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Soil Loss into Water

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Welsh Government




Soil Loss into Water

Soil Policy Evidence Programme

Report SPEP2018-19/09

Submitted to

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EXECUTIVE SUMMARY

Agriculture has been identified as a significant source of soil loss into water in Wales. In order to better support policy development and provide ministerial advice, this study reviewed the evidence base to arrive at a more robust understanding of the contribution of different land uses to overall soil losses into water in Wales.

Wales differs from England in being a largely mountainous country with high rainfall, where the majority (around 80%) of the agricultural land area is managed as grassland or rough grazing. As a consequence key factors controlling soil erosion (and hence soil losses to watercourses) such as slope, rainfall volume and intensity and vegetation cover are very different from those in England.

A considerable amount of Defra and WG funding has been devoted to developing a modelling framework for quantifying sediment (and nutrient) losses to water courses, and apportioning them to different point and diffuse sources. This approach has been used in previous policy focussed studies, including those which have assessed the effectiveness of Welsh Agri-Environment Schemes. Estimates from these previous studies have indicated that, at the national scale, agriculture was the dominant source of sediment losses to water courses in Wales (62%), with river bank erosion contributing around 27%. In comparison, the other sediment sources considered in the model (forest and woodland, urban runoff, discharges from sewage treatment works) provided a relatively minor contribution (<10% each) to the overall losses, although they could be important a local level.

In view of the fact that the existing modelling framework has received considerable investment and has already been widely adopted for policy support studies by both Defra and WG, we do not consider that the development of a new approach is justified.

However, there are a number of areas where the data used to populate the model could be updated or refined, and validation of the model outputs improved. In particular, attention could be given to improving the channel bank erosion model, better representation of landscape and in-channel sediment retention, including some consideration of the condition of land drains and improving the estimates of sediment losses from areas of rough grazing.

There is also an opportunity for future research to complement existing soil erosion monitoring and data collection programmes including:

- An exploration of 'source-to-sink' connectivity using aerial photography or satellite images to identify and map the frequency of erosion features that traverse field margins and field boundaries delivering sediment to rivers or road drainage systems.
- Targeted measurements of specific sediment sources including bankside, footpath and roadside verge erosion, and mining discharges, although the latter are unlikely to affect the overall balance of losses at the national scale.
- An investigation of how stakeholder engagement could be used to encourage farmers to use new technologies to provide feedback on soil health and condition, and more effectively engage with issues surrounding soil erosion and loss.

CONTENTS

1 INTRODUCTION AND OBJECTIVES

This scoping study was initiated in response to a request from the Welsh Government (WG) to develop a robust evidence base to understand the contribution from different land uses of soil losses into water, in order to support policy development and ministerial advice.

Agriculture has been identified as a significant source of soil loss into water in Wales. However, there is a need for robust evidence to confirm agriculture's contribution to soil loss or to identify alternative sources. Data for England may not be pertinent to the situation in Wales because most of the agricultural land is managed as grassland systems and climatic conditions are different.

This report sets out the soils, climatic conditions and land use in Wales, and briefly considers the implications for soil erosion and loss. We then consider previous national studies (for England and Wales as a whole) where sediment losses to water from a range of diffuse and point sources have been modelled and the losses apportioned to different sources. A similar source apportionment methodology was subsequently used for Wales to assess the effectiveness of Welsh Agri-Environment Scheme (WAES) on sediment (and other pollutant) losses to water.

The report contains and in-depth examination of the outcomes of the modelling approach used in Wales, and at the data and other evidence used to populate and validate the model. The sources of soil into water considered in the model are examined together with other potential sources of soil/sediment that are not currently accounted for, including an assessment of evidence and data gaps, and recommendations for further research to improve model outputs.

The report concludes with a brief assessment of novel technologies and alternative approaches that could be used to obtain a more robust evidence base on soil losses to water in Wales.

2 SOILS, CLIMATE AND LAND USE IN WALES

2.1 Topography and soils

Wales is a mostly mountainous country, dominated by the Cambrian Mountains in the centre and north (Snowdonia), and by the Brecon Beacons in the south. In contrast, wide river valleys characterise the land along the eastern border with England.

Soil conditions are strongly influenced by topography, climate, vegetation, land use, the sediments or rocks from which the soils have developed and the underlying geology. There are 10 major soil groups (based on pedogenic characteristics) within the soil classification for England and Wales, all of which are present in Wales (Avery, 1980), with the most common being podzols, brown soils and surface water gleys (Table 1).

Table 1. Major soil groups in Wales (Avery, 1980)

Major soil group	Land cover (%)	Description
Terrestrial raw soils	<0.1	Very young soils with only a superficial organo-mineral layer
Raw gley soils	0.2	Unripened young soils of saltmarshes
Lithomorphic soils	2.2	Shallow soils without a weathered subsoil
Pelosols	0.1	Clayey 'cracking' soils
Brown soils	30.2	Loamy, permeable soils with weathered subsoil
Podzolic soils	32.3	Acid soils with brightly coloured iron-enriched subsoil
Surface-water gley soils	24.7	Loamy and clayey seasonally waterlogged soils with impermeable subsoil
Ground-water gley soils	3.4	Soils associated with high seasonal groundwater
Man-made soil	0.4	Restored soils of disturbed ground
Peat soils	3.4	Soils in deep peat
Unclassified land (urban)	3.0	

To provide an overview of the soil texture, drainage, fertility, land cover, habitats, topsoil carbon, drainage and general cropping guidance Cranfield University recently described 27 'soilscapes' (http://www.landis.org.uk/soilscapes/index.cfm). These were designed to provide "extensive, understandable and useful soil data for a nonsoil specialist". There is no direct relationship between the major soil groups in Table 1 and the 'soilscapes' in Table 2; the first classification forms part of an in-depth site-specific assessment whereas the latter is intended to give a more broad overview.

The predominant soilscapes in Wales (i.e. those with the largest % land cover) and their erosion risks are described in Table 2, with the full range and distribution shown in Figure 1. Overall, there is a scarcity of high quality agricultural land in Wales, with that considered to be the best and most fertile accounting for only around 10-15%[1] of the agricultural land area.

[1] Estimate based on findings of Soil Policy Evidence Programme report SPEP2018-19/03, Historic Loss of BMV Agricultural Land, and Welsh Government specialist advice. BMV land as a percentage of total agricultural area is estimated to be 17.6%; however, this does not take account of frost risk, chemical limitation, pattern limitation and agricultural flood risk; therefore, an estimate range of 10-15% takes account of this uncertainty.

Table 2. Dominant soilscapes of Wales and their susceptibility to erosion.

Soilscape	Area (%)	Texture	Susceptibility to erosion
Freely draining slightly acid soils	27.3	Loamy	Siltation and nutrient enrichment of streams from soil erosion on certain of these soils
Slowly permeable soils	24.9	Loamy & clayey. Peaty or humose loamy	Fine sediment can move in suspension with overland flow from compacted/poached fields or in drain water. Gripping or over grazing, particularly in winter, can lead to run-off and erosion. Out-wintering and stock feeding practices need care to avoid loss of vegetation and erosion.
Freely draining acid loamy soils over rock	22.6	Loamy	Some risk of soil erosion during reseeding, from unmetalled roads accessing higher land and scaring by sheep on very steep banks.
Very acid loamy upland soils with a wet peaty surface	8.9	Peaty	Gripping or over grazing, particularly in winter, can lead to accelerated erosion. Out-wintering and stock feeding practices need care to avoid loss of vegetation and erosion.
Peat soils	3.3	Peaty	Vulnerable to erosion where vegetation is lost, and difficult to revegetate.
Freely draining floodplain soils	1.8	Loamy	Flooding of cultivated fields can scour topsoil and increase silt in rivers.
Slightly acid loamy & clayey soils with impeded drainage	1.8	Loamy, some clayey	Farmed land is drained and therefore vulnerable to run-off and rapid through-flow to streams; surface capping can trigger erosion of fine sediment.
Loamy & clayey floodplain soils with naturally high groundwater	1.6	Loamy & clayey	Close proximity to river results in pollution risk from floodwater scouring.
Shallow soils	1.4	Loamy Peaty	Surface capping and erosion of chalk soils on steeper slopes under cereals. Overgrazing can damage vegetation and lead to erosion of the peaty surface
Loamy & clayey soils of coastal flats with naturally high groundwater	1.3	Loamy & clayey	Mostly drained. Shallow groundwater and marginal ditches to most fields; water is vulnerable to pollution from nutrients, pesticides and wastes.

Excludes soilscapes which comprise <1% of the area i.e. saltmarsh & sand dune soils (0.8%); restored soils mostly from quarry & opencast spoil (0.7%); freely draining very acid sandy & loamy soils (0.7%); naturally wet very acid sandy & loamy soils (0.2%); loamy soils with naturally high groundwater (0.1%). Source: Cranfield University (2016) and Soilscapes website ([http://www.landis.org.uk/soilscapes/index.cfm)](http://www.landis.org.uk/soilscapes/index.cfm)

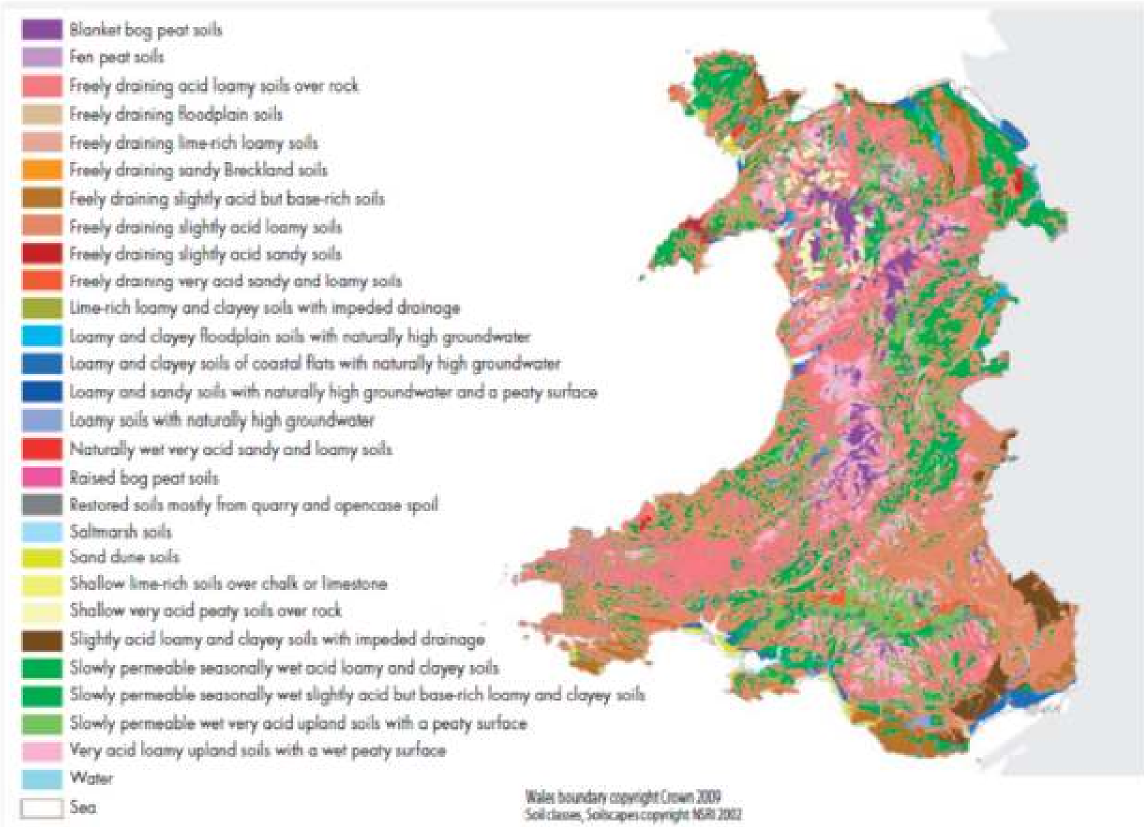


Figure 1. Soilscapes of Wales (Cranfield University, 2017).

2.2 Climatic conditions

Wales has a cool temperate climate, with cool summers and mild winters. The climate is strongly influenced by prevailing weather systems from the Atlantic and the mountainous terrain. Of the total land area of Wales, 60% is more than 150 m above sea level, and 27% is more than 300 m above sea level (Russell et al., 2011).

