is provided in FARMSCOPER to allow for this spatial variation as the whole catchment is treated as a homogeneous hydrological entity. Modelling the catchment as two separate farms, one representing the arable land and the other grassland with cattle, allows some testing of how this is handled within the model. This separation allows arable land to be "drained for arable use" and grassland to be "drained for grassland use".

Because of the very large impact of drainage on P exports, temporal variation in the nature and condition of field drains also regulates rate of transfer of phosphorus to streams and lakes. For example, the discrepancy between sediment inferred diffuse P exports and model predicted exports in the 1940s (Figure 7) could be easily explained by changes in drainage. The problem is that quantitatively reconstructing drainage conditions in the past is not simple, and even the present-day condition of these drainage networks is poorly known and spatially variable (Zhang et al., 2016).

5.9.3 Summary

Soil drainage category and drainage efficiency are simultaneously crucial to the functioning of FARMSCOPER and problematic to assign. To drive change of phosphorus exports in FARMSCOPER, we find drainage to be the most powerful tool; the historical variations in P export shown in Figure 7 can all be replicated by small adjustments in drainage. Yet, while qualitative historical information is available, we lack a protocol for implementing this quantitatively in FARMSCOPER. It is clear that we need to find better ways of specifying drainage condition in catchments if we are to reliably predict diffuse loads using FARMSCOPER.

5.10 Cattle and FARMSCOPER

Cattle consume P as a part of their diet, storing a part of this during growth, but excreting most in the form of faeces and urine. Assuming an adult weight of 500 kg, consuming dry matter at a rate of 2.5 % of body mass per day will result in the excretion of 14 kg P/yr given typical P concentration in grazing matter. Only a proportion of the excreted P will leave the field and enter into draining water, representing the diffuse cattle contribution to the total P budget. Johnes (1996) suggests that 3 % of excreted P will be exported from a dry field, and 6% from riparian fields, from which export rates of 0.42 and 0.84 kg P/yr would be expected from adult cattle. FARMSCOPER allows for the impact of cattle via a linear contribution per head, at a rate that varies with drainage category. These rates are compared with other published values in Figure 9. The FARMSCOPER rates are lower than those just quoted because herds will not all comprise adult cattle. Figure 9 shows that the FARMSCOPER P export rate for the “Drained for grassland use” category is comparable to, though slightly lower than, most other published values. The FARMSCOPER “Free drained” and “Drained for arable use” categories predict P export rates far lower than most of the published estimates, leaving us to question which of these is more accurate.

Two limitations to these figures should be mentioned. First, the quoted rates are based on the principles outlined by Johnes (1996), where 3 % of the voided cattle waste are assumed to transfer to water courses. Johnes (1996) assumed an average cattle annual P export of 7.6 kg/head/yr, half the rate expected for full grown dairy or beef animals. So, the figures quoted are substantially lower than would be expected for mature cattle, and may be regarded as a lower limiting estimate. Second, the quoted rates relate to diffuse export of phosphorus from fields, and do not include any direct contributions to water courses. Direct contributions arise from tendency of cattle to excrete in the vicinity of water if given the opportunity. Gary et al. (1983), based on a study undertaken in Colorado report 6.7-10.5 % of defecation and 6.3 to 9.0 % of urination directly to the stream, with discharges usually soon after drinking. A more recent study of Hampshire chalk streams found 11.7 % of defecation was supplied direct to streams (Bond et al. 2014). Short term observations at Crose Mere (Moyle, 2021) found 15 % of defecation below winter high water levels. Johnes (1996) recognises the importance of proximity to water, proposing that riparian land exports 6 % of waste. Consequently, we can conclude that FARMSCOPER uses a lower export coefficient per head of cattle for dry field exports, and essentially disregards direct contributions to streams and lakes. There is an option in FARMSCOPER to specify the percentage of Fields Next to Watercourse, but testing this showed little impact on P export. FARMSCOPER is therefore likely underestimating the contribution of cattle in agricultural P exports.

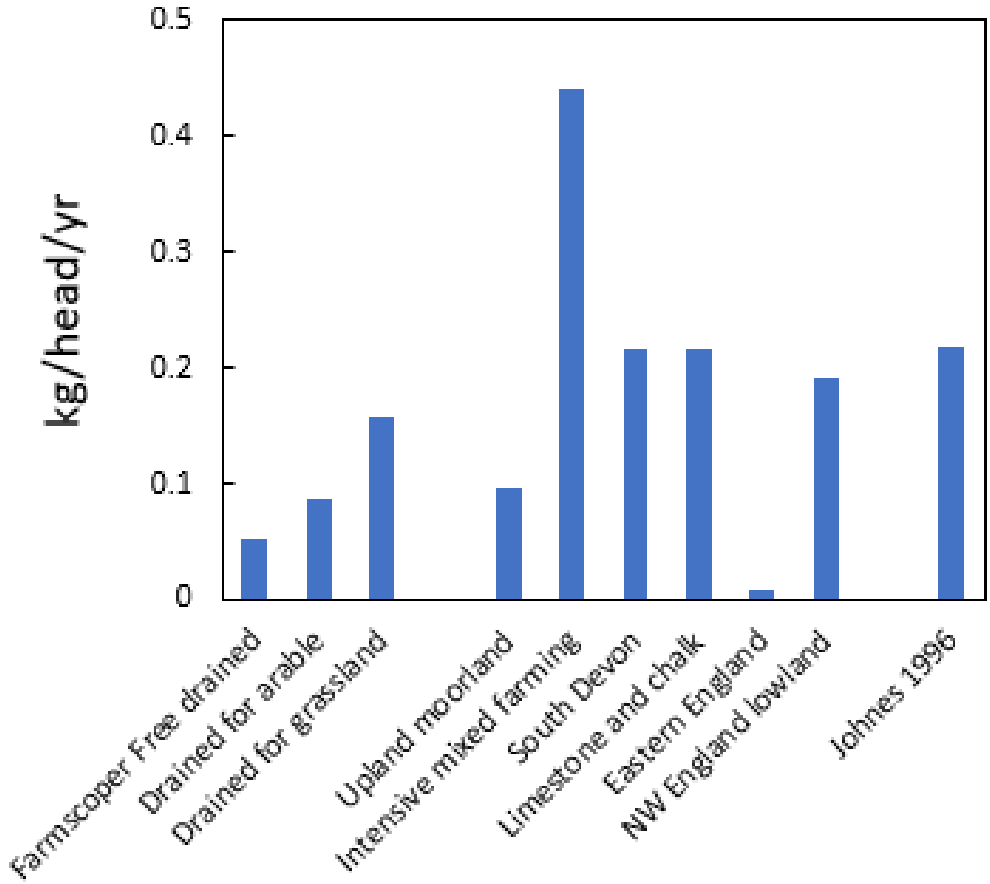

Figure 9. Per head P indirect delivery to runoff from cattle

To illustrate this, we can consider the application of FARMSCOPER at Crose Mere. With the drainage category “Drained for arable use”, the arable land (53 %) contributes 38 kg P/yr and the grassland (47 %) contributes 7 kg P/yr, just 16 % of the total P export. However, if 10 adult cattle have access to the lake, as regularly observed by Moyle (2021), then an additional 14 kg P/yr would arise from direct excretion in the water, bringing the grassland contribution to 36 % of the total P export. Based on the FARMSCOPER drainage setting described above, this scenario assumes that grassland and arable have identical drainage. If the grassland is set as “Drained for grassland use”, as is the case in at least a portion of the Crose Mere catchment, then even without direct excretion to water the grassland is predicted by FARMSCOPER to contribute 46 % of P export. This rises to 55 % if the direction excretion to water is included.

While this is inevitably speculative, in that the FARMSCOPER drainage categories cannot be independently verified at Crose Mere, it does serve to show that the mismatch in historic P export (Figure 7) can realistically be attributed to FARMSCOPER underestimating cattle contributions to landscape P exports in lake catchments.

5.10.1 Summary

Based on comparison with published export coefficient values and the historic lake sediment record, it seems likely that FARMSCOPER provides a lower limiting estimate of cattle impacts on diffuse agricultural P export to water. This is particularly important in for lakes in catchments where cattle have free access to water.

5.11 Internal phosphorus loading

The importance of internal P loading to a lake P budget is dependent on the timescale over which the process is considered (Figure 10). Over short (~annual) timescales internal loading is seen as a separate P source from external loading, whereas over long (~decadal) timescales internal loading is considered linked to the external load. Over long timescales internal loading is a function of the external load, as the sediment captures a portion of the external load supplied to the lake, and the amount of P supplied by the sediment to the water column (i.e. internal load) cannot exceed the maximum historical external loading.

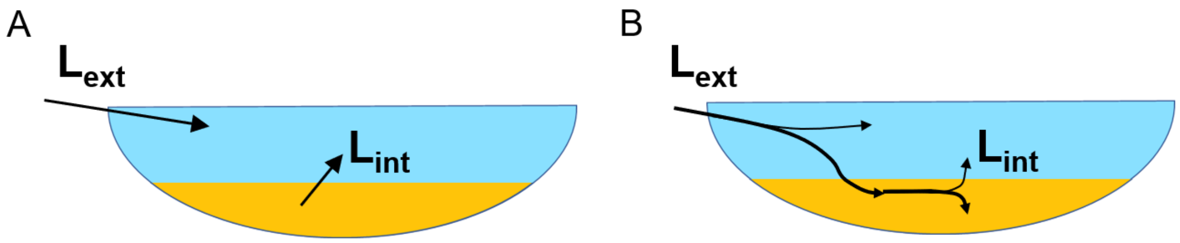

Figure 10. External (L_{ext}) and internal (L_{int}) loading viewed over two timescales. A, The short term view (~annual) considers L_{ext} and L_{int} as two separate sources; B, the long term (~decadal) view considers L_{ext} and L_{int} as linked sources where *L_{int} = f(historical L_{ext})* and *L_{int} < max. historical L_{ext}*.

For Crose Mere, LESA-NP (Atkins 2010) gives extremely high values for internal P loading which dominates the total P budget. The value given by LESA-NP of 347 kg/yr is in excess of the external load of 93 kg/yr, and known historical loading, and therefore cannot be correct. Based Moyle's (2021) hypolimnetic P measurements for Crose Mere (see below) we argue that the LESA-NP estimate is excessively high and offer an alternative method of calculating a realistic value. In this section of the report, we explain this argument and outline how we think internal loadings should be approached. To achieve this it is necessary to outline how internal loads are estimated, and to show why the estimates are uncertain.

5.11.1 Estimating internal loading

There are a number of ways to estimate internal phosphorus loadings for lakes, none of which are fully reliable.

1) Incubation experiments

This is where a sediment core is collected from the bed of a lake, and the release rate of P from the sediment to the overlying water is measured in the laboratory. This is the method used by Atkins (2010), based on measurements made by Kilinc and Moss (2002) that gave an annual mean of 25 mg/m^2/day. The approach is valuable for understanding processes of P release, but it is questionable whether measurements made this way can be directly applied to lake sediments. Mechanical disturbance of the sediment record by the process of coring, and change of environment, mean that the measured rates do not necessarily reflect the *in situ* lake processes. Furthermore, it is not a simple process to scale up a measurement to represent the whole lake, and to represent the whole year. For LESA-NP, Atkins (2010) assumed that a measurement of Kilinc and Moss (2002) applied to half of the lake bed area, and was applicable across half the year. This yielded a total internal load of more than 300 kg P/yr, a load greater than all external loads combined.

2) Measuring potential P loading

This method has not been applied at Crose Mere, and is mentioned here only for the sake of completeness. Chemical methods may be applied to lake sediments to separate P into different chemical fractions, with different degrees of stability in the sediment. This allows direct estimation of the amount of phosphorus in the sediment that is potentially available for supply as internal P loading. The problem with this approach is that there is no way to work out how much of this fraction is actually supplied as internal loading.

3) Measuring accumulation of hypolimnetic P during lake stratification

This approach directly measures the amount of P that accumulates in the hypolimnion (the deeper layers of the lake water) during summer stratification, and assumes that all of this P has leaked from the sediment to the lake water column. Moyle (2021) applied to this approach to Crose Mere across two periods of lake stratification in 2017 and 2018. At Crose Mere, Moyle (2021) found P release rates for the two years of 3 and 1 mg/m^2/day (mean values across the whole lake area), over periods of two months and four months, for 2017 and 2018 respectively. These peak values are at the low end of the values measured by incubation experiments at White Mere of between -7 and 147 mg/m^2/day (Kilinc and Moss, 2002), and are lower still if expressed across the whole year – 0.44 and 0.38 mg/m^2/day for 2017 and 2018 respectively compared to 25 mg/m^2/day at White Mere. However, the measurements of Moyle (2021) must be regarded as overestimates of the true sediment internal loading because of mechanisms within the lake. Part of the accumulated hypolimnetic P will have been supplied from the epilimnion (the upper layers of the lake water). Here, P released from decaying algae settling through the water column, P that would have originally been mixed through the whole water column, will be trapped in the hypolimnion during stratification. We can expect then that the figures of 0.38 and 0.44 mg/m^2/day are representative of upper limiting rates.

4) Inflow-outflow budgets

With sufficient monitoring of inflow and outflow P loads (P concentrations and water discharges), the net internal load (the total internal P load minus any recapture by sedimentation) can be estimated directly by difference. This is what was done at Søbygaard in Denmark (Boyle et al. 2023). It can also be estimated indirectly using retention coefficients (Nurnberg 1984, 2009), visualised in Figure 11.

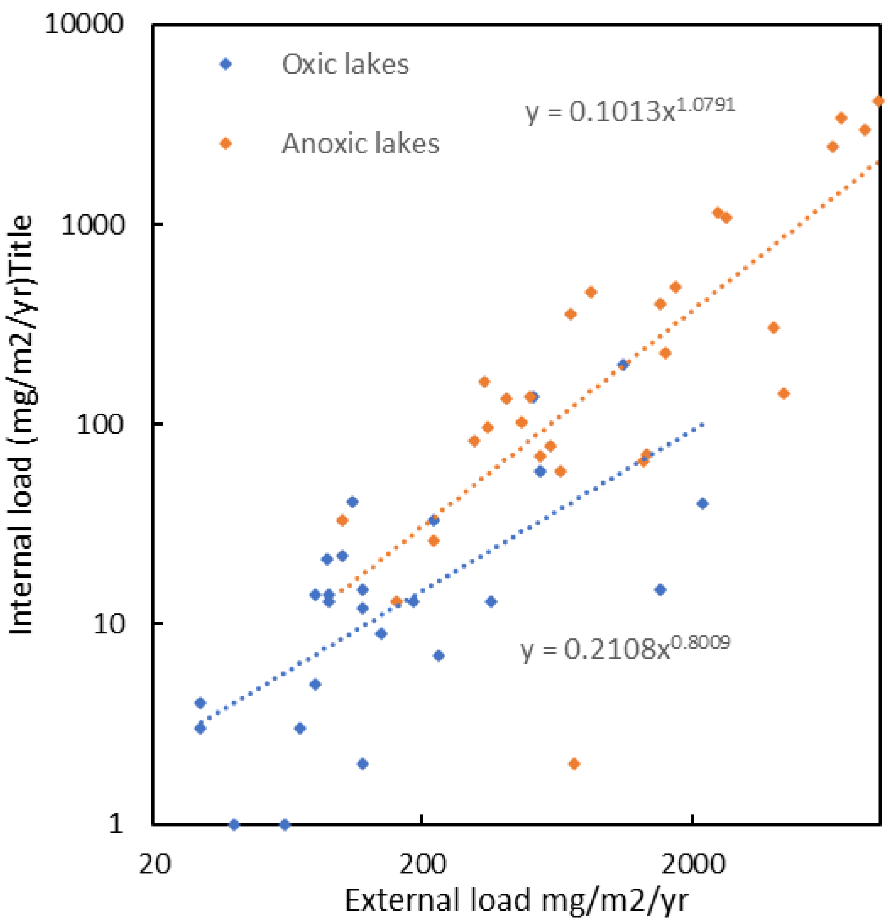

Figure 11. The dependence of internal P load on external P load. Data from Nurnberg (1984). To create this log-log plot all sites with 0 or negative apparent internal P load are excluded. Thus, the plot represents sites with relatively high internal loads, and may be regarded as an upper limiting rate

From these studies it is clear that internal P load varies primarily as a function of external load as Figure 11 shows the strong association of P internal load with P external load. We argue, that to a first order, there is no justification for distinguishing oxic and anoxic lakes. Removing three outliers to better capture the general situation, the Figure 12 represents the best we know of the relationship. The data are somewhat scattered, but then theory says the true relationship depends on the history of pollution (Boyle et al. 2023). Though some of these sites will be more or less in steady state, others will not be, so considerable scatter is expected. That the oxic and anoxic lakes fall on the same trend argues against a dominant role for lake oxygenation as a regulator of internal loading.

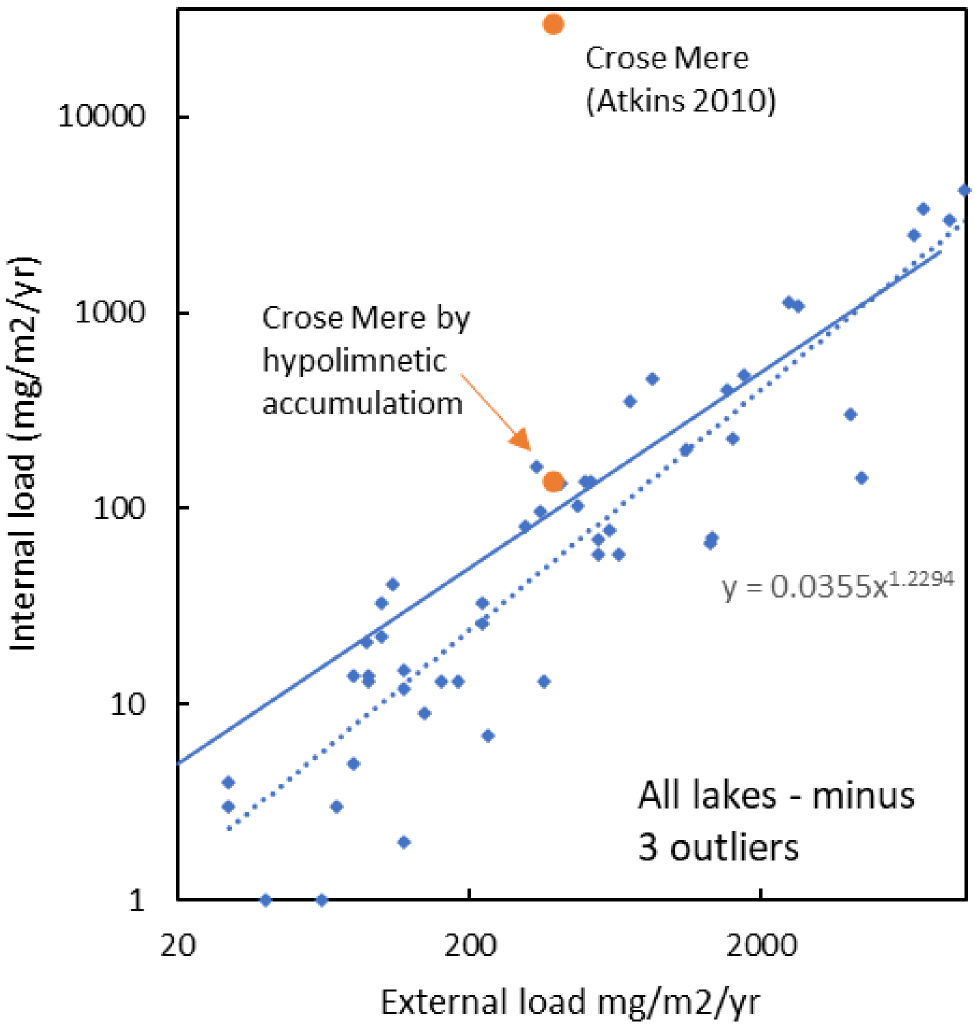

Figure 12. A simple empirical model based on the data of Nurnberg 1984 using only sites with positive internal loads. Note the log-log scale. Dashed line is the line of best fit (power), and the solid line represents $L_{internal} = 0.25 \times L_{external}$

Also shown on the plot are the LESA-NP (Atkins 2010) estimate of internal loading at Crose Mere and the Moyle (2021) estimate based on hypolimnetic accumulation. We can see that the value produced by LESA-NP is far in excess of any calculated value and therefore cannot be considered representative of a realistic internal load for Crose Mere.

5) Modelling

Boyle et al. 2023 propose a method that, if the history is known or can be reasonably estimated, it can be used to predict both the magnitude and future temporal variations in the internal load. This is not yet currently available. An alternative approach is to use an empirical model, and we propose using the line of best fit from Figure 12 ($L_{internal} = 0.0355 \times (L_{external})^{1.2294}$), as this represents our most complete current list of estimated internal P loads.

5.11.2 Application of the empirical prediction of internal P load

The various internal P load estimates described above are compared with the empirical model in Figure 12. For Crose Mere, the empirical model yields a lower estimate than does the hypolimnetic accumulation method, and both are very much lower than the LESA-NP estimate of Atkins (2010). As outlined above, both methods are expected to overestimate the true internal load, but in the absence of alternative approaches we propose that the empirical model yields a justifiable upper limiting estimate of 7.5 kg/yr, equal to 14% of the current external load.

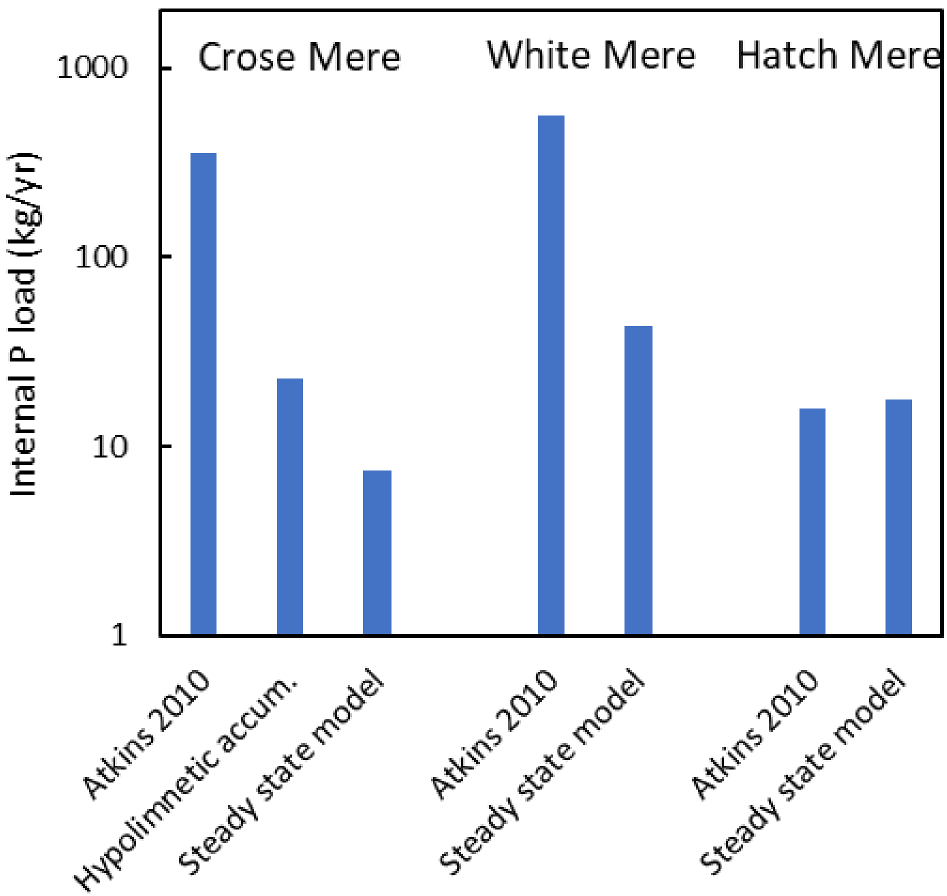

Figure 13. A comparison of internal P load estimates for the three lakes. Steady state model refers to the empirical model from Figure 12. Note log scale

At Hatch Mere, the Atkins (2010) approach, using their 5 mg/m^2/day for shallow lakes, gives a result (16 kg/yr) very close to the empirical model (18 kg/yr). We regard the similarity in values to be essentially coincidental. We do, however, have one additional source of information about internal loading at Hatch Mere, as Phoslock was applied in mid March 2013 which should have greatly reduced the internal load. Unfortunately, there were only 9 monthly measurements of lake TP prior to application, weakening the power of the tested change. In the 9 months prior to application the mean TP was 83 µg/L, while over the 25 months following it was 71 µg/L. Though not a statistically significant fall, this 17% reduction in TP is consistent with the empirical model which predicts a 21% internal load contribution. However, one of the monthly measured values prior to Phoslock application is an extreme outlier, typical of algal bloom impacts at Crose Mere (Moyle 2021), and may be biasing the results. If disregarded, the fall in TP, still not statistically significant, is just 6%. It is reasonable to suppose that the true internal P load at Hatch Mere is no more than 18 kg/yr, and is likely to be rather lower.