Rainfall in Wales varies widely (Figure 2). Snowdonia is the wettest area with average annual totals exceeding 3000 mm, comparable to those in the English Lake District or the western Highlands of Scotland. In contrast, coastal areas and those close to the border with England, are drier, receiving less than 1000 mm a year (Met Office, 2016). Over the ten years 2007-2016, average annual rainfall in Wales was 1467 mm with the wettest months in winter (November to January) and the driest in late spring (March and April) and early summer (May), Figure 3.

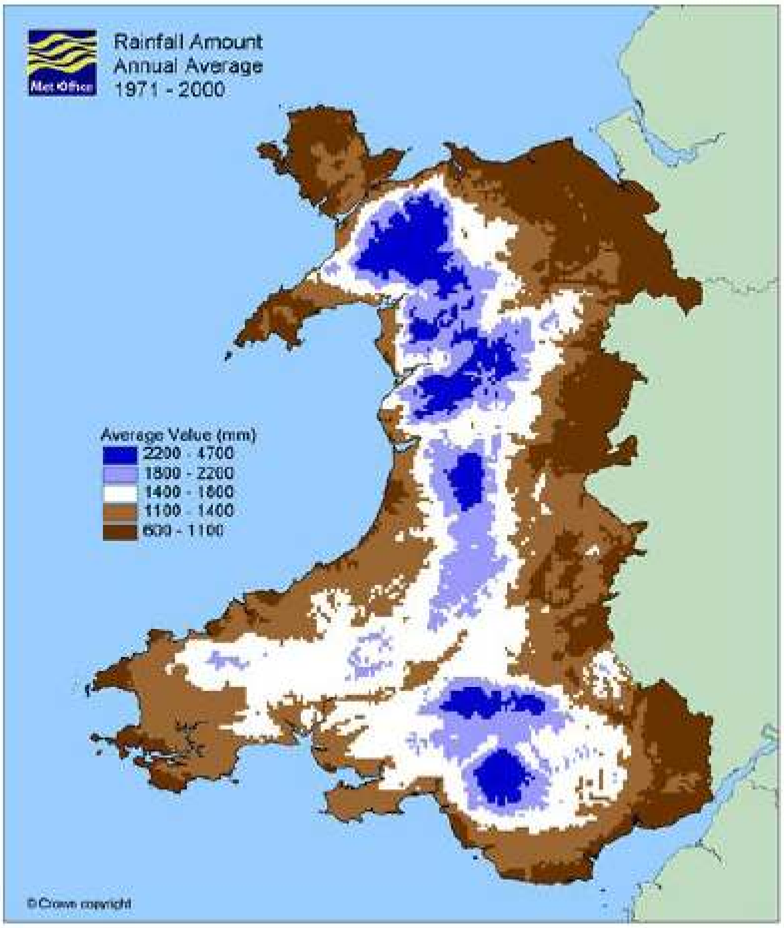


Figure 2. Average annual rainfall for Wales (1971-2000). Source: Met Office (https://www.metoffice.gov.uk)

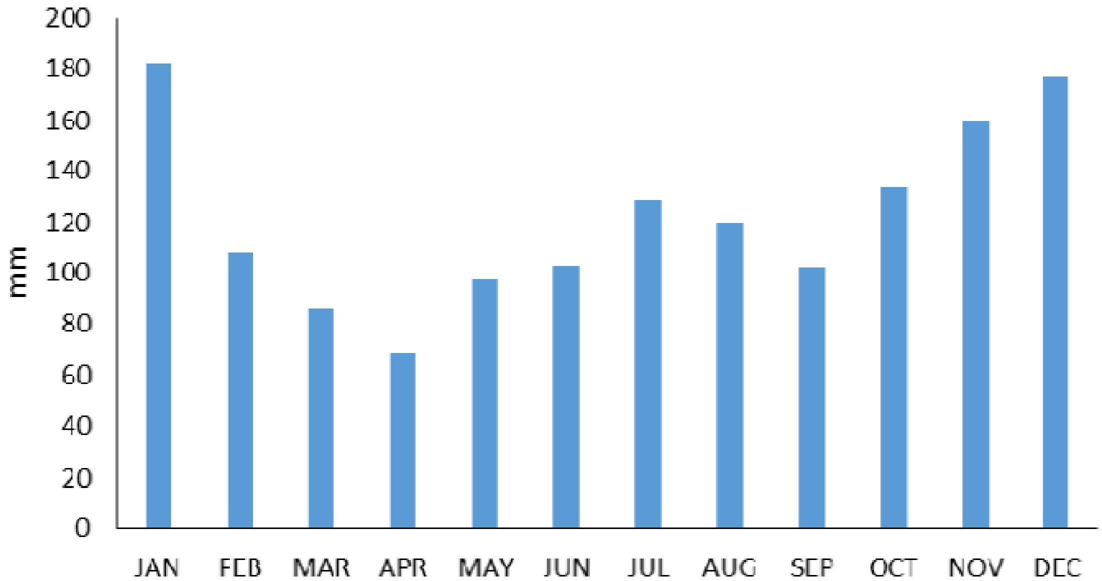


Figure 3. Mean monthly rainfall for Wales (2007-2016)

A report on changing trends in the climate of the UK (Jenkins et al., 2007; Forest Research, 2019) found that in Wales between 1961 and 2006 there had been a 13.6% increase in total precipitation and an increase in the number of rain days (>1mm) of 5.7 days. The data also showed a trend of changing seasonal distribution of rainfall, with slightly less in the summer and more at other times. There has been a reduction in the contribution of heavy rainfall events in summer months; however in the winter months there has been a trend of increasing heavy rainfall events in central and south Wales, with little change in north Wales (Figure 4).

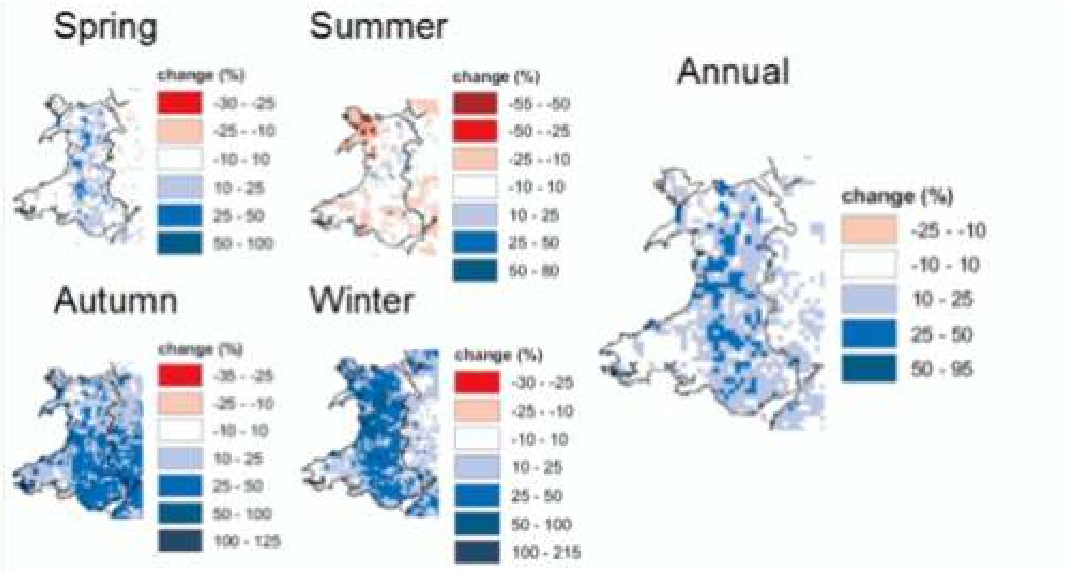


Figure 4. Changes in seasonal and annual total rainfall in Wales between 1961 and 2007 (Source: Forest Research, 2019)

2.3 Land cover and use

Agricultural land accounts for over 88% (1.8 million ha) of the total land area of Wales, and includes land for growing crops, grazing for livestock, woodland on farms, farm buildings, and other land on farms (Welsh Government, 2018). The dominant agricultural activity is cattle and sheep grazing; as a result, permanent grassland and rough grazing comprise 57% and 24% of the agricultural area, respectively. The total area of arable land was 247,059 hectares in 2016, which is equivalent to 13% of the agricultural land area (Welsh Government, 2018); woodland on agricultural holdings was 5% of the total area, with all other land on agricultural holdings accounting for 1% of the total (Figure 5).

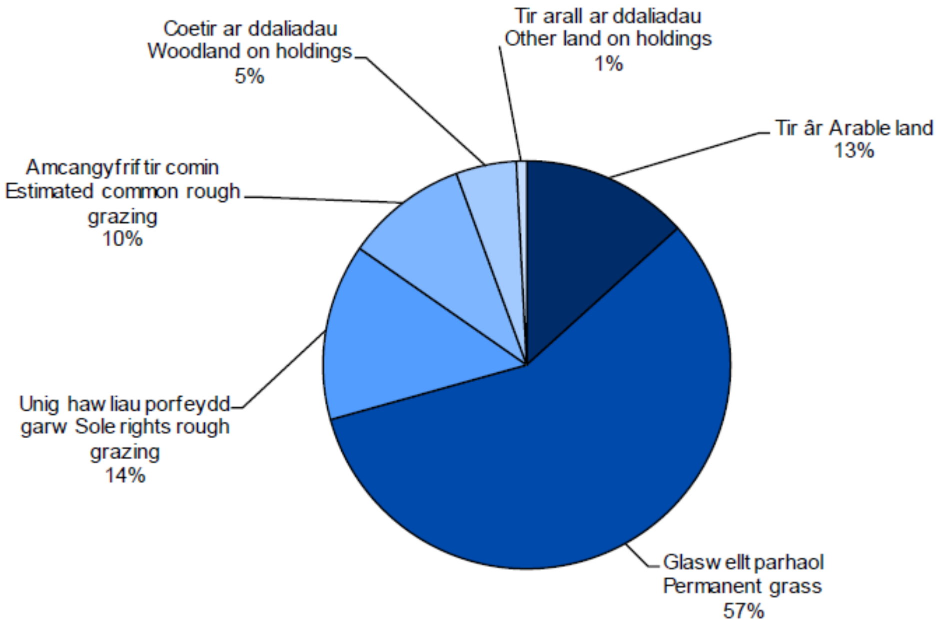


Figure 5. Area of agricultural land in Wales in 2016 (Welsh Government, 2018).

According to the Forestry Commission, woodland accounts for 15% of the total land area of Wales at 309,000 ha, approximately half of which is conifers and half broadleaf woodland (Forestry Commission, 2019). Urban areas including built up areas and gardens account for 4%, whilst bog and heath in upland areas comprise c. 7% of the land area (Morton et al., 2011).

3 SOIL EROSION

3.1 Soil erosion processes

Soil erosion is the displacement or loss of the upper layer of soil and occurs through the action of erosive forces such as water, wind and human/animal activity. Soil erosion has been extensively described in previous reviews and reports (e.g. Boardman, 1990; Knox et al., 2015; Cranfield University, 2016a;b), but a brief outline of the main processes involved is given below.

Water erosion. Water erosion occurs when raindrop impact detaches soil, destroys aggregates and transports soil particles in the runoff water. The severity of soil erosion by water depends on:

- The duration and intensity of rainfall – short duration heavy rainfall events are more damaging than gentle rainfall over a longer time period.
- The nature of the soil – infiltration capacity and structural stability (influenced by soil texture and organic matter content) will influence susceptibility to erosion
- Slope, topography and channels – the greater the degree and length of the slope, the greater the soil loss. The development of channels (rills and gullies) tends to concentrate the damage.
- Vegetative cover – forests and grasses are effective in protecting the soil from water erosion.

Wind erosion. Wind erosion involves the detachment and transportation of soil particles by the action of the wind. Wind erosion is related to soil texture and moisture content and the amount and type of vegetation cover; wet, grassland soils do not blow. The rate of wind erosion can be greater than by water partly because wind erosion is likely to impact on a whole field while erosion by water is limited to areas where the water flow is concentrated (Owens et al., 2006). Wind erosion is mainly found on sandy and peaty soils in the eastern and middle counties of England, and in parts of the uplands of England and Wales. There are few studies on wind erosion specific to Wales, although sand dune erosion/migration in North Wales has been reported (Wiggs et al., 2002; Bailey & Bristow, 2002).

Tillage erosion. Studies have shown that tillage erosion can be of equal or greater importance than water erosion, particularly on fields with short slope lengths. Important factors influencing soil translocation by tillage include bulk density, initial soil conditions (such as moisture content) and tillage history. The type of tillage is also important in terms of depth, direction, speed and size of tillage implement.