At White Mere the Atkins (2010) estimate of internal load (554 kg/yr) is greater than the empirical model estimate (44 kg/yr), as at Crose Mere. However, these two lakes are functioning differently as we know White Mere is not currently in steady state (Figure 14), and thus the empirical model is underestimating the internal load. The steadily falling TP concentration (Figure 14) is consistent with a past high external load from which the lake is currently recovering. Without a reliable estimate of the external load there is no way of quantifying the internal load, nor accurately predicting the recovery time.

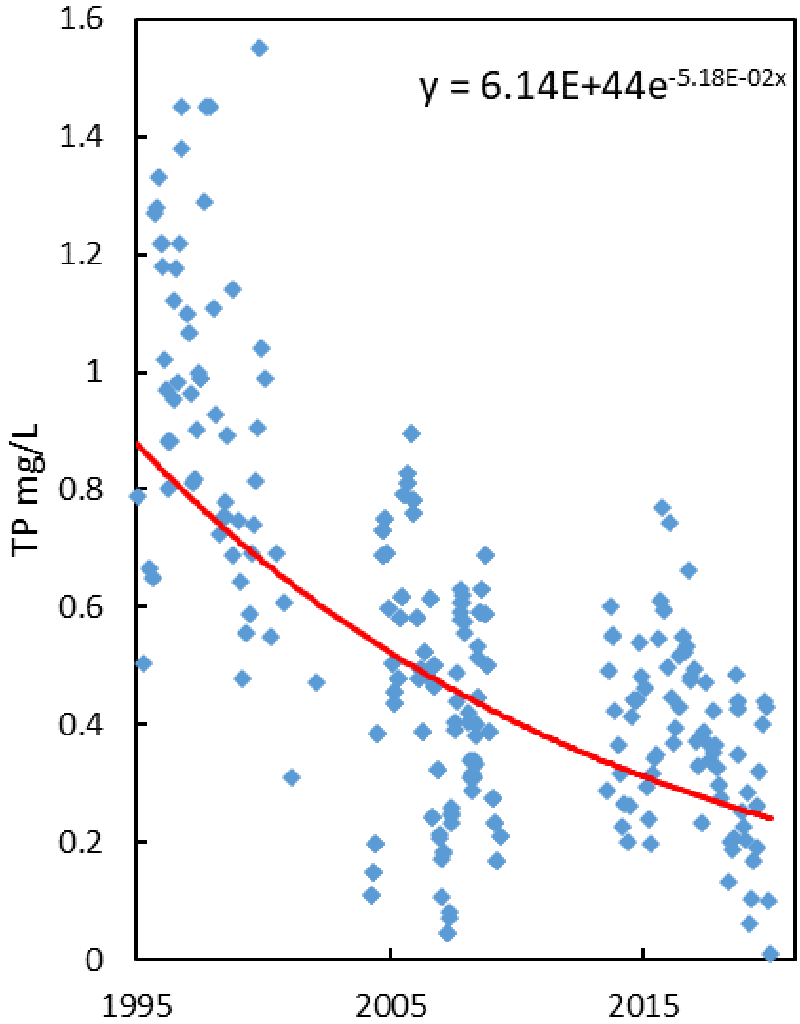

Figure 14. Environment Agency TP monitoring data from White Mere

5.11.3 Summary

Internal loading requires appropriate handing in lake P budget calculations, and has previously been overestimated for the Meres. Simple generalisation from incubation experiments is not a reliable method for quantification and has contributed to previous inaccurate estimates. For lakes approximately in steady state, the empirical model based on data of Nurnberg (1984) offers a reasonable approach to approximating internal loading. However, at sites with high past external loading, for example White Mere, the steady state empirical model is invalid and more complex approaches are required (Boyle et al., 2003). The lack of well dated sediment records or long term hydrochemical monitoring data for White Mere means there is no solid basis for estimating internal loading. Such dated sediment records form a basis for estimating past external P loading, which when combined with a dynamic process model offers a basis for estimating current and future internal P loads. At present, there are few dated sediment records for the Meres and such a dynamic process model has not yet been developed.

As internal load can be considered a function of external load we suggest restoration efforts should concentrate on reducing external loads, as this will act to control the release of P from the sediment. After external P loads have been reduced, the contribution of the internal load to the overall lake budget will become greater, but will steadily decline over time. Where lakes have experienced extremely high past external loading (e.g. sewage supply) and the source of the high loading has been removed, internal loading will continue to contribute significantly to the P budget which will hinder initial recovery, however the rate of internal loading will eventually decline over time as the lake reaches steady state (Boyle et al. 2023).

5.12 Scenarios for potential future change

Scenario testing has the purpose of predicting the effectiveness of potential management interventions. All models are capable of generating such predictions, but the value of these depends on whether they are accurate. A rigorous analytical treatment of model prediction errors is not possible, for several different reasons. In principle, such a treatment might be possible for export coefficients, provided sufficient truly independent measured values were available for suitable temperate landscapes. In reality, however, this approach is ruled out by the scarcity of measurements. We are compelled to take the results at face value, and to encourage future efforts at increasing the amount of suitable data. For modelling output the situation is no better. While rigorous sensitivity testing is possible for models such as FARMSCOPER, this does not permit evaluation of output accuracy. Accuracy, instead, depends on the suitability of process representation within the model, and this is much harder to test. Model output can be tested against other independent measures, but this provides no information about which parts of the model deliver accurate P loads, and it is not even possible to distinguish erroneous data from unsuitable modelling. Consequently, we cannot comment on whether the FARMSCOPER P reduction scenarios are realistic.

Preclusion of analytical error analysis does not, however, rule out all assessment of prediction reliability. In particular, measured export coefficient values for forest in temperate landscapes are sufficiently lower than equivalent values for arable and improved grassland that arable reversion scenarios are relatively reliable. This observation is supported by lake sediment analysis, showing the pre Bronze Age landscapes to have P yields comparable to modern estimates of forest export coefficients (Moyle 2021). These arguments are used to assess prediction reliability for each scenario.

Five scenarios of land use change are described in Section 6.6, from which expected reductions in steady state catchment P export can be estimated using FARMSCOPER or other models. In each case, lake water TP concentration associated with the estimated load is calculated using the Vollenweider (1968) method as described in the Section 6.6.

Steady state conditions in the lake will not be reached immediately owing to three main factors that cause delays.

- First, landscape P export rates do not instantaneously respond to land use change. We currently have no empirical data or modelling approaches to predict this delay. Furthermore, implementation of land use change will take time.
- Second, there is a large reservoir of P in the lake water, marginal plants, and lake bed that will deplete in response to lowered external P supply, releasing P to the lake water in doing so, and thus damping the lake TP concentration response. The exact dynamics of this effect depend in reservoirs magnitudes of which only the lake water body store is known. We assume a first order rate constant of -1 /yr to represent this damping, an arbitrary value chosen to represent annual cycling.
- Third, in addition to short-term stores in the lake bed surface sediment, there is a large reservoir of P in the upper sediment layers that will deplete more slowly in response to lowered external P supply – the internal sediment P load (Section 7.4). We assume a first order rate constant of -0.1 /yr to represent this (Penn et al., 1995).

Applying this simple model to Crose Mere, we obtain predicted future TP records for the lake (Figure 15). In these simulated futures, the magnitude of the changes depends on the landscape P export models, and can be regarded as reliable. The rates of change, the rate at which the lake moves towards a final steady state TP concentration, are less certain. The slower internal sediment P loading is well studied such that the predicted rate in Figure 11 has a firm basis. The short-term rates are very much less certain owing partly to lack of specific information about the site, and partly to the scarcity of empirical information.

Scenario 1: FARMSCOPER optimized changes in arable practice to conserve nutrients and prevent pollution of water course (Figure 15) is assumed to lead to a 40% reduction in P load, taking TP from 68 to 51 µg/L after 5 years. There is no direct way of assessing whether these reductions are realistic.

Scenarios 2 & 3: Replacing half of the farmland by woodland produces slightly greater impacts, whether that’s 50 % replacement of arable land as calculated using FARMSCOPER (Scenario 2), or 50 % replacement of either arable or improved grassland calculated on the basis of the sediment inferred record (Scenario 3).

Scenario 4: Exclusion of cattle from the riparian zone is the least effective of the scenarios tested, but is almost as impactful as Scenario 1 leading to approximately 40 % reduction in TP, and can be most easily implemented. The reliability of this scenario is discussed below (Cattle exclusion).

Scenario 5: Full replacement of all farmland by forest, together with removal of all septic waste, which predicts an eventual steady state lake water TP of just 10 µg/L. The plausibility of this prediction is discussed further below (Afforestation and land cover change).

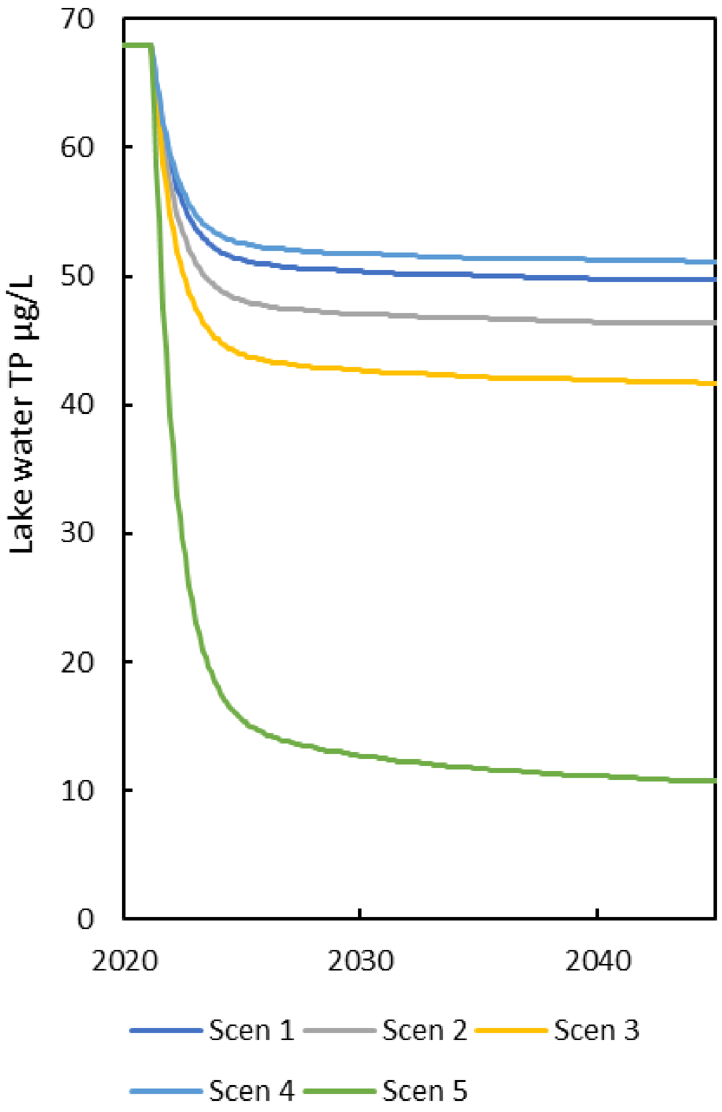

Figure 15. Inferred lake water TP change in response to five scenarios of catchment P export reduction. This simulates representative lags due to long term sediment stores and short-term stores sediment and marginal vegetation stores

This analysis leads to a number of general observations.

1. **Achievability of restoration targets**

All of the scenarios tested lead to a reduction in lake water TP concentrations at Crose Mere. However, four of the five scenarios do not reduce TP concentrations to below the target concentrations for the site. Only Scenario 5, total reforestation and removal of septic waste, brings TP to below 35 µg/L (CSM) or 36 µg/L (WFD GES) and these changes are sufficient to achieve a TP concentration of below the WFD High target of 26 µg/L.