Soil loss through co-extraction on crops and vehicles. Soil can be transported 'out of field' when it is co-extracted on the roots of crops and/or on the wheels and implements of farm machinery (Owens et al., 2006), and significant amounts of soil may be lost from fields in this manner. No data on the magnitude of such losses specific to England and Wales have been reported. However, Frost & Speirs (1996) assessed soil loss for a site in East Lothian, Scotland due to the harvest of (mainly) potatoes and carrots to be 1 t/ha/harvest. In 94% of the studied area; this was greater than the soil loss resulting from water erosion caused by a severe storm with a 20 year return period.

Upland erosion. The combined effect of water, wind, frost and animals acting on bare soil in the uplands is often referred to as 'upland erosion'. Areas of land, typically in the uplands are often subject to recreational and agricultural pressures resulting in accelerated erosion through soil disruption and vegetation removal. Overgrazing is one example of this whereby soil exposed to the elements in upland areas can increase the likelihood of wind and water erosion (Cranfield University, 2016b).

3.2 Extent and rate of soil erosion in Wales

There have been a number of literature reviews that have reported estimated rates of soil erosion for England and Wales (e.g. Owens et al., 2006; Knox et al., 2015; Rickson et al., 2016).

In England and Wales, 'natural' (background) rates of erosion are estimated to be generally <0.5 t/ha/yr (Table 3), mostly ranging from 0.1-1.0 t/ha/yr based on sediment yields (Evans, 2006), although a portion of this can be attributed to river channel erosion (see Section 5.4). Higher erosion rates occur most commonly on sandy and light silty soils on sloping land, often when the surface roughness is low and vegetation cover is less than 25-30% (Evans, 1990). Accelerated soil erosion rates are in the order of 1-10 t/ha/yr (Chambers *et al*., 1992; Chambers *et al*., 2000; Evans, 2005; EA, 2002), but rates in excess of 200 t/ha/yr have been reported for thin silty clay and silt loam soils of the eastern South Downs in England related to wet autumns and winter cereal cropping (Boardman, 1990).

Table 3. Annual rates of erosion (t/ha/yr) in the UK

Background	Cultivated	Bare soil	Grassland (by cross compliance soil type)*
0.1-0.5	0.1-10	10-200	Light - 0.54 Medium – 0.56 Heavy – 0.72 Chalk/limestone - 1.37

Sources: Boardman (1990); Morgan (2004); Evans (2005)
*Data from Evans et al. (2017). Mean erosion rates calculated using conventional methods (i.e. a combination of sediment source apportionment, suspended sediment yield and landscape retention factors). Assumes a landscape retention multiplier of 1.45 (i.e. long-term sediment storage of 30% = a sediment delivery ratio of 0.7)

A commentary on the susceptibility of the different Soilscapes of Wales to soil erosion is provided in Table 2. A recent analysis of the extent and severity of soil degradation in Wales (Cranfield University, 2016a) reported that Wales is predominantly susceptible to water and upland erosion (Figure 6). They stated that quantified estimates of the severity or extent of erosion processes have not been recorded, and modelled estimates of soil erosion by water have not been validated due to lack of field data/observations. Large areas of Wales are assumed to have low erosion rates due to the predominance of grassland and rough grazing (see Section 3.3), although this has not been quantified by measurements or observations (Cranfield University, 2016a). Therefore the estimates in Table 3 require further validation before they can be assumed to be applicable to Welsh soils.

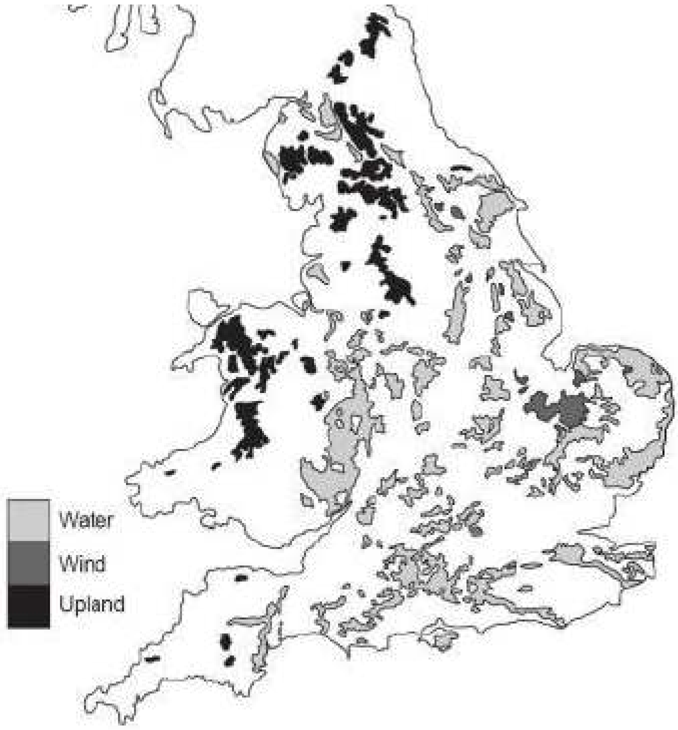


Figure 6. Areas of soil erosion risk in England and Wales, based on land use, soil type and slope. Note: upland erosion includes both water and wind erosion. Source: Boardman & Evans, 2006.

In addition, Cranfield University (2016a) reviewed more than 173 reports and summarised the predicted effects of climate change in Wales which could affect soil erosion and sediment losses, *viz*:

- There will be little or no change in the duration of soil wetness (field capacity days) or potential soil moisture deficits until 2080.
- Shifts in the seasonal distribution of the field capacity period will mean drier autumn and wetter spring periods.
- The waterlogging duration of slowly permeable soils will be reduced by 2080
- Increased grassland productivity will lead to higher stocking densities which could be associated with more severe and extensive soil compaction and erosion.
- The severity and extent of soil erosion in upland areas will be increased in autumn by 2090, but there will be less erosion in winter (despite higher rainfall amounts), due to increased vegetation cover.

4 PREVIOUS STUDIES ON SEDIMENT LOSSES TO WATER AND SOURCE APPORTIONMENT

There have been a number of previous research studies where sediment losses to water from diffuse agricultural and other sources have been calculated, and the outputs used to apportion the sediment loads to the different sources. Whilst the methodology was developed for England and Wales as a whole, it has also been used to support policy studies specific to Wales (i.e. the impact of WAES). In this section we briefly describe these studies and their main findings.

4.1 National (England and Wales) Source Apportionment Studies.

4.1.1 Sediment Gap Analysis (Defra Project WQ0106)

In 2006, a substantial piece of work was conducted as part of Defra Project WQ0106 (Sediment Gap Analysis to Support WFD) to develop and implement a methodology for estimating the diffuse agricultural source contribution to critical suspended sediment (SS) concentrations in receiving fresh river waters across England and Wales. This also involved an assessment of the contribution of river bank erosion, diffuse urban sources and point source sediment discharges to the total sediment load in river catchments.

Development, calibration and application of a model of predicting SS concentrations required the collation and construction of a number of environment datasets, describing diffuse and point source sediment inputs to catchments across England and Wales. Sediment losses were assessed using the Diffuse Pollution Inventory (DPI) methodology based on farm type, soil texture and climatic conditions.

Sediment inputs from diffuse agricultural sources across England and Wales, excluding urban areas, were calculated using the PSYCHIC - Phosphorus and Sediment Yield Characterisation in Catchments – model (Davison et al., 2006; Stromqvist et al., 2006; Collins et al., 2007) with agricultural census and land cover data for the year 2000. Inputs from diffuse urban sources were estimated using an Event Mean Concentration (EMC) methodology whereby calculations of average annual runoff are combined with event mean concentrations to estimate the annual sediment load. Sediment from river bank erosion was estimated using a prototype channel bank erosion model and inputs from sewage treatment works (STWs) using an inventory of effluent point discharges. The total SS load for each WFD sub-catchment in England and Wales was estimated by summing the individual predicted loads from the different sources, taking account of landscape retention.

Table 4 details the estimated national total SS loads in England and Wales apportioned to the major sources; for all the regions the most important source of SS was from diffuse agricultural losses (including losses from arable, managed grassland, rough grassland, and wood and forest); in Wales *c.*74% of total sediment losses were estimated to come from agricultural land. Figure 7 shows that for Wales the great majority of the diffuse agricultural losses were from grassland (*c.*87%), with a much smaller contribution from arable land (*c.*10%) and the remainder (*c.*4%) from wood and forest.

Figure 8 maps the total calculated sediment export from all sources, illustrating the relatively high predicted SS yields in Wales and northern England compared with central and eastern England. Figure 9 shows the percentage contribution from bank erosion and agricultural land (excluding rough grazing), respectively.

Table 4. Modelled total annual average suspended sediment inputs (kt)[1] to river systems in England and Wales, summarised by EA region, and apportioned by source. (Anthony & Collins, 2006)

Region	Diffuse Agricultural[2]	Diffuse Urban	Bank Erosion	Sewage Works
Anglian	181.9	10.3	8.7	11.3
Wales	430.7	15.5	135.5	3.3
Midlands	288.5	22.8	53.0	19.6
North East	329.8	19.2	73.6	12.0
North West	233.6	26.7	45.9	15.7
South West	246.7	14.7	52.2	2.6
Southern	157.3	12.5	11.2	2.1
Thames	60.6	25.2	14.0	9.8

[1]The suspended sediment load originating from each source was estimated net of landscape retention (i.e. not all the eroded soil will reach the river system as a proportion will be trapped and retained by landscape features such as hedgerows and bankside vegetation).
[2]Diffuse agricultural estimates are based on 2000 Agricultural Census and landcover data and 1961-1990 climate data.

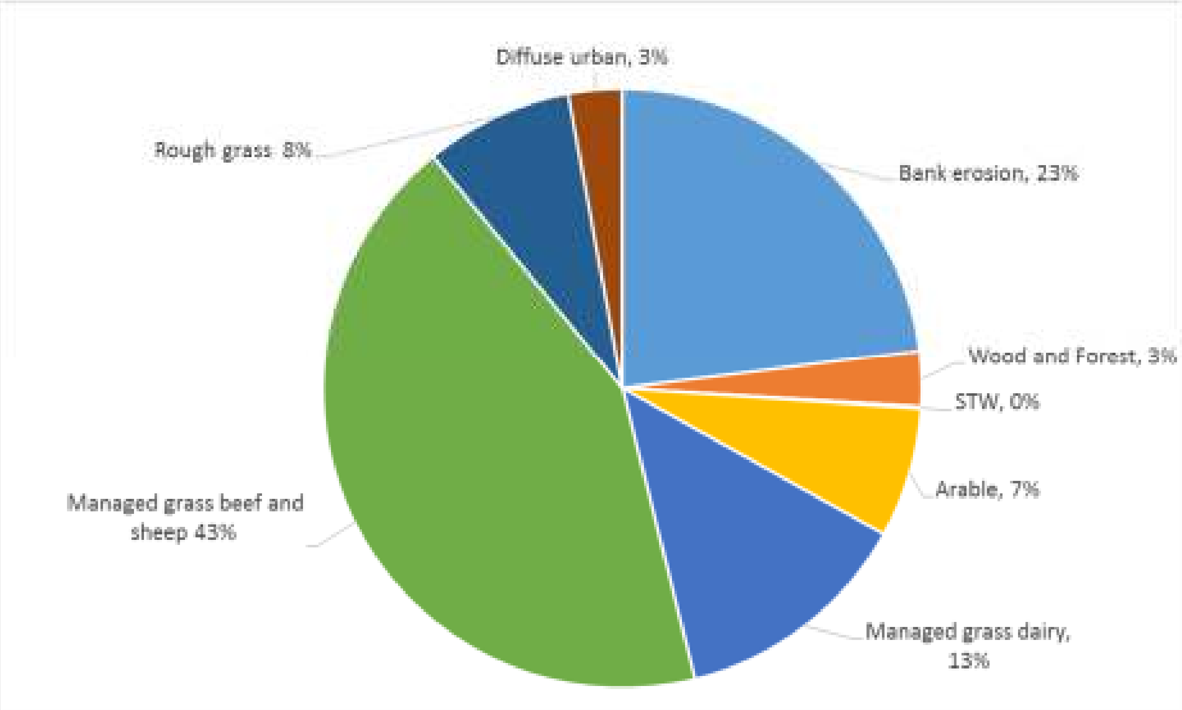


Figure 7. Sources of suspended sediment inputs to river systems in Wales (derived from Anthony & Collins, 2006).