2. **Afforestation and land cover change**

On the basis of the LESA-NP export coefficient, partial or complete forest replacement of agriculture is expected to lead to dramatic reductions in P export. This raises a series of questions relating to the reliability of the prediction, and of what sort of forest is required to achieve it? Indeed, would other forms of arable or grassland reversion achieve the same result?

The export coefficient used by LESA-NP is widely reported in comparable studies, and likely derives originally from eastern US hardwood forests (Hubbard Brook, Yanai, 1990). A review of export coefficient for forest (John Boyle, unpublished) found a mean of 4.7, range 2-5.6 $mg/m^2/yr$, for forests comparable those in the UK. Appling this mean value to Crose Mere still leaves very substantial benefit from replacement of farmland by forest. That this finding is realistic for Crose Mere is demonstrated by the longer sediment record, which shows early Holocene inferred lake water TP values of less than 5 µg/L (Moyle 2021). Despite modern enrichment of soils with P, there is no reasons to suppose that modern forests are functionally different, so similar benefits may be expected. How much benefit is offered by productive alternatives, such as arable reversion to scrub, agroforestry mixes, or short rotation crop willow, cannot be predicted with current models, though some empirical studies likely exist.

3. **Cattle exclusion**

Cattle exclusion is widely reported as an effective solution for reducing P pollution in streams and lakes, and yet is commonly not done. At Crose Mere, cattle are given access to the southern shoreline for the purpose of browsing alder and willow to prevent shading of the littoral zone. We estimate that 14 kg P/yr are delivered to the lake by excretion (Moyle 2021) either directly to the lake, or indirectly via the shoreline. Exclusion of cattle can be achieved simply by closing gates if alternative water supply is already available. The only barrier to exclusion then is finding an alternative approach to managing the growth of alder and willow saplings. A potential solution is to restrict cattle access to shorter time periods, giving pro rata reduction in direct P contributions. Such an approach would require careful monitoring of alder and willow regeneration.

4. **The role of internal loading**

The TP trajectories in Figure 15 show only a relatively small impact of long term internal phosphorus loading, the effect being greatest for scenarios that have the largest impact on total phosphorus supply. The weak damping effect at Crose Mere is due to the relatively low value for internal loading at this site. At White Mere, the very much greater internal P loading leads to a very strong lag as can be seen in Figure 14.

5. The role of carp

While carp have been shown to increase the turbidity in lakes, and to impact the balance between macrophytes and phytoplankton, their impact on lake water TP is less certain. Most studies have been of short duration and spatially restricted, leaving long-term, whole-lake changes unmeasured. The two best studied lakes are in Minnesota, revealing conflicting results. At Lake Susan, a small hypertrophic lake, Bajer and Sorensen (2015) found carp removal to cause a substantial reduction in total suspended solids, but no effect on lake water TP. In contrast, carp removal from Pickerel Lake led to reduced lake water TP. At both lakes, TP data are from only the summer months, and therefore the annual impact remains highly uncertain. We can conclude that removal of carp would reduce turbidity, but it is not possible to predict whether lake water TP would change.

5.12.1 Summary

The scenarios show best estimates of potential future lake water TP trajectories in response to possible catchment P mitigation measures at Crose Mere. All scenarios yield substantial reductions, though only complete replacement of farmland by forest (possibly also related arable reversion) is sufficient to achieve CSM and WFD targets. Internal loading delays full benefits from mitigation measures, but not to a great extent at this site.

6 Priorities for restoration of lakes

We make three classes of recommendation based on the identified evidence gaps and the scenarios: targeted **data gathering** to improve the reliability of interventions; **model development** to allow prediction of lake response to changing land use and climate; and **action** to implement target P reduction at specific sites. We regard all the recommendations as important, and suggest setting priorities on the basis of both importance and achievability, the latter combining cost and time frame.

For each recommendation class, we propose the priorities for restoration of lakes to be:

6.1 Data gathering

A priority is the measurement of lake P budgets at all Natural England priority sites. This requires hydrochemical monitoring of the lake and catchment and the collection of well-dated and well-measured sediment records of historic lake water TP. This is partially underway, but expansion to other sites is important. Paleo data is essential for understanding past lake response to changes in the catchment and the identification of true natural P baselines.

6.2 Model development

A priority to understanding these sites is improved modelling of lake response to changes in catchment land use and climate. LESA-NP is not a dynamic model and therefore cannot model this change through time. There are two steps to creating a suitable dynamic lake response model. First, creation of a comprehensive data base on lake internal loading, both dynamics and snapshot magnitudes. Second, coding, parameterisation and validation of a dynamic lake P model combining the approaches of Moyle (2021), Moyle and Boyle (2021) and Boyle et al. (2023). Another model priority is to test the current handling of cattle by FARMSCOPER as we find it is likely to be underestimating the contribution of cattle in agricultural P exports.

6.3 Action

At Crose Mere, the most easily implemented action to reduce lake water TP is cattle exclusion from the lake edge. However, excluding cattle removes the current management strategy for suppression of alder and willow growth on the south shore and therefore an alternative approach to supressing tree growth would need to be implemented alongside this.

Of the tested scenarios, the strategy that produces the largest reduction in lake water TP at Crose Mere is the replacement of farmed land with woodland. This is the only scenario that reduces TP concentrations to below the CSM and WFD target values. Implementation of this strategy requires careful consideration of ELM opportunities (Creation and management of a riparian strip; development of a Woodland Creation Plan; capital grants for woodland creation and maintenance).

7 Evidence gaps

The results and discussion highlight a number of areas where lack of information is preventing us from reliably predicting the consequences of management decisions on the TP concentration in lakes. Specifically, pathways to achieving TP targets depend upon suitable models and data that are both reliable and comprehensive. Here, we list the most important evidence gaps and suggest approaches to filling them.

7.1 Quantifying internal P load

Internal P load, the leaking of previously deposited P from sediments back to the water column, can be a major source of P enrichment in lakes. Where this is the case, the practitioner making choices about nutrient management needs to know the magnitude of the internal load, and the timeframe over which it will operate. The optimal method for achieving this would be monitoring of the vertical lake TP profile to quantify hypolimnetic P, coupled with a well-measured and well-dated sediment record of past lake water TP. This method is expensive and time consuming and may therefore not be a viable option, particularly as it would need implementing at all sites of interest. An alternative approach would be the development of an appropriate process model, tested at a small number of well-quantified sites. The simple empirical model outlined in this report offers a way or predicting internal load magnitude where lakes are in steady state, but this does not work where lakes are recovering from past high but undocumented episodes of external loading. For this a sophisticated dynamic model is required.

Solution:

Lake sediment records provide a means of determining the current and historic external P loading history. This information allows accurate prediction of current internal loading, and provides data essential to estimation of its future progression. To achieve this estimation a dynamic model of lake P capture and rerelease is required. The principles to doing this exist (Boyle et al., 2023) but model development and gathering of suitable empirical data for model validation are required.

7.2 Suitable data at small catchment scale

The analyses undertaken in this project demonstrate that both LESA-NP and FARMSCOPER display considerable skill in predicting P loads to lakes, provided they are supplied with accurate data. While we conclude that some refinement of both models would be desirable in the case at least of lake catchments, it is sourcing of suitable data that is the primary barrier to application of the models.

We find that reliable quantification of the number of septic tanks, average populations of geese, and average stocking densities are particularly important, yet hard to come by for small areas. For implementation of FARMSCOPER, it is also necessary to have independent information about the nature of soil drainage and the state of repair of tile drains. Existing maps of soil hydrological properties are useful but are on too course a scale to be directly applied.

Solution:

There are no short-term fixes to the absence of mapped data needed to drive models of diffuse P exports, and the solution here is to collect site-specific data for all sites of interest. Alternatively, lake sediment records can be used to accurately quantify the dynamics of the P budget. While this does not compensate for the lack driving data, it does permit critical independent evaluation of whether the model is delivering sufficiently accurate information.

7.3 Identifying hydrological catchment area

Knowing the location of the catchment boundary is essential for determining which P sources contribute to the nutrient budget of a specific water body. There is no practicable way of determining this from topographical information alone in a region where substantial amounts of water can flow through the subsurface. However, if the water budget of a lake can be quantified then we can at least know catchment magnitude, and this can constrain options for where the hydrological catchment boundary lies. Without this information there will always be uncertainty about how much P is supplied and from where, and this severely compromises nutrient loading management.

Solution:

There are two sources of information which can reduce the uncertainty in the water budget. First, where outflow streams are present, the discharge can be monitored on a regular basis, providing fully reliable evidence for the minimum size to the hydrological catchment. Second, passive or active chemical tracers can be used to determine the water budget. Passive tracers work by tracking change through the year, placing constraints on the water residence time in the lake. In practice, very long records are required and the results are relatively imprecise. Active tracing involves the addition of a harmless chemical substance. This substance needs to be conservative, meaning that it does not break down through time or become absorbed to materials within the lake. Various dyes have been employed for this purpose, but are not sufficiently stable for long residence time lakes owing to photodegradation. NaCl is a suitable substance, its loss from a lake reflecting only the water throughput. This would require doubling the natural concentrations which we would expect to have relatively little impact on biota. However, very large quantities will be required to achieve a concentration doubling, at considerable expense.

7.4 Identifying the past excessive P loads at White Mere

The very high lake water TP concentration at White Mere is not consistent with the small catchment and relatively low-intensity farming. The high but falling TP concentration is consistent with high internal P load, evidencing a very substantial past P source that is currently unknown. At present, management of the lake P budget at White Mere must assume that this P source is now inactive, an assumption that is unsafe if we cannot identify the source. Potential candidates include past hemp retting; former temporary settlements (for example, a Second Word War encampment); leakage from the aggregate quarry.

Solution:

A detailed lake sediment analysis based on a robust ^{210}Pb chronology would reveal the total P loading history, tightly constraining the timing of past elevated P loadings. This information would allow us to discriminate between potential past drivers of high external P loading. It would also allow us to confidently predict the future trajectory of TP concentrations at this site.

8 Recommendations

We reiterate a statement made in the Atkins (2012) hydrology report that while modelling is essential, it is no substitute for empirical data collection. In the evidence gaps section, we list a number of classes of data that limit our capability to predict the P dynamics of our case study lakes.

Here we propose some targeted data gathering that will enhance the value of existing information. Our proposals fall into three types: data gathering, model development, and action:

8.1 Data gathering

8.1.1 Hydrological data

- To improve the quantification of P budgets at the meres using hydrochemical balances, we recommend that lake inflow and outflow stream flows should be monitored as the best-case scenario for quantifying P budgets at the Meres
- At a minimum, we recommend the inflow and outflow streams at Crose Mere and Hatch Mere should be monitored. These sites are EA monitoring sites and without flow data, the monitored TP values cannot be reliably used to calculate P budgets. This would be of still greater value if the EA sampling site could be moved to the outflow stream at all monitored sites, as this would reduce the incidence of unrepresentative local enrichment due windblown algal blooms.
- Subsurface flows at the meres are currently unknown due to a lack of any empirical quantification. We recommend that options for using an active tracer to quantify total flows should be considered, and this should be done at a site where surface flows are also measured. Again, we recommend Crose Mere and Hatch Mere as test cases for this. At Hatch Mere, the water residence time is likely to be short enough that rhodamine B dye tracing might be applied.
- Following quantification of both surface and subsurface flows, we recommend that an updated hydrological model is produced and tested using nearby NRFA flow data and gauged rainfall. This would enable better hydrological modelling at the unmonitored sites.
- Refined characterisation of the hydrological catchment boundaries should be mapped, taking topographical information, borehole data, and lake water budgets into account. This would allow more reliable application of catchment P budget models such as FARMSCOPER.

8.1.2 P source data

- This report has shown the potential for successful application of FARMSCOPER and LESA-NP at the case study sites. The application of the models at these sites was based on the best available data which included downscaled regional data. For this proof-of-concept study, this was sufficient to test the viability of this coupled modelling approach and achieve the study aims. For a full quantification of the P budgets at the meres, we recommend that site specific catchment scale data is collected.
- For historical projection of P budgets for the meres, site-specific past land cover and land use data is required at catchment scale. For White Mere, we have identified an issue with high historic P loading. A historical land use survey and hydrological budget would enable the nature and source of high past external P load to be properly understood at the site. We recommend historical data is collected for the meres.
- We have identified that internal loading is poorly quantified at the meres. We recommend that internal loading estimation by hypolimnetic P accumulation should be applied at each site, requiring monthly monitoring of water column TP concentration profiles at the deepest point in the lake.