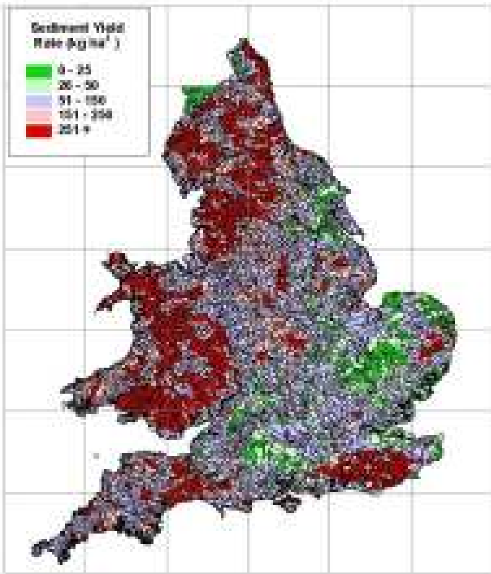


Figure 8. Modelled annual average SS yield at the outlet of each WFD sub-catchment across England and Wales, from all sediment sources (Anthony & Collins, 2006).

a) b)

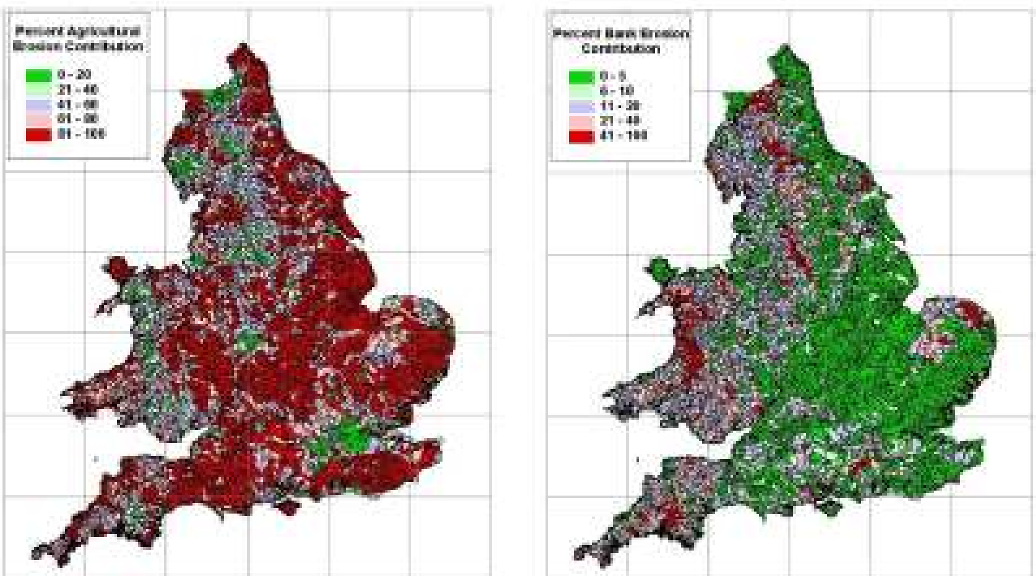


Figure 9. Modelled percentage contribution of a) diffuse agricultural sources and b) channel bank erosion to the annual average SS yield at the outlet of each WFD sub-catchment across England and Wales (Anthony & Collins, 2006)

Full details of the methodology and data used in the study can be found in the Defra final project report (Anthony & Collins, 2006) and in several published scientific papers (Collins & Anthony, 2008; Collins et al., 2009a;b).

4.1.2 SEPARATE (Defra project WQ0223).

The Defra-funded SEPARATE study built further on the approach described above. It comprised the development of a new national multiple pollutant (i.e. nitrogen, phosphorus and sediment) source apportionment screening framework for England and Wales. SEPARATE (SEctor Pollutant AppoRtionment for the AquaTic Environment) includes emissions to the aquatic environment from both diffuse (agriculture, urban, river channel banks, atmospheric) and point (STWs, septic tanks, combined sewer overflows -CSOs, storm tanks) sources and summarises the source apportionment on the basis of Water Framework Directive cycle 2 waterbodies (Zhang et al., 2014).

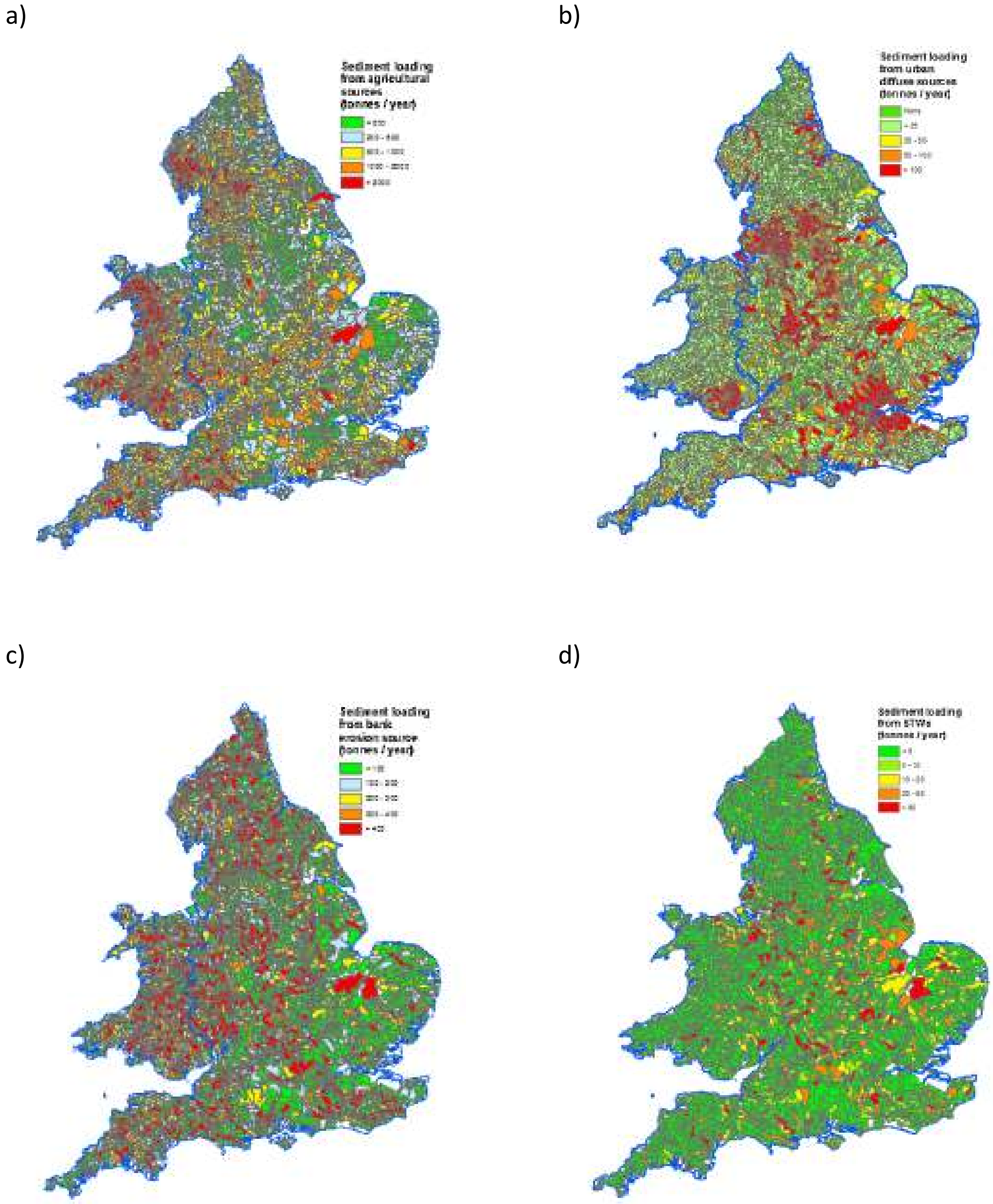


Figure 9. Waterbody scale total annual delivery of sediment from a) diffuse agricultural sources b) diffuse urban sources c) channel bank erosion and d) STWs to rivers across England and Wales.

Zhang et al. (2014) reported that total sediment delivery to rivers at waterbody scale ranged from 0 to 17,726 t/yr from agriculture, 0–1,398 t/yr from diffuse urban sources, 0–4,178 t/yr from river channel banks and 0–1,510 t/yr from STWs (Figure 9). National scale source proportions (with water-body ranges) for sediment were estimated to be in the order; agriculture (72%) > river channel banks (22%) > diffuse urban (5%) > STWs (1%). This national scale source apportionment is consistent with that reported previously (Collins & Anthony, 2008; Collins et al., 2009a,b) but differs slightly because an updated model was used for the agricultural sector, a modified index for channel bank erosion and the fact that the monitored sediment concentrations for STW outfalls for the period 2010–2012 (10–30 mg/L) used by this study were lower than those used previously (30–70 mg/L) reflecting stricter consents.

4.2 Source apportionment studies for Wales

To ensure consistency across a number of projects used to support government policy development, the modelling approach used by Anthony & Collins (2006) for the 'Sediment Gap Analysis' described above was adopted as part of a major project to estimate the effectiveness of the Welsh agri-environment schemes (WAES) in maintaining and improving soil and water quality (Project No. 183/2007/08; Anthony et al., 2012). The schemes considered were Tir Cynnal, Tir Gofal and the Organic Farming and Maintenance Schemes.

4.2.1 Diffuse agricultural sources

The study modelled 'present day' diffuse pollution from agricultural land that incorporated the effect of any mitigation method implementation (according to the results of the Wales Farm Practice Survey) on both scheme and non-scheme farms, and any changes in fertiliser use and livestock numbers that occurred on entry to Tir Gofal and the Organic Farming and Maintenance schemes. The calculations were based on agricultural census and farm activity data for 2004, but the farm types and areas of land participating in the WAES were based on scheme data for 2009. The model results represented the best estimate of pollutant losses from agricultural land that might be compared with environment monitoring data.

Average modelled sediment emissions from agricultural land were 120 kg/ha, with an inter quartile range of 59 to 225 kg/ha (Figure 10). The modelled losses compare favourably with measured sediment yields in the range 40 to 320 kg/ha for 12 small upland lake catchments in England and Wales, with forest, rough grazing and moorland land cover (Foster et al., 2011), but were lower than the measured average field rates of erosion on lowland arable and grassland in the range 220 to 4,890 kg/ha derived by ground survey and interpretation of aerial photography (Evans, 1988). However, the latter survey was targeted at areas known to erode and also does not account for the retention of sediment in the landscape.

Present day losses were calculated and summarised as pollutant loads across the total agricultural land area by farm type, and for organic and conventional management across the whole of Wales. For sediment, the highest loads (kg/ha) were estimated to be from the Conventional Horticulture and Organic Cereal farm types, however the greatest proportion (74%) of the national total losses was from farms in less favoured areas (CSLFA) reflecting the high proportion of welsh agricultural land which is of this farm type (Table 5).

Table 5. Modelled present day national annual average sediment loads and the modelled proportion of the national total emission from all agricultural land in Wales (Anthony et al., 2012).

Farm type	Conventional load (kg/ha)	Organic load (kg/ha)	Proportion of national total emission (%)
POULTRY*	168	388	<0.1
CEREAL	323	395	2.8
GENERAL	276	321	1.0
HORTIC	399	278	0.4
PIG	129	n/a	0.1
DAIRY	162	155	11.7
CSLFA	207	187	74.0
CSLOW	122	100	5.6
MIXED	303	257	4.4

*POULTRY pollutant emission is artificially high as the farm area definition does not include neighbouring agricultural land that receives exported manure.
Note: The pollutant loads are averaged over the total agricultural land area including common land.

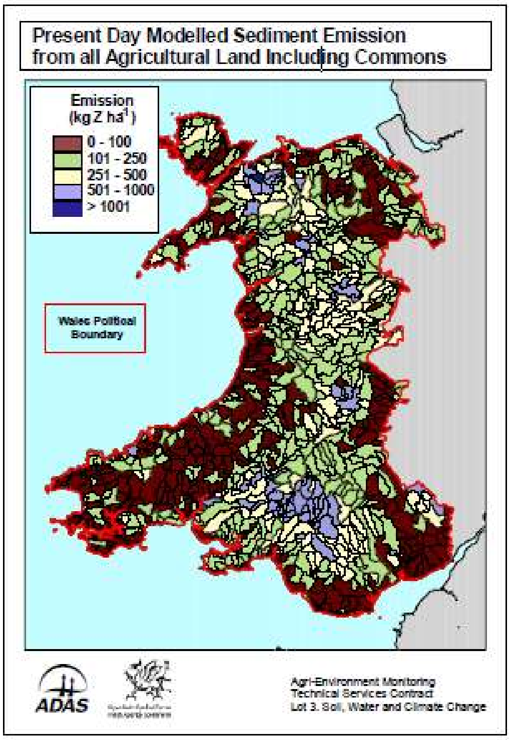


Figure 10. Modelled present day annual average sediment emissions from agricultural land in Wales. Includes the effects of soil compaction/poaching, the impacts of implementing mitigation methods (on both scheme and non-scheme farms) and changes in livestock numbers/fertiliser use due to participation in WAES. Sediment loads are averaged over the total agricultural land area including common land.