8.1.3 Palaeo data

- Validation of the FARMSCOPER and LESA-NP outputs was only possible by direct comparison with the combined monitoring and sediment data. For this proof-of-concept study, we have shown that the coupled model approach can be successful. However, in the case of White Mere we show that further palaeo data is needed to understand historic and present P budget at the site. We recommend that well dated and well measured sediment records are collected from across a wider range of lakes to enable past and current P budgets to be reliably quantified.

8.1.4 Export Coefficients

- The export coefficients used by LESA-NP are those reported by Johnes (1996) and White and Hammond (2009), and are based on a very restricted set of measurements precluding statistical treatment of errors, or reliable assessment of regional variation. This scarcity of export coefficient measurements is not wholly for lack of suitable data; a vast network of stream and lake TP measurements are made by Environment Agency, Scottish Environment Protection Agency, and Natural Resources Wales, which when combined with the equally vast network of measured discharges (National River Flow Archive) permits calculation of annual P export loads across all of Britain. Few of the rivers drain the single landscape types ideal for estimating export coefficients, and many would need correction for treated waste water contributions. Nevertheless, statistical procedures exist for separating the signals, and the principal barrier is the wholly separate collection of water quality and discharge information. We recommend that Natural England consult with other agencies with a view exploring options for unlocking this resource.

8.2 Model development

- Our testing of LESA-NP highlighted that internal loading is not adequately handled within the model. We recommend that internal loading is revisited in this model.
- Our testing of FARMSCOPER suggests it is likely underestimating the contribution of cattle in agricultural P exports. We recommend that cattle are revisited in this model.
- We have shown that FARMSCOPER and LESA-NP can be used to quantify present-day P budgets at lake sites. In this study, scenario testing required the development of a basic dynamic model that used the output of LESA-NP to define the initial model conditions. LESA-NP is not set up to model change over time in response to changing drivers like land use and evolving internal loading or climate. To fully understand how these lake systems will respond to these changes, we recommend that a dynamic lake model is developed to predict current lake TP and future trajectories in the face of changing land use and climate change.

8.3 Action

- Our testing of FARMSCOPER and LESA-NP at Crose Mere has shown a number of potential management strategies that could be implemented to reduce lake water TP concentrations at the site. We recommend that these management strategies are considered and implemented to improve water quality at the site. In this instance we recommend that the hydrochemistry at the site is monitored to evaluate the lake response to these management strategies, enabling better informed future water quality management at the meres.

9 Detailed methods and data

9.1 Crose Mere topographic catchment

A polygon shapefile of the Crose Mere catchment was created manually in ArcMAP, guided by ArcHydro flow accumulation maps based on the 2 m LIDAR DTM (DIGIMAP DOWNLOAD). The topographical catchment is well-defined (Figure 1), but the hydrological balance of the lake is uncertain because we find a mismatch between the measured outflow and the topographic catchment (Moyle 2021). There are two limiting assumptions. 1) If we assume that all outflowing water leaves the mere via its surface out flow, then the true hydrological catchment is smaller than the observed hydrological catchment. 2) If we assume that the hydrological catchment is the same as the topographical catchment, then then some outflowing water leaves the lake via subsurface pathways (This is known to happened for some meres, where there is no surface outflow at all). This leads to two different estimates of the total P budget because of the differing estimates for the outflow P load. The estimates are described in the Section 6.

9.2 Crose Mere catchment land use data

9.2.1 Land cover

CEH Land cover maps are available via DIGIMAP Download as vector shapefiles for 1990, 2000, 2007, 2015, and then 2017 to 2021. A non-georectified image is available for 1935 (The Dudley-Stamp survey) in DIGIMAP ROAM (map viewer). The vector files were clipped to the catchment boundary, and areas for each land use polygon were found using the $Area function in QGIS. These areas were summed for each land cover class to give the values in Table 10. The 1935 land cover image showed the same polygons as the vector land cover shapefile, allowing land cover class to be manually assigned. All LCM maps except LCM2000 incorrectly assigned the boggy grassland adjacent to the south eastern corner of the mere to the land cover class “Freshwater”, which explains why most of the LCM maps overestimate the lake area (0.154 km^2). The LCM2000 map allocates emergent phragmites as not-freshwater, hence the underestimated lake area. Finally, LCM2019 over represents deciduous woodland. Table 11 shows data corrected for these issues. It also shows values interpolated between 1935 and 1990. It also shows some average values that represent typical recent conditions.

Table 10. Land cover areas (km^2) based on CEH Land Cover Maps

Year	1935	1990	2000	2007	2015	2017	2018	2019	2020	2021
Deciduous woodland	0.032	0.011	0.021	0.01	0.011	0.011	0.011	0.019	0.011	0.011
Arable	1.038	0.695	0.791	0.92	0.857	0.881	0.91	0.827	0.808	0.87
Improved grass	0.396	0.846	0.751	0.62	0.673	0.659	0.63	0.705	0.722	0.666
Freshwater	0.166	0.161	0.137	0.151	0.172	0.162	0.161	0.161	0.167	0.166
Suburban	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007
total	1.639	1.72	1.707	1.708	1.72	1.72	1.719	1.719	1.715	1.72

Table 11. Corrected data (km^2): lake fixed at 0.154, with excess attributed to rough grass

Year	1935	1990	2000	2007	2015	2017	2018	2019	2020	2021	Mean 2015-18	Mean 2018-21
Deciduous woodland	0.032	0.011	0.021	0.01	0.011	0.011	0.011	0.011	0.011	0.011	0.011	0.011
Arable	1.038	0.695	0.791	0.92	0.857	0.881	0.91	0.827	0.808	0.87	0.883	0.835
Improved grassland	0.396	0.846	0.751	0.62	0.673	0.659	0.63	0.705	0.722	0.666	0.654	0.698
Rough grassland	0.012	0.007	0.007	0.007	0.018	0.008	0.007	0.007	0.013	0.012	0.011	0.011
Freshwater	0.154	0.154	0.154	0.154	0.154	0.154	0.154	0.154	0.154	0.154	0.154	0.154
Suburban	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007
Total	1.639	1.72	1.731	1.718	1.72	1.72	1.719	1.711	1.715	1.72	1.72	1.716

9.2.2 CEH crop maps

CEH Land Cover plus crop maps (Figure 2) are available via DIGIMAP download as shapefiles for 2016 and 2017. The crop maps are also available for 2018-2021, but only as images shown in the DIGIMAP ROAM map viewer. The vector files were clipped to the catchment boundary, and areas for each land use polygon were found using the $Area function in QGIS. These areas were summed for each land cover class to give the values in Table 12. The 2018-21 images showed the same field boundaries as the vector maps, allowing land cover class to be manually assigned. The LCM+ maps showed polygons for all agricultural land cover, including small patches of woodland, and left built areas and gardens blank.

Table 12. CEH Land Cover Maps Plus crop data, 2015-2021, in km^2

	2016	2017	2018	2019	2020	2021	Mean
Grass	0.723	0.632	0.577	0.591	0.591	0.591	0.618
Maize	0.269	0.12	0.082	0.175	0.153	0.184	0.164
Potatoes	0.102	0	0.166	0.079	0	0	0.058
Spring barley	0.03	0.03	0	0	0.117	0.091	0.045
Winter barley	0.236	0.05	0.149	0.129	0.311	0.217	0.182
Winter wheat	0.095	0.622	0.39	0.354	0.211	0.291	0.327
Oil seed rape	0	0	0	0.128	0.03	0.009	0.042
Winter oats	0	0	0	0	0	0.002	0.001
Spring wheat	0	0	0.091	0	0.009	0	0.05
Other	0	0	0	0	0.03	0.07	0.05
Missing grass	0.073	0.073	0.073	0.073	0.073	0.073	0.073
Total grass	0.796	0.705	0.65	0.664	0.664	0.664	0.691
Total crops	0.732	0.822	0.878	0.865	0.861	0.864	0.837
Total	1.528	1.527	1.528	1.529	1.525	1.528	1.528

To apply crop data to the earlier time periods covered by the LCM maps (1990-2021), which show just total arable land and total grassland, the crop data in Table 12 was recalculated as percentages of the cropped land (Table 13). Not shown in the table is that individual fields clearly rotate crops such that a multi-year average is needed to obtain representative cropping.

Table 13. CEH Land Cover Maps Plus crop data, 2015-2021, crop type expressed as percentage (%) of total crop area

	2016	2017	2018	2019	2020	2021	Mean
Maize	36.7	14.6	9.3	20.2	17.8	21.3	20
Potatoes	13.9	0	18.9	9.1	0	0	7
Spring barley	4.1	3.6	0	0	13.6	10.5	5.3
Winter barley	32.2	6.1	17	14.9	36.1	25.1	21.9
Winter wheat	13	75.7	54.8	40.9	25.6	33.7	40.6
Oil seed rape	0	0	0	14.8	3.5	1	3.2
Winter oats	0	0	0	0	0	0.2	0
Other	0	0	0	0	3.5	8.1	1.9

9.2.3 County agricultural statistics

For time periods prior to 2016, crop information can be inferred from parish agricultural data, or from county data. The county data have been compiled in approximately decadal steps from 1905 forwards (Data file structure-england-june21-county-23jun22.ods, provided by Daisy Burris, NE), and this is what we use for historical land cover estimation. To apply these data to Crose Mere we must adjust the county level values. Table 14 shows the county crop data from 2016 and 2021, and compares it with our 2015-21 mean values. We assume that the proportional differences between county (mean of 2016 and 2021) and catchment land cover are maintained for the whole of the 20th century. The projected historical land cover values are presented in the section, Historical Land Cover.

Table 14. Comparison of CEH Land Cover Plus crop data for Crose Mere with county data, expressed as percentage (%) of total crop land

	LCM Plus 2015-2021	County 2016	County 2021
Maize	20	9.8	11.1

Potatoes	7	5.8	5
Spring barley	5.3	9.2	9.1
Winter barley	21.9	12.8	10.4
Winter wheat	40.6	41.8	44.3
Oil seed rape	3.2	11.2	9.9
Winter oats	0	5.2	6.9

Data for county level historical changes in crop type, and the relative proportions of arable versus improved grassland, are available at county level (Table 15). Note that the original data from 1925-1975 contains a crop category “Mixed corn”. Assuming this to comprise barley, wheat and oats (Maize being negligible before 1970), “mixed corn” has been distributed equally divided among them.

To apply these data to the Crose Mere catchment, scaling is required to adjust for the difference between local and county level proportions. Table 16 shows estimated historical crop types for Crose Mere, where values prior to 2016 are county data scaled to align the 2016-2021 values. While it is evident that Crose Mere will have land use proportions that are different to the county data, this approach yields a sensible approximation that can be used to test the magnitude of impact from changing crop types. Note that oats have been added to Crose Mere, at county levels, to allow for past production as horse feed. Other crops have been scaled down correspondingly to maintain 100% coverage.

As the historical stocking and crop-type data are expressed as proportions of arable or grassland areas, an estimate is required for these at Crose Mere. These proportions are at county level are shown in Table 17.

Table 15. Crop type (as percentage of cropped land) and livestock numbers (per hectare of grazed land), with "mixed corn" distributed equally across spring barely, wheat and oats

	2021	2016	2005	2000	1995	1985	1975	1965	1950	1945	1935	1925	1915	1905
Maize	11.4	10.3	6.5	4.8	5	0	0.3	0	0	0	0	0	0	0
Potatoes	5.1	6	5.3	6	7.5	6.7	5.4	6.3	10	11.3	6.6	6.5	3.5	5.4
Spring barley	9.4	9.7	10.9	10.9	11.7	56.4	65.4	56.1	22.1	26.4	12.2	25.6	28.5	35.6
Winter barley	10.8	13.3	13	21.6	26.3	0	0	0	0	0	0	0	0	0
Winter wheat	45.9	43.6	48.8	46.5	39.7	33.8	20.5	30.1	35.8	27.6	41	24	28.3	23.6
Oil seed rape	10.2	11.7	9.6	3.7	3.2	3.1	0.5	0.9	2.2	1.3	0	0	0	0
Winter oats	7.2	5.4	5.9	6.5	6.5	NA	7.8	6.5	29.9	33.3	40.4	43.9	39.6	35.2
Cattle	1.52	1.59	1.69	1.99	1.99	2.09	2.94	2.5	2.09	1.91	1.15	1.11	1.08	0.96
Sheep	4.82	5.01	5.36	6.67	6.49	4.96	4.91	4.83	2.61	2.47	2.39	2.25	2.47	2.45
Pigs	0.35	0.28	0.49	0.75	0.79	0.87	1.33	1.46	0.41	0.31	0.56	0.42	0.39	0.35
Poultry	37.0	23.7	39.5	39.9	0.0	22.4	19.5	18.2	13.1	0.2	NA	NA	NA	NA
Horses	NA	NA	NA	NA	NA	NA	0.03	NA	0.08	0.14	0.12	0.16	0.18	0.19

Table 16. Crop type (as percentage of cropped land) and livestock numbers (per hectare of grazed land), scaled to the 2016-2021 Crose Mere catchment values. Some data gaps filled by simple interpolation

	2021	2016	2005	2000	1995	1985	1975	1965	1950	1945	1935	1925	1915	1905
Maize	21.9	19.3	12.8	8.5	8.4	0	0.9	0	0	0	0	0	0	0
Potatoes	6.9	7.9	7.4	7.5	8.9	12.2	11.2	12.5	15.7	17.7	9.7	10	5.6	9
Spring barley	5.3	5.4	6.3	5.7	5.8	43	56.2	46.3	14.5	17.2	7.5	16.5	19.1	24.5
Winter barley	20.8	24.9	25.4	38	44	0	0	0	0	0	0	0	0	0
Winter wheat	41.5	38.5	44.8	38.5	31.3	41.2	28.2	39.7	37.4	28.8	40.3	24.7	30.2	26
Oil seed rape	3.5	3.9	3.3	1.1	1	1.5	0.3	0.5	0.9	0.5	0	0	0	0.1
Winter oats	0	0	0	0.6	0.6	2.1	3.3	1	31.4	35.7	42.4	48.8	45.1	40.4
Cattle	0.71	0.75	0.79	0.94	0.94	0.98	1.38	1.18	0.98	0.9	0.54	0.52	0.51	0.45
Sheep	0.77	0.8	0.86	1.07	1.04	0.79	0.79	0.77	0.42	0.4	0.38	0.36	0.4	0.39
Pigs	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Poultry	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Horses	0	0	0	0	0	0	0.03	0	0.08	0.14	0.12	0.16	0.18	0.19

Table 17. Historical county data for percentage land cover for cropland, improved grass, and rough grass

	Arable	Improved grass	Rough grass
2021	40.5	58.1	1.4
2016	40.9	57.5	1.6
2005	37.4	60.2	2.4
2000	39.7	57.6	2.7
1995	37.3	59.6	3.0
1985	52.6	44.4	3.0
1975	50.5	44.6	5.0
1965	49.5	44.8	5.7
1950	46.0	46.0	8.1
1945	46.5	44.9	8.5
1935	25.0	67.7	7.3
1925	27.4	66.0	6.6
1915	30.9	69.1	NA
1905	34.0	66.0	NA

9.2.4 Livestock

The Atkins (2010) report simply notes that Livestock information was unavailable for Crose Mere. Moyle (2021) monitored cattle and sheep monthly from late 2016 to the end of 2018. Google Earth Pro can be used to count cattle, with sufficiently clear historical images available for Sep 2021, July 2021, June 2021, July 2020, June 2018, Dec 2010, Dec 2009, Dec 1999. The mean value across these images is used. Sheep can be seen in some images, but cannot be confidently counted. There is no information about pigs and poultry for the catchment, and these are assumed to be negligible. Values are shown in Table 18.

Table 18. Livestock data for Crose Mere

	Moyle (2021)2016-18 observation #	Atkins ##	Google Earth Pro	County data scaled to grassland ###		Assumed for model
				2016	***2021***	
Cattle	42	NA	48	127	101	45
Sheep	50	NA	NA	399	320	50
Pigs	NA	NA	NA	22	23	0
Poultry	NA	NA	NA	3360	4390	0

This did not include Kenwick

Atkins (2010) stated that the data were unavailable

Scaled to farmed area

The county data for 2016 and 2021 show broadly comparable expected numbers for cattle, but very much greater number for sheep, pigs and poultry. We assume these differences reflect spatial heterogeneity across the county. These data are shown in Table 17.

A summary of all historical land cover information for Crose Mere for use in FARMSCOPER and LESA-NP are shown in Table 19.

Table 19. Reconstructed historic land cover (ha), stocking (head), and population (count) for the Crose Mere catchment

	2021	2020	2019	2018	2017	2015	2007	2000	1990
Deciduous woodland	0.011	0.011	0.019	0.011	0.011	0.011	0.01	0.021	0.011
Arable	0.87	0.808	0.827	0.91	0.881	0.857	0.92	0.791	0.695
Permanent grassland	0.666	0.722	0.705	0.63	0.659	0.673	0.62	0.751	0.846
Rough grassland	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007
Freshwater	0.166	0.167	0.161	0.161	0.162	0.172	0.154	0.154	0.154
Suburban	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007
Maize	0.185	0.144	0.167	0.085	0.129	0.315	0.118	0.067	0.058
Potatoes	0	0	0.075	0.172	0	0.119	0.068	0.059	0.062
Spring barley	0.091	0.11	0	0	0.032	0.035	0.058	0.045	0.04
Winter barley	0.218	0.292	0.123	0.155	0.054	0.276	0.234	0.301	0.306
Winter wheat	0.293	0.207	0.338	0.499	0.667	0.111	0.412	0.305	0.218

	2021	2020	2019	2018	2017	2015	2007	2000	1990
Oil seed rape	0.009	0.028	0.122	0	0	0	0.03	0.009	0.007
Winter oats	0.002	0	0	0	0	0	0	0.005	0.004
Cattle	47	52	51	47	49	50	49	71	80
Sheep	51	56	56	50	53	54	53	80	88
Pigs	0	0	0	0	0	0	0	0	0
Poultry	0	0	0	0	0	0	0	0	0
Horses	0	0	0	0	0	0	0	0	0
People	12	12	12	12	12	12	12	12	12

Continued	1985	1975	1965	1950	1945	1935	1925	1915	1905
Deciduous woodland	0.011	0.011	0.011	0.011	0.011	0.011	0.011	0.011	0.011
Arable	0.925	0.908	0.899	0.861	0.875	0.491	0.531	0.557	0.609
Permanent grassland	0.616	0.633	0.642	0.68	0.666	1.05	1.01	0.984	0.932
Rough grassland	0.007	0.011	0.013	0.018	0.019	0.016	0.015	0.015	0.015
Freshwater	0.154	0.154	0.154	0.154	0.154	0.154	0.154	0.154	0.154
Suburban	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007	0.007
Maize	0	0.008	0	0	0	0	0	0	0
Potatoes	0.113	0.102	0.112	0.135	0.155	0.048	0.053	0.031	0.055
Spring barley	0.398	0.51	0.416	0.125	0.151	0.037	0.088	0.106	0.149
Winter barley	0	0	0	0	0	0	0	0	0
Winter wheat	0.381	0.256	0.357	0.322	0.252	0.198	0.131	0.168	0.158
Oil seed rape	0.014	0.003	0.004	0.008	0.004	0	0	0	0.001

Continued	1985	1975	1965	1950	1945	1935	1925	1915	1905
Winter oats	0.019	0.03	0.009	0.27	0.312	0.208	0.259	0.251	0.246
Cattle	60	87	76	67	60	57	53	50	42
Sheep	49	50	49	29	27	40	36	39	36
Pigs	0	0	0	0	0	0	0	0	0
Poultry	0	0	0	0	0	0	0	0	0
Horses	0	2	0	5	9	13	16	18	18
People	12	12	12	12	13	13	15	16	15

9.2.5 Wildfowl

Atkins (2010) cites information supplied by Natural England on number of geese, reporting a maximum of 668 and mean of 46. Moyle (2021), across the 2016-2018 survey, found an average of 26 geese, 40 black headed gulls, 0.8 cormorants, and 0.7 swans. Scaling by average mass, this gives 32 goose-equivalents. Averaging with the Atkins (2010) reported number gives a value of 39.

9.2.6 Population

Atkins (2010) reports that 18 households are present in the catchment to Crose Mere, Sweat Mere and Whatthall Moss. However, the majority of these are not in the Crose Mere catchment, the map showing only Kenwick Wood and Kenwick farm to be included. The Natural England 2022 Audit reports that “It is known that the septic tank overflow from Kenwick Wood, Kenwick Farm and cottages at Kenwick Farm (5 households in total) all enter the main arterial sub surface field drain that flows into Crose Mere”. From this we conclude that there are 5 households in the catchment. We further assume that these are occupied at national average rates (for 2021 the UK average household occupancy was 2.4). This gives a population for the catchment of 12 persons.

Historical population data are available Cockshut from 1891 (Vision of Britain; Figure 16). This shows a steady slight decline to the 1961 census. We assume this trend of variations applies at Crose Mere.

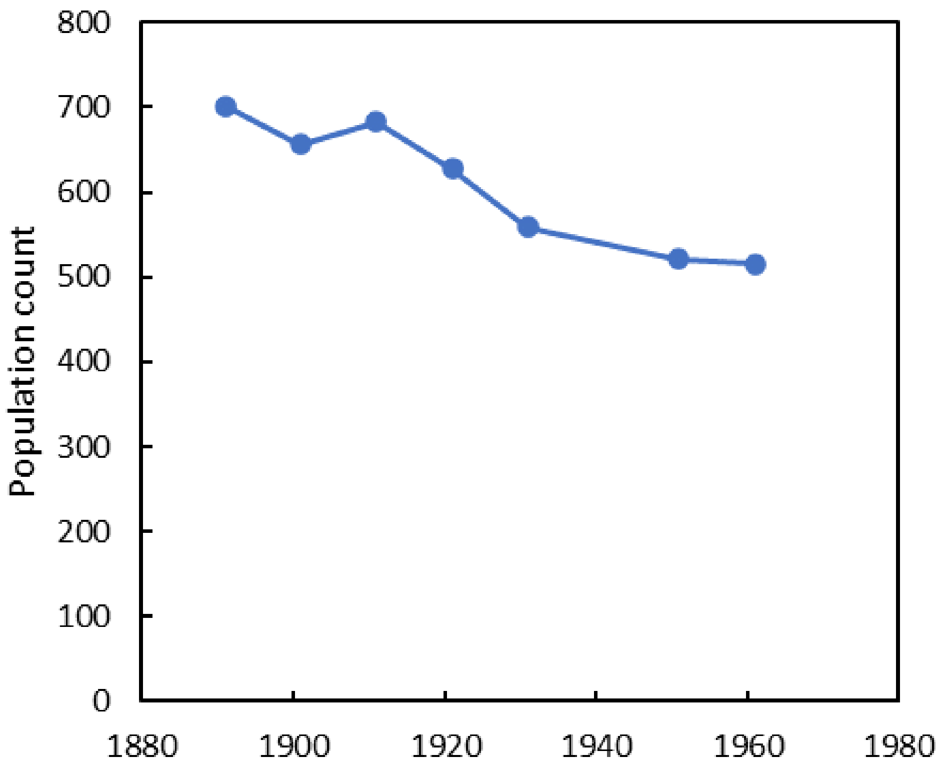

Figure 16. Population of Cockshut (1km south of Crose Mere)

9.2.7 Fertilizer application

Fertilizer application rates are assumed to be at national average rates (Table 20). These data are obtained from British Survey of Fertilizer Practice reports combined with historic total N and P application values (ktons/yr, Withers et al., 2014), scaled to overlapping application rates (kg/ha).