Looking at the source areas on the farm, the model outputs showed that 59% of the estimated sediment loss was from grassland, with 29% from rough grazing and 13% from arable land, reflecting the dominance of the grassland area. In terms of the delivery pathway, 61% of the sediment loss was estimated to originate from surface runoff and 39% from preferential/drain flow.

4.2.2 Non-agricultural sources

Estimates of non-agricultural pollutant losses from point and diffuse sources to rivers in Wales were developed. The point sources considered were sewage effluent discharges and septic tank discharges, whilst the diffuse sources comprised woodland runoff, road and urban runoff, other runoff (including bare soil, mining activity and the military training ground at Mynydd Epynt), and bank erosion. The emissions estimates were used to construct a spatial database of non-agricultural pollutant loads for each WFD river catchment (Figure 11).

4.2.3 Source apportionment

Combining the estimated losses from agricultural and non-agricultural sources showed that for sediment the major source was agricultural land at 62% of the total, with bank erosion accounting for 27% of the losses (Figure 12). The differences between these estimates and those made previously (see Figure 7) can be accounted for by updates to the various datasets used and refinements in the methodology. Nevertheless, the message presented is still that agricultural land is the major source of sediment losses to rivers in Wales, with bank erosion also making a significant contribution.

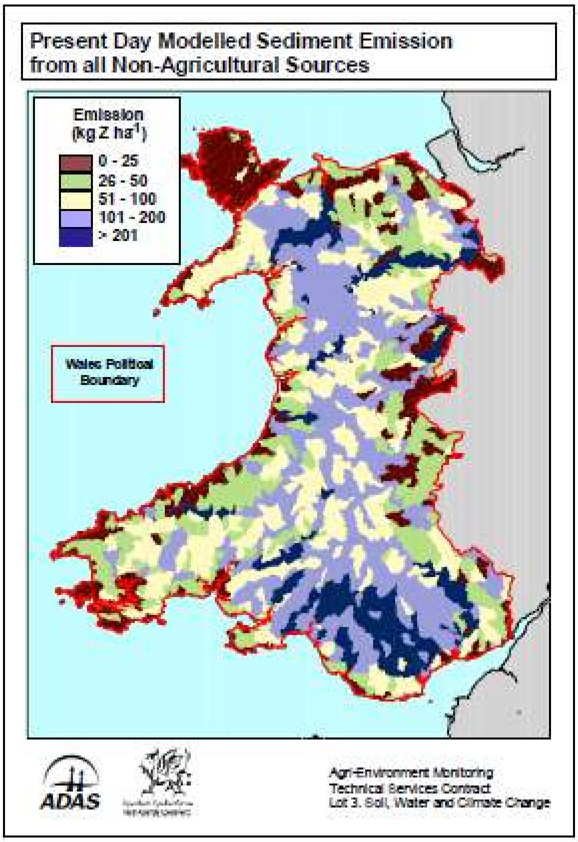


Figure 11. Estimated total annual emissions from all non-agricultural diffuse and point sources of sediment (Anthony et al., 2012)

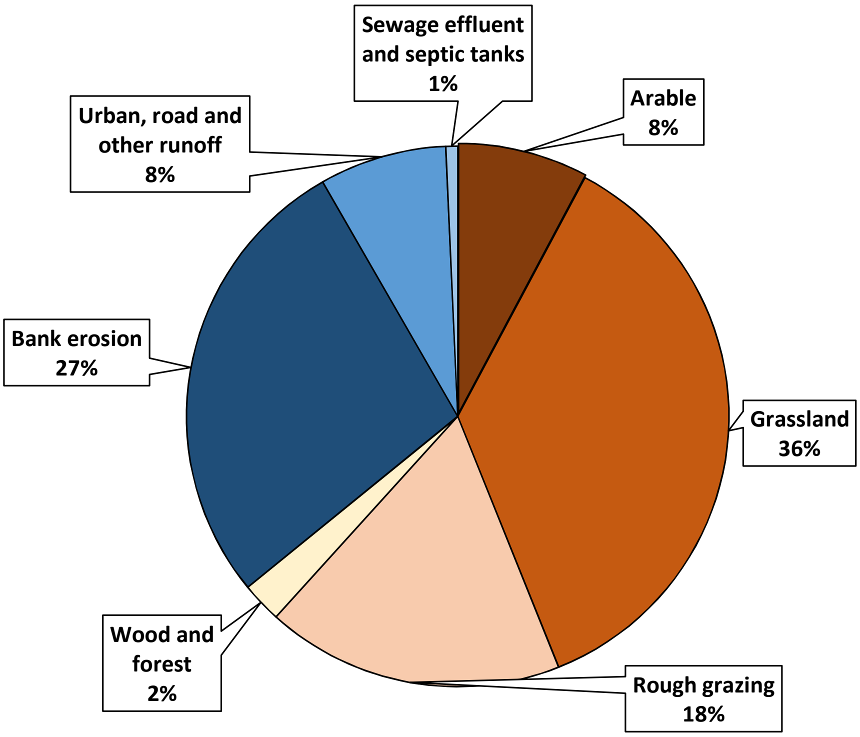


Figure 12. Sources of suspended sediment inputs to river systems in Wales (derived from Anthony et al., 2012). The total sediment input is c.509 kt/yr.

4.2.4 Impact of the Glastir scheme

The study of Anthony et al (2012) was undertaken to assess the impact of participation in WAES on maintaining and improving soil and water quality. The calculation of 'present day' emissions included the effects of soil compaction and poaching, as well as the implementation of mitigation methods and changes in livestock numbers and fertiliser use both on farms participating in schemes at the time (i.e. Tir Cynnal, Tir Gofal and the Organic Farming and Maintenance Schemes) and on non-scheme farms. The averaged 'present day' sediment load for agricultural land was estimated to be 197 kg/ha and the total sediment loss from all agricultural land in Wales was 315 kt.

Subsequently in 2017, a further study was undertaken to calculate the anticipated benefits of changes in management and stocking rates on farms participating in the new Glastir Scheme (Gooday & Whitworth, 2017). The study used the Farmscoper model (Gooday et al., 2014; 2015) and incorporated results from both the Second Welsh Farm Practice Survey (Anthony et al. 2016) and information from Glastir Scheme agreements to determine changes in farm practices or land management.

The calculated sediment losses before accounting for Glastir (203 kg/ha) and for farms in the Glastir scheme (212 kg/ha) were similar to those estimated by Anthony et al. (2012) and to measurements made throughout the UK for similar enterprises and environmental conditions. Beef and cattle enterprises within the Less Favoured Area remained the dominant source of sediment losses for farms participating in Glastir (see also Table 5). Overall, the current impact of Glastir was to reduce national sediment loads by 0.1%, and sediment loads for farms in the scheme by 0.4%. This small effect was because changing fertiliser use has no impact on sediment losses and the Farmscoper model does not account for the impact of changes in stocking density on compaction or poaching. Whilst individual Glastir management options can be very effective locally (e.g. field margin and riparian management for control of runoff), they are less effective at landscape level as any one option was generally taken up by less than

10% of scheme participants. However, if mitigation methods were fully implemented on all relevant land across Wales a reduction of 11% in the national sediment load might be achieved.

5 REVIEW OF DATA SOURCES AND APPLICABILITY FOR WALES

The methodology and data sources used for the previous source apportionment modelling undertaken for Wales (see Section 4.2) are described at length in Anthony et al. (2012). Here we look in detail at the data sources used by Anthony et al. (2012) and review the literature published since 2012 to identify whether there are any opportunities to tailor the model more specifically to Welsh conditions or to provide updated information/data to populate or validate the model for Wales.

5.1 Datasets

5.1.1 Data sources and mapping

Anthony et al. (2012) provides an overview of the agricultural, environment and WAES participation information that were integrated to provide the data framework for the source apportionment modelling undertaken for Wales.

The physical environment data comprised:

- Climate data (monthly average rainfall, temperature and number of rain days)
- Soils data (texture, organic matter content, bulk density, HOST – Hydrology of Soil Types - class)
- Landscape connectivity data (a connectivity index, calculated for each HOST class, and modified based on soil texture and slope to estimate sediment retention in the landscape).

A land cover map was created from the following source data on areas of:

- Agricultural land
- Common grazing land
- Urban settlements and roads
- Forest and woodland
- Open water
- Other land (i.e. bare soil and rock, mining activity and similar ‘other’ types of land cover plus the military training area at Mynydd Epynt).

A land use map was then created by separating the mapped arable, grassland and rough grazing land covers into specific crop types based on data from the 2004 June Agricultural Census. The agricultural land use data were disaggregated using a previously established methodology to map the area by farm type, and also by participation in WAES (representing the situation at the end of 2008).

5.1.2 Commentary on data sources.

Most of the physical environment data (i.e. soil data, topography) are unlikely to have changed greatly since the modelling was undertaken in 2012. However, the climate data used (monthly average rainfall, temperature and number of rain days) were for the ‘standard’ period 1961-1990. There is clear evidence that rainfall patterns in Wales have changed between 1961 and 2006 (see Section 3.2), and this may affect soil erosion (see Section 3.4). It would be interesting to explore the effect on sediment losses into watercourses using more recent climate data for Wales, and/or projections of future rainfall amounts and patterns (see Sections 2.2 and 3.2). There may also be the opportunity to explore the Basic Payment Scheme (BPS) crop code data for land use on 1st May averaged out over a rotation period, along with predicted changes in seasonality and intensity of rainfall.

5.2 Diffuse agricultural sources

The studies described in Section 4.1-4.3 estimated diffuse sediment losses from agriculture using the existing, validated PSYCHIC model (Collins et al., 2007, 2009a,b; Davison et al., 2008; Stromqvist et al., 2008; Collins & Anthony, 2008) and are net of landscape retention. PSYCHIC is capable of representing sediment losses both via surface flow and via preferential/drain flow, which is known to be an important sediment source in some catchments (e.g. Walling & Collins, 2005).

Some aspects of the PSYCHIC model estimates of diffuse agricultural sediment sources are worthy of further comment as discussed in the following sections.

5.2.1 Land drainage.

The extent and condition of under-drainage of arable and grassland has a significant influence on the delivery of sediments to watercourses, with an estimated 39% of the present day sediment loss arising via from the preferential and drainage flow pathway (Anthony et al., 2012). Because the PSYCHIC model explicitly represents this pathway, Anthony et al. (2012) verified the rules used to determine the presence of field drains for Wales.

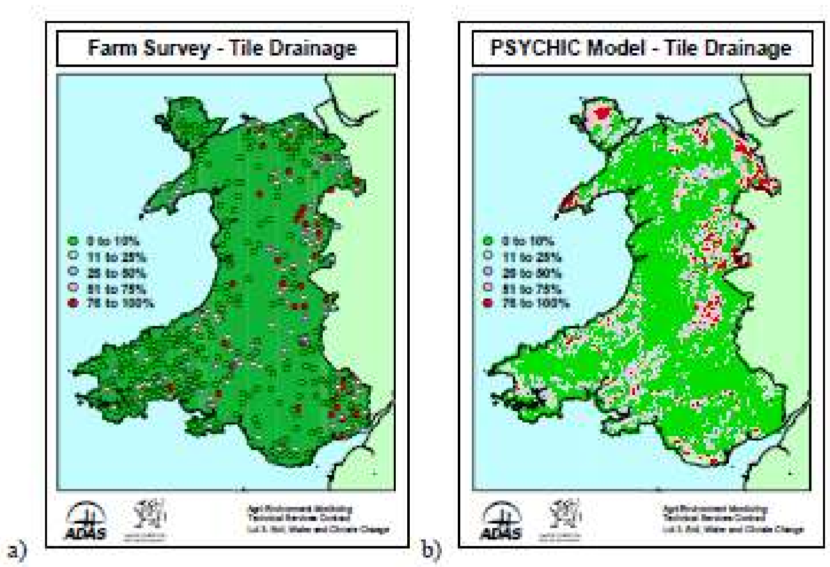


Figure 13. a) Proportion of the total arable and grassland area drained by field drains on farms surveyed in the WFPS; and b) proportion of the arable and grassland area predicted to be drained by the PSYCHIC model based on soil characteristics (Source: Anthony et al., 2012).