Table 20. Historic average UK fertilizer application rates (kg/ha). From 2010 the data is from the annual reports of the British Survey of Fertilizer Practice (DEFRA 2015). Earlier values are from Withers et al 2014), interpolated form some earlier dates where data are sparse

	P_2O_5	N		P_2O_5	N
2021	14	87	**1985**	40.9	143.9
2020	15	83	**1975**	36.7	90.9
2019	16	92	**1965**	44.7	52.1
2018	17	95	**1950**	44.7	17.4
2017	18.1	91.1	**1945**	33.9	15.9
2015	18.1	98.1	**1935**	16.8	5
2007	23.1	102.2	**1925**	19.1	4.3
2000	31.5	122.6	**1915**	17.5	2.8
1990	38.5	137.8	**1905**	15	2.3

9.3 White Mere and Hatch Mere catchments

9.3.1 Topographic catchment for White Mere

A polygon shapefile of the White Mere catchment was digitised using QGIS 3.22 based on 1 m LIDAR DTM data (DIGIMAP DOWNLOAD) (Figure 17). There is considerable uncertainty in the topographical catchment area as a hummocky region to the southeast of the lake is topographically isolated from both the lake and areas further east. We have assumed that this area drains into White Mere.

Based on this digitization, the catchment area at White Mere, including the lake, is 0.914 km^2. The lake area, based on Master Map topo, is 0.243 km^2. The lake area agrees very well with the UK Lakes Portal figure, but the catchment area automatically generate for the latter is incorrect, incorporating substantial areas of land that are at lower elevation that White Mere.

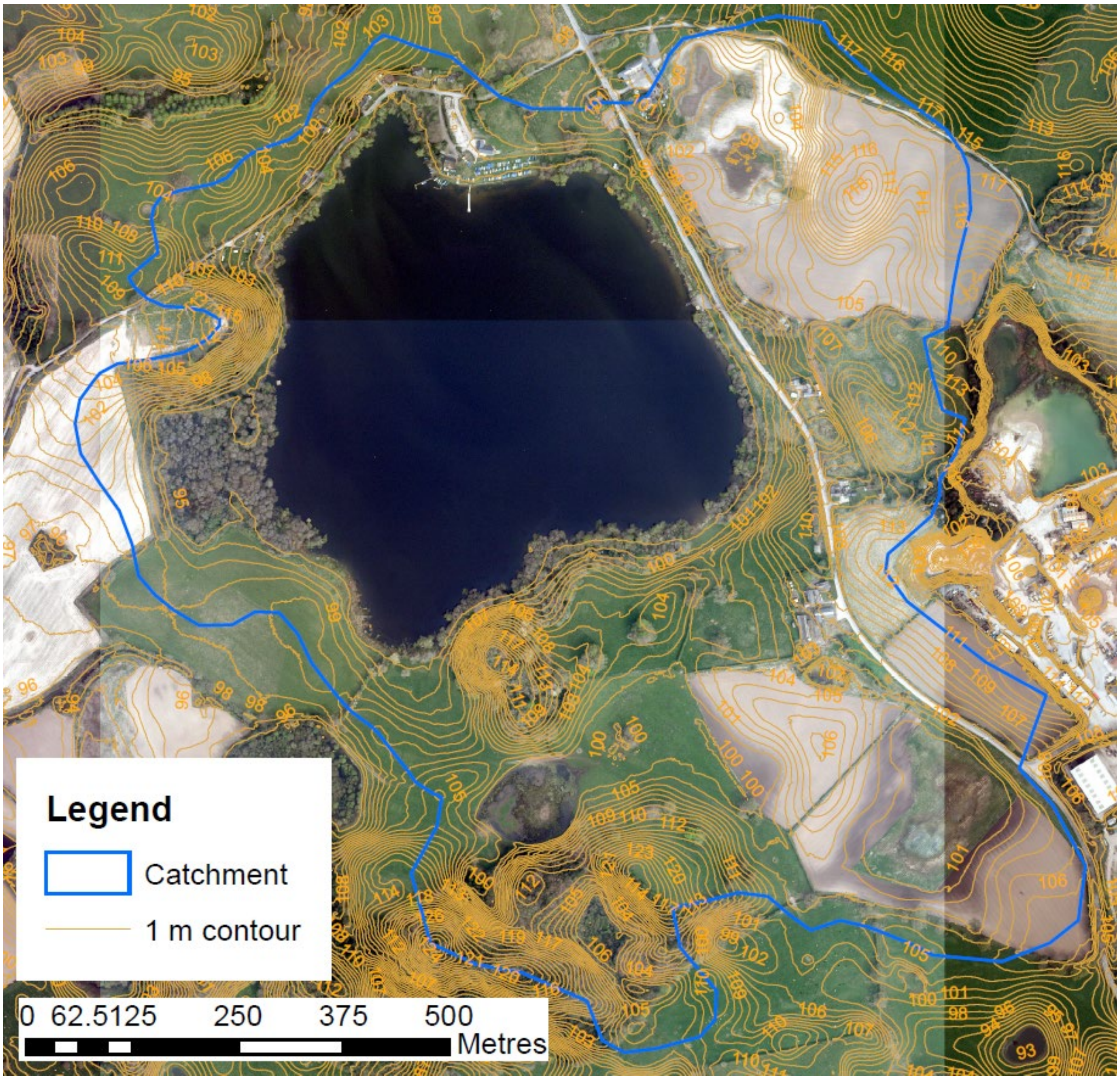

Figure 17. Topographical map of White Mere, with 1 m LIDAR-based contours superimposed on the 2012 Air photograph

9.3.2 Land use for White Mere

CEH Land cover maps have been downloaded from DIGIMAP as vector shapefiles for 2018, 2019, 2020, and 2022. The vector files were clipped to the catchment boundary, and areas for each land use polygon were found using the $Area function in QGIS. These areas were summed for each land cover class to give the values in Table 21. All LCM maps incorrectly assign some of the low-lying ground adject to the land cover class “Freshwater”, which explains why the LCM maps overestimate the lake area (0.243 km^2). This area has been allocated to Deciduous woodland.

Table 21. LCM landcover data for White Mere

	LCM code	LCM2021	LCM2020	LCM2019	LCM2018
		km^2	km^2	km^2	km^2
Deciduous woodland	1	0.105	0.105	0.105	0.105
Arable	3	0.191	0.191	0.212	0.140
Improved grass	4	0.336	0.336	0.324	0.396
Freshwater	14	0.270	0.270	0.270	0.270
Suburban	21	0.012	0.012	0.003	0.003

CEH Land Cover plus crop maps were downloaded from DIGIMAP as shapefiles for 2016 and 2017 (shown in Figure 18), the only data available (Table 22). The vector files were clipped to the catchment boundary, and areas for each land use polygon were found using the $Area function in QGIS. The crop maps contain many gaps – fields that are assigned to neither grassland nor a crop. We assume that the reported types are representative.

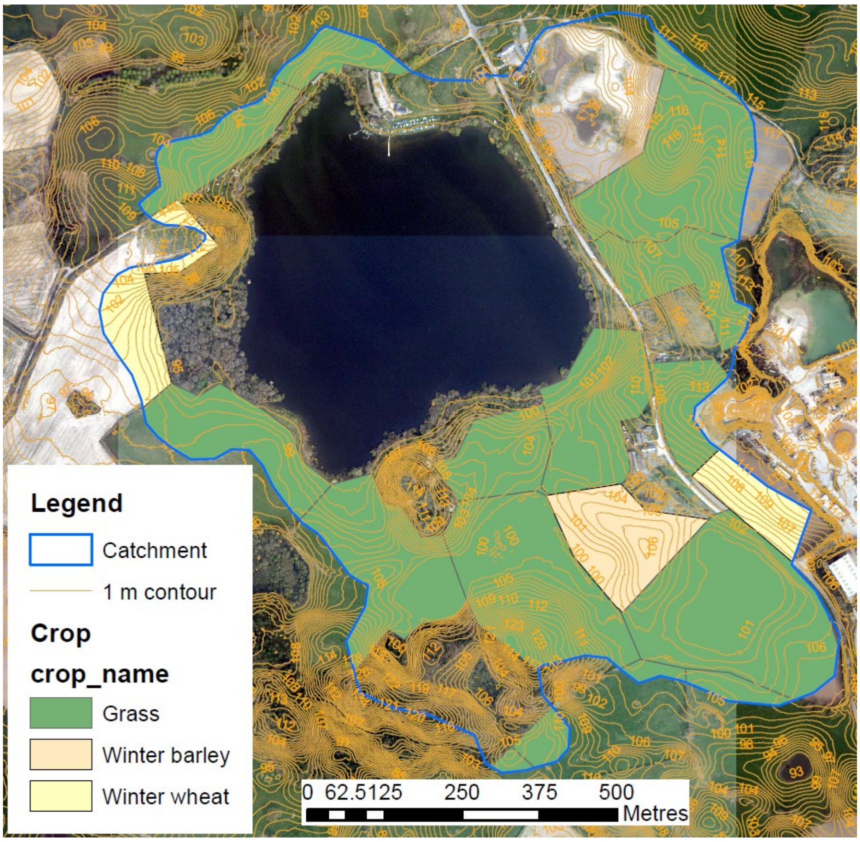

Figure 18. CEH Land Cover plus crop map, illustrated for 2016

Table 22. Land Cover Plus crop map data for White Mere

	2016	2017	Mean as % of cropland
	km^2	km^2	%
Grass	0.378	0.378	NA
Winter barley	0.027	0	22.5
Winter wheat (includes winter oats)	0.033	0	27.5
Spring barley	0	0.041	34.2
Oil seed rape	0	0.019	15.8
Sum of crops #	0.06	0.06	100
Sum	0.438	0.438	NA

Sum of crops disregarding grass

9.3.3 Stocking and population data for White Mere

Google Earth Pro shows the Hatch Mere catchment to contain 14 households, which at 2.6 person per household points to a total population of 36.

Eight snapshots of Google Earth Pro historical images spanning 1999-2022 shows an average of 11 cattle and 16 sheep.

9.3.4 Topographic catchment for Hatch Mere

A polygon shapefile of the Hatch Mere catchment was digitised using QGIS 3.22 based on 1 m LIDAR DTM data (DIGIMAP DOWNLOAD). The topographical catchment area is quite well-constrained (Figure 19). Based on this digitization, the catchment area, including the lake, is 2.221 km^2. The lake area, based on Master Map topo, is 0.0347 km^2. The lake area does not agree well with the UK Lakes Portal figure, which includes much reed swamp. The catchment area automatically generate for the latter is more similar, comprising 1.92 km^2.

9.3.5 Land use for Hatch Mere

CEH Land cover maps have been downloaded from DIGIMAP as vector shapefiles for 2018, 2019, 2020, and 2022. The vector files were clipped to the catchment boundary, and areas for each land use polygon were found using the $Area function in QGIS. These areas were summed for each land cover class to give the values in Table 23.

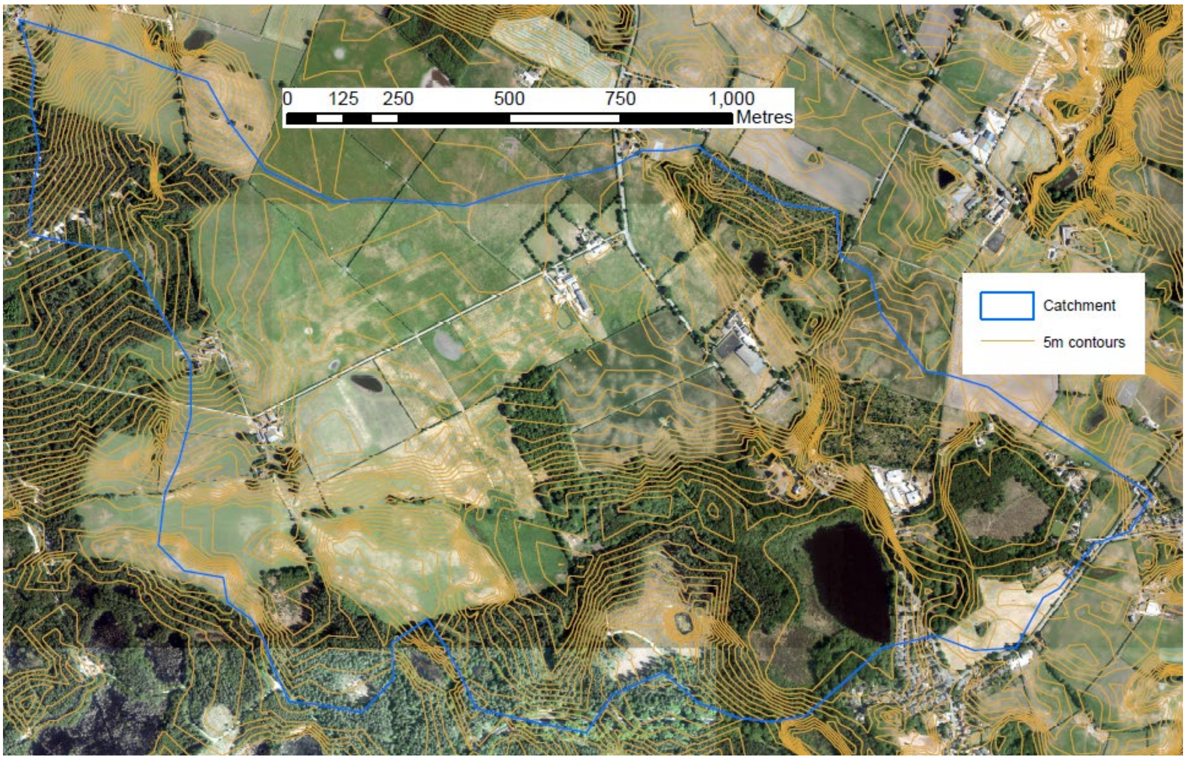

Figure 19. Topographical map of Hatch Mere, with 5 m DTM-based contours superimposed on the 2012 Air photograph