As part of the Welsh Farm Practices Survey (WFPS), respondents were asked what proportion of the total arable and grassland area (excluding rough grazing) was under-drained. The farm area weighted average value was 15% but this varied considerably spatially from 0 to 100% reflecting the underlying soil types. It was estimated that 12% of the improved grass and arable land area in Wales was drained, representing 44% of the area of land that required drainage. The PSYCHIC model (Davison et al., 2008) uses information on land use and soil texture to determine the need for field drainage. Application of the PSYCHIC model soil hydrology rules resulted in the tile drainage of 19% of the total area of improved grass and arable land in Wales. The WFPS results and other reports therefore broadly supported the drainage requirement rules used by the modelling framework (Figure 13).

As well as a simple representation of their presence/absence, the condition of land drains will also affect sediment delivery. This is not factored into the current modelling approach but it might be possible to use survey or observational data (e.g. from future WFPS) to inform future adjustments to the model for drain condition.

5.2.2 Landscape and in-channel retention

Diffuse sediment losses from agriculture estimated using the PSYCHIC model are net of landscape retention. There is a considerable body of evidence which suggests that sediment mobilised by overland flow within agricultural fields is rarely delivered to receiving waterbodies without some retention due to sedimentation or trapping along the flow path (Anthony et al., 2012). The model therefore uses a modifier to represent the retention of pollutants in the landscape.

Anthony et al. (2012) used a sediment delivery model to estimate a base connectivity index for soils of each HOST class which was modified to account for slope and soil texture. The resulting landscape delivery coefficient for Wales ranged from 0.1 to 0.95 with an average value of 0.55. Sediment inputs to watercourses via the surface pathway would therefore be expected to be c. 50% of the modelled losses to the edge of field. This connectivity index was used in a previous study of phosphorus emissions in Wales (Anthony et al., 2008), ensuring consistency with the previous work. In future, it may be possible to modify the connectivity index using data on landscape features which are known to control retention such as riparian area and hedgerows. Whilst such features have been included in physically based model of soil erosion, they are generally viewed as too complex and data demanding for use at the regional scale.

In order to compare modelled and measured pollutant concentrations (see Section 5.6), it was also necessary to account for in-channel sedimentation. The retention of sediment was modelled as a function of catchment area according to the empirical model of Vanoni (1975), which was based on an analysis of sediment input and yield data for 300 worldwide catchments.

Thus it can be concluded that both landscape and in-river sediment retention are still quite crudely represented in the PSYCHIC model, and there is therefore scope for improving what could currently be seen as a rather blunt tool.

5.3 Diffuse non-agricultural sources

5.3.1 Woodland runoff

The area of woodland and forestry within each WFD river catchment was taken from the land cover mapping (see Section 5.1). Particulate losses were based on PSYCHIC model estimates, which use a factor for ground cover in an equation for calculating suspended sediment losses from runoff shear. This factor represents obstacles on the ground which hinder runoff and ranges from 0.1 for bare soil to 0.95 for woodland (Davison et al., 2008). Average sediment losses were estimated to be 45 and 30 kg/ha for coniferous forest and broadleaved woodland, respectively, whilst the national total suspended solids load from woodland runoff was around 12,000 t. However, the accuracy of the predictions is uncertain as the PSYCHIC model estimates for woodland have not been calibrated against field-based measurements. In the following section we assess whether there is any data or other evidence that could potentially be used for validation purposes or to improve/update the model predictions.

There are some older studies on sediment losses from forestry which were either conducted in or are of relevance to Wales. For example, Robinson & Blyth (1982) reported that in Britain most forestry is on land that requires extensive drainage; the drainage of a small upland (peat) catchment in Scotland resulted in sediment yields over the following five years equivalent to nearly half a century's load at pre-drainage rates. Subsequent sediment yields did not decline to pre-drainage levels, but remained about four times higher, as a result of erosion of the drains. In mid-Wales, a twinned-catchment approach was used to measure the effects of afforestation and clearfelling on water and sediment yields (Leeks & Roberts, 1987). Afforestation and its associated practices increased sediment losses with drainage implicated as the main cause; felling operations also enhanced sediment yields (see Table 6). The erosion of forestry roads has also been identified as a cause of increased sediment loads in Scottish streams (Carling et al., 2001), although quantitative evidence is lacking, particularly in upland Britain.

A review by Moffat (1988) also found that erosion can increase as a result of afforestation in the uplands (due to ploughing, drainage, roadmaking etc) and following clearfelling. Little is known about long-term effects but the risk of soil erosion could be reduced by the replacement of ploughing by subsoiling and the control of drain gradients. At the time, there were few published rates of erosion for forestry in Britain, although some data were given for Welsh catchments both by Moffat (1988) and Soutar (1989), Table 6. Notably, the reported erosion rates from mature afforestation (353 – 1130 kg/ha) were much higher than the PSYCHIC estimates of sediment losses from forestry (30-45 kg/ha); it is very unlikely that data of this age would still be available for more detailed validation of the model estimates.

Table 6. Erosion rates from forestry in Wales (from Moffat et al. 1988 and Soutar, 1989).

Catchment	Management	Erosion rate (kg/ha/yr)[1]	Source
Llanbrynmair Catchment A	Unafforested	37 (ss)	Francis & Taylor (1989)
	After ploughing	90 (ss)	
Llanbrynmair Catchment B	Unafforested	7 (ss)	Francis & Taylor (1989)
	After ploughing	31 (ss)	
Hore	Mature afforestation	362 (total)	Leeks & Roberts (1987)
	After felling	571 (ss)	
Hafren	Mature afforestation	353 (ss)	Leeks & Roberts (1987
Tanyllwyth	Mature afforestation	505 (total)	Leeks & Roberts (1987
Cyff	Unafforested pasture	125 (total)	Leeks & Roberts (1987
Marchnant	Mature afforestation	1130	Newson (1980)

[1]ss: sediment yield as suspended sediment; total: sum of sediment yield as suspended sediment and bedload/settled sediment.

Nisbet et al. (2004; 2011) describe ways that woodland can reduce soil erosion as part of a whole-catchment approach to tackling sediment problems, although the authors admit that most of the supporting evidence for the effectiveness of this approach is based on overseas work. An exception is a study in the Nant Pontbren catchment in mid-Wales which found that woodland shelterbelts were very effective at intercepting and reducing surface run-off from agricultural land by enhancing soil infiltration, although sediment losses were not measured (Carroll et al., 2004).

In summary, it is widely recognised that soils under woodland are well protected from erosion. Measurements generally show lower sediment losses for watercourses draining woodland compared to other land, although most of the supporting evidence is based on overseas studies. When trees are felled the soil is left bare and erosion rates are likely to increase, although again there is little supporting evidence from the UK; most published UK data is now around 30 years old and unlikely to be available for validation. The Forestry Commission do not publish data on the area of woodland felled each year, although data is available on restocking (i.e. the replacement of trees on areas of woodland that have been felled) which could be inferred to be the same. In 2018-19 the area restocked in Wales was 1440 ha (approx. 1% of the total woodland area in Wales of 146,000 ha). Sediment losses from woodland runoff only represent an estimated 2% of the total for Wales (Figure 12), so even if losses were 10 times higher from felled areas, this would still only marginally increase the total loss. It may therefore be prudent to concentrate on improving the accuracy of some of the larger sources of sediment losses.

5.3.2 Road and urban runoff

Suspended sediment inputs to watercourses from roads and urban runoff across Wales were estimated using an Event Mean Concentration (EMC) methodology wherein calculations of average annual runoff are combined with event mean concentrations to estimate the annual load (Mitchell, 2005). A representative sediment EMC for all urban areas and roads was selected as 100 mg/l, based on measured inter-quartile ranges of 18.1 - 140.4 mg/l for industrial areas, 37.6 - 192.5 mg/l for residential areas and 62 - 396 mg/l for main roads (Mitchell et al., 2001). Subsequently, Leverett et al. (2013) in an assessment for Defra of the scale and impact of urban run-off commented that much of the data from Mitchell et al. (2001) was “relatively dated and….would benefit from an update”. They reported more recent mean total suspended sediment concentrations in urban run-off of 78.1 mg/l (for English towns) based on data from the first phase of the UK Chemicals Investigation Programme (CIP) monitoring, however this figure was not broken down by different urban land use. A third phase of CIP monitoring is currently being conducted and may provide useful supporting data on sediment (and other contaminant) concentrations in future.

The national suspended solids load from road and urban runoff was reported to be *c*.36,000 t or 7% of the total. Anthony et al (2012) noted that this was a first approximation and may be improved by further disaggregation of the urban land cover into component types with corresponding individual EMC values. We investigated the feasibility of doing this for the following additional urban land covers which could be sources of sediment/soil loss:

- **Construction sites**. Runoff flows over construction sites, picking up sediment and other pollutants; it then enters the storm water system and is discharged into local watercourses. The Office for National Statistics publishes annual statistics for Great Britain showing that the value of new construction work has continued to rise, reaching a peak in 2017; however data on the area of land being developed is not available (ONS, 2018). Moreover, we could find no data on typical sediment loads or runoff volumes from UK construction sites. It is therefore unlikely that construction runoff could be included as a separate land cover in an updated model. Further investigation may be useful to understand the value of OS Master Map, Earth Observation techniques and other such products to estimate land area and variance in annual data.

- **Gardens and allotments**. It has been reported that private gardens cover an area around 4 million ha in the UK (Thompson & Head, undated). Data on areas of allotments is published by individual English councils. However, the National Allotment Society states that in 1996 there were around 297,000 plots of 250 m^2 available (roughly 7,500 ha), although numbers have been declining. (https://www.allotment-garden.org/allotment-information/allotment-history/). We were not able to locate any information on soil erosion rates from gardens and allotments, although plentiful advice on erosion prevention is available for gardeners. As most gardens and allotment would be expected to be grassed or have plant cover most of the year, soil erosion losses would be expected to be low.

5.3.3 Other non-agricultural sources

Other non-agricultural land cover types in Wales include bare soil, mining activity and the military training ground at Mynydd Epynt, with an estimated area of 32,900 ha. Sediment losses from these areas were estimated using PSYCHIC (Davison et al., 2008) assuming that the areas could be represented as unfertilised rough grassland with no grazing animals. The national suspended solids load from other runoff was *c.*2,400 t (0.5% of the total).

Whilst it is reasonable to assume that sediment losses from bare ground and the military training area can be represented as diffuse losses similar to those from ungrazed rough grassland, there are strong grounds for treating sediment losses from mining sites as point sources, in a similar way to discharges from sewage treatment works (see Section 5.6).

5.4 River bank erosion

5.4.1 Bank erosion model

The contribution of river bank erosion to sediment losses was calculated using a preliminary national-scale model developed and described in detail by Anthony & Collins (2006). In this model, bank erosion is calculated as a function of the duration of excess bank shear stress using national environmental and hydrological datasets. The sediment input from bank erosion was estimated to be in the range 10 - 580 kg/ha of the contributing catchment area. The total load was *c.*141,000 t (27% of the total) making this a significant source of sediment in Wales and one where it would be worth investing effort to improve the estimates reported by Anthony et al. (2012).

Recently, models for estimating river bank erosion have been criticised by Janes et al. (2018) because they do not account for many of the factors which influence the severity of channel bank erosion such as the presence of bank vegetation, discharge and flow regime, lithology, channel confinement and anthropogenic influences; this limits their ability to simulate observed spatial and temporal variations in sediment generation. Janes et al. (2018) developed the sediment component of the catchment-scale model SHETRAN to incorporate key factors such as channel sinuosity and channel bank vegetation. The model was applied to the Eden catchment in north-west England, and validated using data derived from a GIS methodology. Estimated total catchment annual bank erosion was 617–4063 t/yr compared with observed values of 211–4426 t/yr. The authors suggest that the representation of bank erosion processes that have been applied to the SHETRAN model could also be applied to a number of existing physically based models, and this could be worth exploring further. However, when Anthony & Collins (2006) investigated the use of bank poaching and tree line data (from the River Habitat Survey database) this did not improve the correlation between modelled and measured erosion rates.