Table 23: LCM landcover data for Hatch Mere

	LCM code	LCM2021	LCM2020	LCM2019	LCM2018
		km^2	km^2	km^2	km^2
Deciduous woodland	1	0.808	0.810	0.722	0.711
Coniferous woodland	2	0.000	0.000	0.000	0.051
Arable	3	0.215	0.172	0.540	0.375
Improved grass	4	1.028	1.077	0.794	0.828
Neutral grassland	5	0.041	0.016	0.016	0.116
Freshwater	14	0.050	0.050	0.050	0.050
Urban	20	0.000	0.015	0.010	0.021
Suburban	21	0.080	0.081	0.088	0.069
Sum		2.221	2.221	2.221	2.221

CEH Land Cover plus crop maps were downloaded from DIGIMAP as shapefiles for 2016-2021 (see Figure 20), the only data available (Table 24). The vector files were clipped to the catchment boundary, and areas for each land use polygon were found using the $Area function in QGIS. The crop maps contain many gaps – fields that are assigned to neither grassland nor a crop. We assume that the reported types are representative.

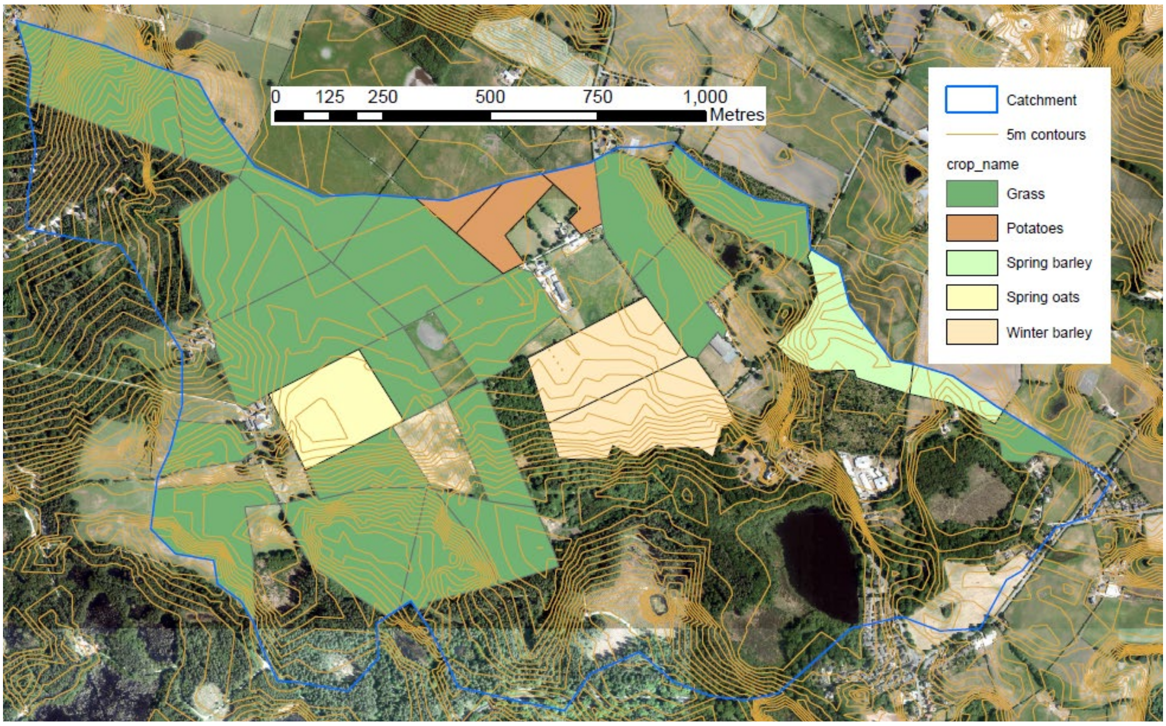

Figure 20. CEH Land Cover plus crop map for Hatch Mere, illustrated for 2021

Table 24. LCM plus crop map data for Hatch Mere

	2021	2020	2019	2018	2017	2016
	km^2	km^2	km^2	km^2	km^2	km^2
Grass	0.762	0.916	0.853	0.853	0.942	0.942
Potatoes	0.043	0.012	0.105	0	0	0
Spring barley	0.054	0	0.012	0	0	0
Oats	0.048	0	0	0	0	0
Winter barley	0.118	0	0	0	0	0
Spring wheat	0	0.042	0	0.042	0.042	0.042
Winter wheat	0	0.055	0.055	0	0	0

	2021	2020	2019	2018	2017	2016
Maize	0	0	0	0.063	0.029	0.029
Other crops	0	0	0	0.055	0.012	0.012
Sum of crops #	0.263	0.109	0.172	0.16	0.084	0.084
Sum	1.025	0.928	0.97	0.853	0.942	0.942

Sum of crops excluding grass

9.3.6 Stocking and population data for Hatch Mere

Google Earth Pro shows the Hatch Mere catchment to contain 82 households, which at 2.6 person per household points to a total population of 213.

Nine snapshots of Google Earth Pro historical images spanning 1999-2022 shows an average of 118 cattle and 52 sheep.

9.4 FARMSCOPER

In addition to site specific information about cropping and stocking, various additional site properties need to be specified in FARMSCOPER. The values and rationales are described here.

9.4.1 Control page

1. **Climate**: this was based on long-term data from the weather station at Shawbury (52.794°, -2.663°E), which has mean annual precipitation of 668 mm/yr for the period 1946-2022, and 707 mm/yr for 2016-2022 (https://www.metoffice.gov.uk/research/climate/maps-and-data/historic-station-data)
2. **Select soil**: Other>Drained for Arable use as selected as Arable is the dominant land cover class (83.5 ha arable, 69.8 ha improved grass).
3. **Select farm type**: Lowland Grazing was selected and then customised using data from Tables 8 and 9.
4. **Economics**: not selected

9.4.2 Farm page

- **Fields next to water courses**: based on the LCM plus crop maps, there are 37 fields included within the Crose Mere topographical catchment. Of these, 9 are field fragments of less than 1 ha. Of the 28 substantial fields, 5 border the lake, 2 of these also bordering the inflow stream. 1 additional field borders the inflow stream but not the lake. So, 6 of 28 (= 21%) are next to water courses. This answer is problematic, however, as no arable field border water courses.

- **Area of organic soils**: the DIGIMAP superficial deposit category "peat" comprises 0.13 km^2 of the Crose Mere catchment, or 8% of the terrestrial area (1.562 km^2)
- **Soil P indices**: The UK Soil Observatory interactive maps (https://mapapps2.bgs.ac.uk/ukso/home.html) gives Olsen P as the highest category (>40 mg/kg) for the 1 km^2 grid cells representing the Crose Mere catchment. For this reason we have scored the site 100% in the high category.
- **Connectivity**: We have no knowledge of the connectivity, and have left this with default values (52, 80, 90).
- **Field boundaries**: Based on a combination of field observation and Google Earth pro, approximately 85% of field boundaries are hedges, the remainder being fences (neglecting stream and lake edge boundaries).
- **Dirty water options**: We have no specific information on this. We have selected "Minimal dirty water collected and sent to dirty water store", as this is the default setting for Mixed combinable farms.
- **Farm type for estimation of method implementation**: We have no specific information. Lowland grazing defaults to "Extensive Grazing", while "Mixed combinable" defaults to "Other"
- **Grazing option**: We have selected "Livestock have access to watercourse whilst grazing", but this in only true for half the cattle.
- **Livestock**: We have selected 2 bulls, and split the remainder between "Beef Cows and Heifers" and "Other cattle (1-2 years)". All 50 sheep are categorised as "Sheep".
- **Fertilizers applied**: Values are taken from the various default farms, and so are broadly realistic. We rely on sensitivity testing to judge fertilizer application rate impacts. The values used for Crose Mere are shown in Table 25.

Table 25. FARMSCOPER crop specific fertilizer application rates used for Crose Mere

Cropping	N kg/ha	P_2O_5 kg/ha
Permanent pasture	47	14
Rotational grassland	90	25
Rough grazing	0	0
Winter wheat	154	19
Winter barley	116	48
Spring barley	95	37
Winter oil seed rape	199	17
Maize	20	19
Potatoes	157	53

9.4.3 Field operations

- These values have been left at their default settings.

9.4.4 Historical application

For past crop cover and stocking density we have used the values shown in Table 19. For historic fertilizer application rates, the values in Table 25 have been scaled to the total application rates shown in Table 20.

9.5 LESA-NP

LESA-NP is an export coefficient model of the type championed in Britain by Prof. Penny Johnes (Johnes, 1996). This is an empirical model that uses generalised landscape P export loading values measured at multiple study sites, termed “export coefficients”, and applies these to new sites that have no measured loadings. Expressed in area-normalised form (typically, either mg/m^2/yr, or kg/ha/yr), an export coefficient can be multiplied by the corresponding land cover area (m^2 or ha) to predict the annual P load. This approach neatly allows for varying catchment size, but doesn't allow for between-site differences in climate, soil type, drainage, etc.

LESA-NP model trials were performed on EXCEL worksheet Version 1 - v1_NE_LakeSA_Tool_LESA-NP_v3.xls, received 6th July 2022. The following stages of modification were implemented prior to testing.

9.5.1 Repair of broken links

As received, the worksheet had 12 broken links (values shown on the Front end that do not come from the data tabs, and thus do not change when the site is changed). These include:

- All items in the Catchment & Mere characteristics
- Septic tanks
- Wildfowl
- Fishing & Angling
- Atmospheric inputs
- Diffuse inputs (some: Water, Open shrub heath, Broadleaf/mixed woodland, coniferous forest, improved grassland, and Fen)
- Outflow annual volume

In repairing these links it was observed that peak counts had been used for geese contributions instead of mean count. This was corrected.

9.5.2 Error in seasonal flow distribution coefficients

In the Front End tab, Diffuse INPUTS (LAND USE) section, a table of monthly coefficients allows for seasonal variation in flows to be approximated from mean annual values (not applied to all land cover types). There was an error in the case of the December coefficient, which was accidentally assigned a value of 50% rather than 0.5%.

9.5.3 Introducing revised data

Data used to drive the model for each site are contained in a series of tabs (Sediment, Land cover, Stocking, WwTW, Septic tanks, Annual rainfall, Birds, Fish and angling, Catchment, mere size and depth, Groundwater, Mean monthly P and Mean monthly N). Data contained in these tabs are presented and discussed in Atkins 2010. All field are editable, and revised values can be entered. To prevent confusion of data sources, a series of version of LESA-NP were created, differing only data values used. In each case new data columns were clearly labelled, and the original data was copied to an inactive column in the corresponding tab.

9.5.4 Sediment P load

Discussed at length in Section 7.4, the approach adopted by Atkins (2010) for estimated the internal P load (flux of P from the sediment the lake water) is too high by more than 10-fold. We have removed the Atkins (2010) internal P load by assigning a value of zero to the P coefficient in the SEDIMENT box (FRONT END tab), and added in a value based on a simple empirical model (see Section 7.4 for details).

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11 List of abbreviations

CSM – Common Standards Monitoring

GES – Good Ecological Status

HES – High Ecological Status

WFD – Water Framework Directive

WwTW – Wastewater treatment work

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