5.4.2 Model validation

Anthony & Collins (2006) used sediment fingerprinting data to validate their model of bank erosion rather than data on erosion rates collected using other more direct methods. This was because most direct erosion measurement or monitoring techniques, including erosion pin deployment, are used to target severely eroding sites within a catchment. Channel bank erosion is highly site-specific so focusing on a limited number of observation sites, means that erosion pin studies are potentially biased towards higher erosion rates. However, the sediment fingerprinting method has itself been criticised by Smith & Blake (2014) who re-examined some of the assumptions underpinning the methodology and concluded that estimates of source contributions in many such studies may contain significant unquantified errors. Similarly, Walling (2013) highlighted the need to direct increased attention to the uncertainty associated with the results of such studies.

Table 7: Estimated channel bank erosion rates, determined using sediment source fingerprinting data in conjunction with suspended sediment load monitoring, for selected bank erosion calibration catchments (Anthony & Collins, 2006).

ID	Area (km^2)	Grid Reference	Location	Bank Erosion Yield Rate (kg/ha)	Bank Length (km)	Bank Erosion Rate (kg/km)
1	21	ST903274	Wiltshire	11.0	20.5	1,127
2	46	SX935076	Devon	29.0	65.2	2,046
3	54	SS975135	Devon	15.6	69.8	1,207
4	55	SO441553	Herefordshire	56.8	49.2	6,350
5	65	SS959223	Devon	31.5	73.0	2,805
6	69	SO401285	West Midlands	152.4	99.5	10,568
7	81	SU134559	Wiltshire	8.0	50.6	1,281
8	89	SU133558	Wiltshire	2.0	66.0	270
9	93	SO560193	West Midlands	80.4	111.6	6,700
10	216	SU098307	Wiltshire	20.0	105.9	4,079
11	109	SU161264	Wiltshire	4.8	45.6	1,147
12	128	SS927258	Somerset	11.2	176.5	812
13	166	SU634747	Berkshire	0.3	66.3	75
14	183	SY913876	Dorset	8.3	115.8	1,312
15	231	SJ648232	Shropshire	10.5	261.6	927
16	234	SU485676	Berkshire	0.5	40.9	286
17	276	SX946992	Devon	38.4	324.6	3,265
18	437	SY891867	Dorset	29.1	310.7	4,093
19	484	SE504570	North Yorkshire	47.4	652.8	3,514
20	818	SE487436	North Yorkshire	59.3	1,062.1	4,567
21	914	SE396671	North Yorkshire	178.3	1,408.6	11,569
22	3315	SE571553	North Yorkshire	138.8	4,577.5	10,052

Data from: Collins et al. (1997a); Walling et al. (1999b); Walling & Collins (2005); Walling et al. (2006; 2008).

Anthony & Collins (2006) collated measurements of net bank erosion for 22 catchments clustered on the Rivers Avon, Exe, Ouse and Wye (there were none from Wales) from the literature published up to 2006 (Table 7). Measured bank erosion rates were in the range <1 to 178 kg/ha and accounted for 1 to 37% of the modelled total sediment yields for the validation catchments; the proportion of the variance explained by the model was modest (r^2=59%). As bank erosion rates can vary considerably particularly in areas of highest risk, Anthony & Collins (2006) stressed that there was substantial uncertainty in the modelled bank erosion rates in these areas.

Table 8. Summary of identified papers published since 2006 which may contain additional data for determination of channel bank erosion rates in England and Wales.

Catchments/rivers studied	Paper title	Reference and notes
Frome & Piddle (Dorset)	Sources of fine sediment recovered from the channel bed of lowland groundwater-fed catchments in the UK	Collins & Walling (2007a)
Tern (Shropshire), Pang & Lambourne (Berkshire)	The storage and provenance of fine sediment on the channel bed of two contrasting lowland permeable catchments, UK	Collins & Walling (2007b)
Camel, Fal, Lynher, Plym, Tamar & Tavy (south west England)	A preliminary investigation of the efficacy of riparian fencing schemes for reducing contributions from eroding channel banks to the siltation of salmonid spawning gravels across the south west UK	Collins et al. (2010a)
Somerset levels	Apportioning catchment scale sediment sources using a modified composite fingerprinting technique incorporating property weightings and prior information	Collins et al. (2010b)
Wensum (Norfolk)	Contemporary fine-grained bed sediment sources across the River Wensum Demonstration Test Catchment, UK	Collins et al. (2013)
Wensum (Norfolk)	High-temporal resolution fluvial sediment source fingerprinting with uncertainty: a Bayesian approach	Cooper et al. (2015)
Torridge (Devon), Axe (Somerset), Arrow (Worcs.), Waver (Cumbria), Rye (Yorkshire), Wensum (Norfolk).	The application of sediment source fingerprinting techniques to river floodplain cores, to examine recent changes in sediment sources in selected UK river basins	Haley (2010).
	Identifying causes and controls of river bank erosion in a British upland catchment	Henshaw et al. (2013). Erosion pin study.
Nene (eastern England)	Exploring fine sediment dynamics and the uncertainties associated with sediment fingerprinting in the Nene river basin, UK	Pulley (2014). See also: Pulley et al. (2015a;b), Pulley et al., (2017a;b), Pulley et al (2018)
Tamar (Devon/Cornwall)	Sediment fingerprinting in agricultural catchments: A critical re-examination of source discrimination and data corrections	Smith & Blake (2014)
Lugg (Herefordshire)	Provenance and Transfer of Fine Sediment in the Lugg Catchment, Herefordshire, UK	Stopps (2018).
	The catchment sediment budget as a management tool	Walling & Collins (2008).

We investigated whether there was any more recent data which could be used to revisit and revalidate the Anthony & Collins (2006) model, and whether there was any specific data for Welsh rivers. To this end, a preliminary search of the literature was undertaken in Google Scholar (for the years 2007-2019). The papers identified from this literature search (Table 8) could contain potentially useful data on bank erosion rates and warrant further investigation.

Anthony & Collins (2006) commented that the estimated contribution of channel bank erosion to the total suspended sediment load represented a preliminary investigation, and they highlighted areas where future work could help to refine the model. They suggested that spatial coverage of sediment source fingerprinting information needs improving to support the development and validation of a more robust model. To some extent this has happened, as evidenced by the more recent sediment fingerprinting studies summarised in Table 8, although none/few of these have been conducted in Wales. It would therefore still be valuable to obtain data for catchment typologies representing Wales. The significant contribution of channel bank erosion to the total sediment input in Wales (27%, Figure 12) further underpins the need for additional work.

5.5 Point sources

5.5.1 Sewage treatment works

Sewage effluent and industrial discharges to each WFD river catchment were calculated using a discharge consents database developed for the Environment Agency (EA) by Kelly et al. (2006). The database identified 330 effluent discharges within Wales of which 270 were sewage treatment works (STW) outfalls, the remainder being industrial and fisheries discharges. The EA database did not include measurements of suspended solids in the effluent. The Register of Consents was therefore used to identify the maximum consented value for suspended solids for each discharge. These ranged from <1 to 800 mg/l and were provided for 304 of the consented discharges in Wales; missing data were replaced by the average consented value of 62 mg/l.

Based on previous findings, actual concentrations were expected to be less than the maximum consented values. Therefore a relationship between consented and measured average suspended solids concentrations developed by Roberts & Williams (1997) was used to estimate the total suspended solids load from every effluent discharge. The national total suspended solids load from sewage effluent discharges was 3,558 tonnes.

Recent (2016) data on consented discharges are available from the Lle Geo-Portal for Wales (Lle, 2019a). Suspended solid inputs to rivers from STWs were fully updated in the creation of the SEPARATE database (Zhang et al., 2014).

5.5.2 Mining discharges

Wales has a long history of coal mining, although the last deep mine in Wales, Tower Colliery, closed in 2008. There are also non-coal (metal ore and mineral) mines although many are now abandoned and the land has been reclaimed. The exact number of mines in Britain is not known, although work in Wales, the South West and Northumbria has identified over 3,700 sites (Environment Agency Wales, 2002).

Pollution from mines arises from discharges from shafts and adits as well as from surface activities such as mineral processing, tailings and waste disposal (Figure 14), though not all mines will cause serious pollution. The pollution is often spread over a wide area and many small individual discharges can add up to create a significant diffuse source, suggesting that representing mines as rough grassland (see Section 5.4.3) is probably not very accurate.

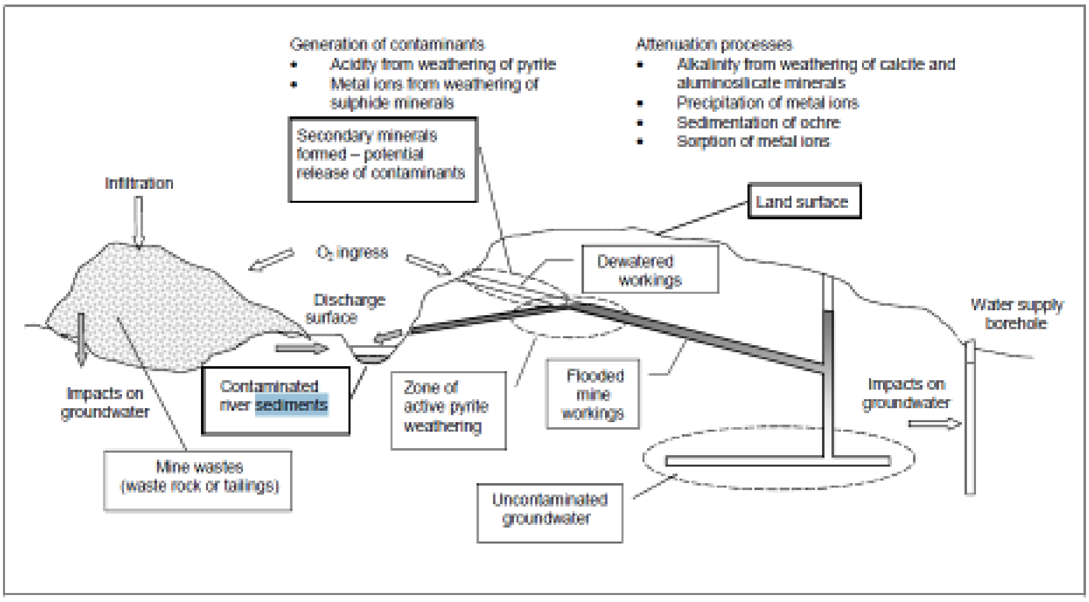


Figure 14: Sources and pathways of mine pollution (from Younger *et al.* 2002)

Over 100 priority abandoned coal mining sites have been identified in the coalfield areas of Britain by the environment agencies and the Coal Authority. Seven hundred monitoring sites record water level and quality across the coalfields, and around 60 water treatment facilities have been built in the UK to prevent pollution (CL:AIRE, 2018), Figure 15. Despite this, 18 river waterbodies (231 km) were deemed to be 'at risk' from abandoned non-coal mines in West Wales (Figure 16a; Johnston et al., 2008).

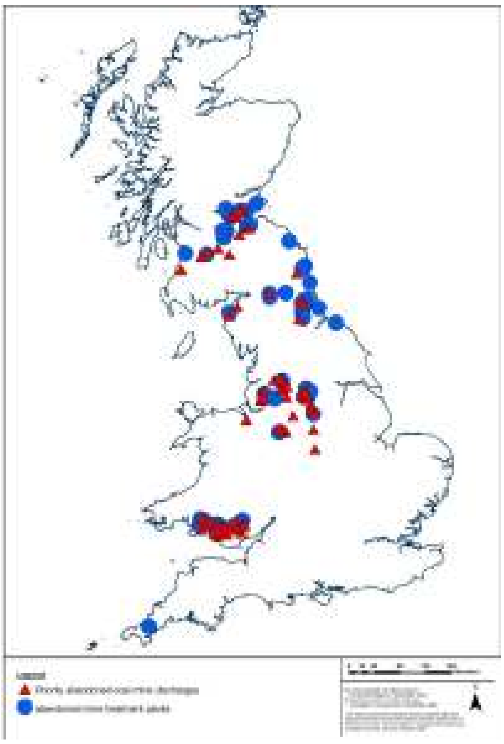
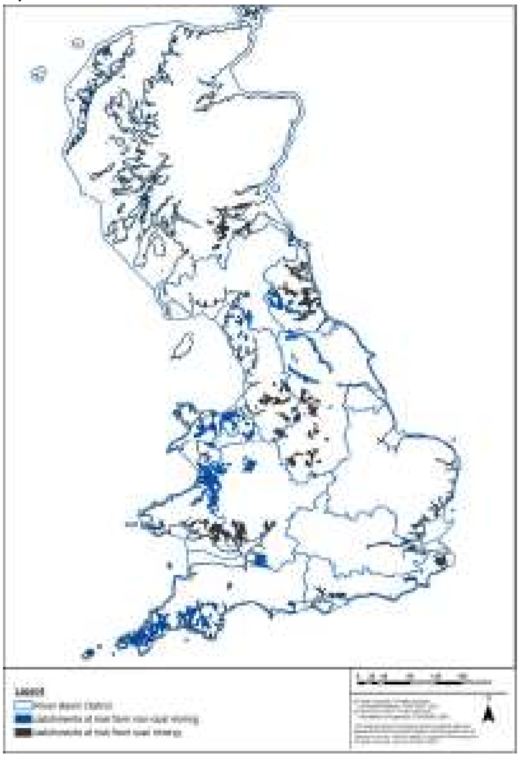


Figure 15. Minewater treatment plants and priority coal mine discharges in Britain (from Johnston et al., 2008)

Monitoring has shown that some abandoned metal mines are significant contributors to heavy metal pollution in our rivers and seas. Johnston et al. (2008) reported that in West Wales 87 river waterbodies (687 km) were 'at risk' from abandoned non-coal mines (Figure 16a); this assessment was subsequently updated by CL:AIRE (2014), drawing on data from Mayes et al. (2010) as illustrated in Figure 17b.

There has been some research published on runoff amounts and quality from open and abandoned mines and coal fields, including from reclaimed mined land, although in Wales much of this has focussed on contaminants such as heavy metals rather than on sediment. For example, Byrne et al. (2012) investigated metal concentrations in the Afon Twymyn river draining an abandoned metal mine in central Wales, whilst Bearcock et al (2010) looked at the hydrochemistry of the Frongoch lead/zinc mine, also in mid Wales. A comprehensive government-funded inventory of pollution discharges across England and Wales identified 338 discharges from 4923 abandoned metal mines; metal, but not sediment, concentrations in the discharges were reported together with mean flow rates (Mayes et al., 2010). More recently, Mayes et al. (2015) undertook a national scale GIS-based analysis to identify potential diffuse metal mining pollution sources across England and Wales, using datasets that included national topographic layers (elevation, slope), Landmark waste rock data (from national Ordnance Survey mapping) and GBASE stream sediment metal analysis data.

a)



b)

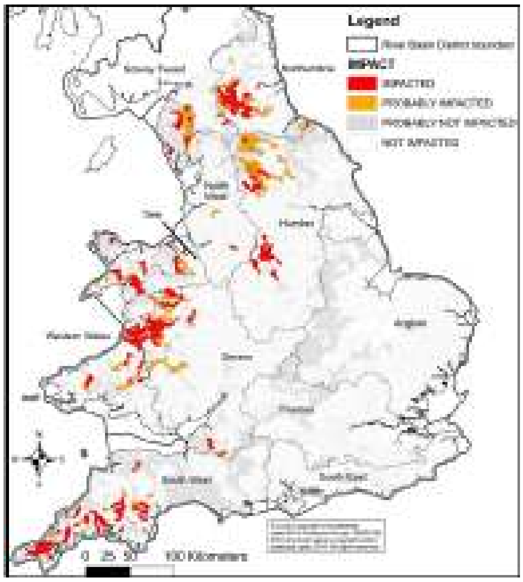


Figure 16. a) River Basin Districts and river catchments at risk from abandoned mine pollution (from Johnston et al., 2008); b) Categorisation of water bodies in England and Wales according to abandoned metal mine impacts (CL:AIRE, 2014)

The only suspended sediment concentrations for Welsh streams impacted by coal and metal mine drainage that we were able to locate were measured by Auladell Mestre (2010) as shown in Table 9.

Some part of the input from mines would be taken account of the by PSYCHIC estimates of losses from non-agricultural land. However if there are specifically high inputs from waste heaps and discharges from below ground, then the inputs are likely to be underestimated in the current model. It should be possible to obtain data from the inventory of abandoned non-coal mine sites on the locations of discharges from metal mines, together with flow rate data (Mayes et al., 2010; 2015). Similar data on the locations of coal mines should be available from the Environment Agency and/or the Coal Authority (see https://www.gov.uk/guidance/using-coal-mining-information) although it is not clear if flow rate data is available. This information could be added into the model to define mines as additional point sources and treated in a similar way to STWs, although there is a lack of data on sediment concentrations in mine drainage water.

Table 9. Suspended sediment concentrations in streams impacted by coal and metal mine drainage, measured upstream and downstream of the mine discharge (2006/2007).

Mine type	Stream name	Suspended sediment (ppm) [upstream/downstream]
Metal	Dyliffe	1.45/0.97
	Frongoch	1.69/1.31
	Cwm Ystwyth	1.70/0.49
	Cwm Symlog	2.09/1.28
Coal	Aberbeiden	3.18/7.35
	NantyFyllon	0.54/2.44
	Clyne	4.74/6.71
	Cwn Gross	3.61/7.69

5.5.3 Reclaimed mining land

Haigh (1992) reported that large tracts of land in South Wales, officially listed as 'reclaimed' from former mineral workings, were in very poor condition, with problems including gullying, accelerated runoff, poor vegetation cover, erosion and poor soil structure. Mine spoils contain a large proportion of water unstable mudstones and shales. These break down on wetting, flood the soil with fine particles and raise the soil density thus impeding water infiltration and root penetration. In rainfall simulation tests on surface coalmine spoils near Blaenavon, Wales, Haigh & Sansom (1999) found that poorly vegetated plots converted less rainfall to runoff and yielded 25% of the sediment of unvegetated plots. Similar tests at Doncaster, England, found that soil losses for the same rain event were more than four times greater on the unvegetated sites. Haigh & Kilmartin (2015) reviewed the findings from these studies of reclaimed mining land, concluding that the effect of the runoff on runoff river water quality in southeast Wales is not great, although it may lead to slightly higher flood peaks and elevated concentrations of some metals. The impact on river water sediment concentrations was not discussed by Haigh & Kilmartin (2015) and we were not able to locate any other data on sediment concentrations in runoff water from reclaimed mining sites.

The International Centre for Regional Regeneration & Development Studies (2003) gives the area of former coalfields in South Wales as 2,500 km^2 (250,000 ha) but does not state how much has been reclaimed. However, the Soilscapes of Wales (Table 2 and Figure 1) includes restored soils (mostly from quarry & opencast spoil) which comprise 0.7% of the total land area (*c*.14,000 ha).

It is difficult to see how reclaimed mining land could be usefully included as a separate source in the Anthony et al. (2012) model because of the lack of data on sediment concentrations; however, some/all of this land will already be accounted for in the sediment apportionment estimates as losses from diffuse agricultural or non-agricultural land.

5.5.4 Landfill sites

Sediment may be generated during the construction of a new landfill site when the surface cover is removed or slope gradients are altered. Engineering techniques are available to contain or filter sediment laden water, and these should be implemented by contractors in accordance with the EU Landfill Directive (EEC/1999/31/EC). Guidance issued by Natural Resources Wales (NRW, 2014) provides generic guidance on the best practice for the design and management of landfill sites including the ongoing monitoring of leachate, groundwater and surface water to comply with Environmental Permits. There has been very little information published on sediment losses from landfill; most research has focussed on leachate and pollutants that may reach groundwater (e.g. Thornton et al., 2000a;b).

If there were a requirement to account for sediment losses from landfill sites, then the locations of current and historic sites are available from the Welsh Government (2019) and the Lle Geo-Portal (Lle, 2019b), respectively.

5.5.5 Footpath erosion

Footpath erosion is a matter of concern in various parts of Wales including Snowdonia, the Brecon Beacons and the Wales Coast Path where recreational pressure is increasing (Figure 17).

Figure 17. Footpath erosion in the central Brecon Beacons (National Trust, 2018)

There has been little research quantifying the amount of soil loss from footpath erosion in the UK. A broad-scale survey of 485 sites on 25 paths in the Lake District (Coleman, 1981) found that footpath erosion (measured as path width, extent of bare ground or maximum depth) increased with the square root of the slope angle and the square of the recreation pressure. These two variables interacted with each other, while other factors, such as vegetation type, soil type and topographic position, also influenced rate of erosion. A threshold slope angle of 15–17° separated actively eroding from stable slopes. Rodway-Dyer & Walling (2010) assessed the physical aspects of soil erosion using caesium-137 (^{137}Cs) on-sites within Dartmoor National Park and the South West Coast Path in south-west England. Longer-term erosion rates (*c*. 40 years) estimates showed that the mean soil loss for footpaths was 1.41 kg/m^2/yr whereas the combined 'off path' soil loss was 0.79 kg/m^2/yr. Recreational pressure was shown to increase erosion in the long-term, as greater soil erosion occurred on the paths, especially where there was higher visitor pressure. In the US, Harden (1992) undertook preliminary work towards incorporating the runoff and erosion-initiating effects of rural roads and paths in watershed models, although to our knowledge this has not been attempted in the UK.

There are an estimated 20,700 miles (33,000 km) of public rights of ways in Wales including footpaths, bridleways and other types of byways (Natural Resources Wales, 2019). It would be possible to use Ordnance Survey data to map these, but an estimate would be needed of the location of those that are actively eroding to be able to derive spatial data on sediment losses to watercourses. In Snowdonia, a recent survey reported 2.5 miles of critically eroded paths that are in need of urgent repair (National Trust, 2019), and in the Brecon Beacons, erosion of the 70 km of footpaths is an ongoing problem (National Trust, 2018). As a rough 'worst case' estimate, an assumption could be made that 100km of Welsh footpaths (at an assumed width of 5m) are eroding at a rate of 1.4 kg/m^2/yr (Rodway-Dyer & Walling, 2010). The total soil loss would be *c*.700 t/yr for Wales as a whole, which is very small in comparison with the other sources of sediment input to river systems considered by Anthony et al. (2012), Figure 12, although footpath erosion could be an important localised source of sediment in watercourses.

5.5.6 Damaged road verges

Losses of particulates from roads *per se* (as sourced from vehicles etc.) are addressed using the EMC approach (see Section 5.3.2). However, soil loss from rural roadside verges and tracks can also be a significant source of sediment into watercourses due to damage from vehicle and livestock traffic, and this is not accounted for in the current model. In addition, rural roads can act as a conduit to flow providing enhanced connectivity between the sediment source (e.g. a field) and the receiving water.

Collins et al. (2010c) used a source fingerprinting technique to provide information on the relative importance of sediment sources in a grassland catchment in Cumbria and found that damaged road verges contributed around 6% (±1%) to the total. Similar work for the River Glaven priority catchment in Eastern England showed that the relative contribution from damaged road verges was 2 - 50% (Collins et al., 2010d), whilst Collins et al (2017) estimated that 35% (uncertainty range 0–100%) of the sediment associated organic matter in the River Axe Special Area of Conservation was from damaged road verges.

Whilst there is no data specifically for Wales, similar processes will operate and erosion of verges may be a significant sediment source in some Welsh catchments. More work is required to establish if and how this source could be included in any future version of the Anthony et al. (2012) model.

5.6 Validation of model outputs

5.6.1 Validation of model estimates for Wales

The study by Anthony et al. (2012) included a validation exercise to assess whether the outputs from the modelling framework were able to reproduce the measured pollutant loads and spatial pattern of emissions in Wales. Modelled sediment loads were compared to measured sediment loads derived from the Environment Agency General Quality Assessment (GQA) and Harmonised Monitoring Scheme (HMS) networks and measured rivers flows for the purpose of reporting maritime inputs under the Oslo-Paris Commission (OSPAR-COM) (Littlewood and Marsh, 2005; Jarvis et al., 1997). Comparisons were made for five monitoring contributing areas (E23 to E27) located entirely within the policy boundary of Wales, and broadly covering the Western wales and Dee River Basin Districts (RBDs). The comparison used measured sediment loads for the period 2004 to 2007. Measured sediment loads were in the range 70 to 470 kg/ha and modelled were in the range 160 to 320 kg/ha (Figure 18). The largest absolute difference between modelled and measured was for monitoring area E23, draining the catchments of the rivers Usk and Rhymney. This area had the lowest contributions from agricultural sources and so it was possible that the differences were in estimates of sewage and other sediment inputs to the river systems.

Sediment concentrations were predicted for a larger number of inland GQA monitoring sites, by diluting modelled loads into modelled flows, and applying an empirical in-river sediment retention model. It was not possible to predict sediment concentrations with any skill (Figure 19). Modelled flow weighted average concentrations generally exceeded measured concentrations, notably in upland areas of extensive rough grazing where measured concentrations were low. This implied that modelled sediment loads from the rough grazing areas of Wales were over-estimated.

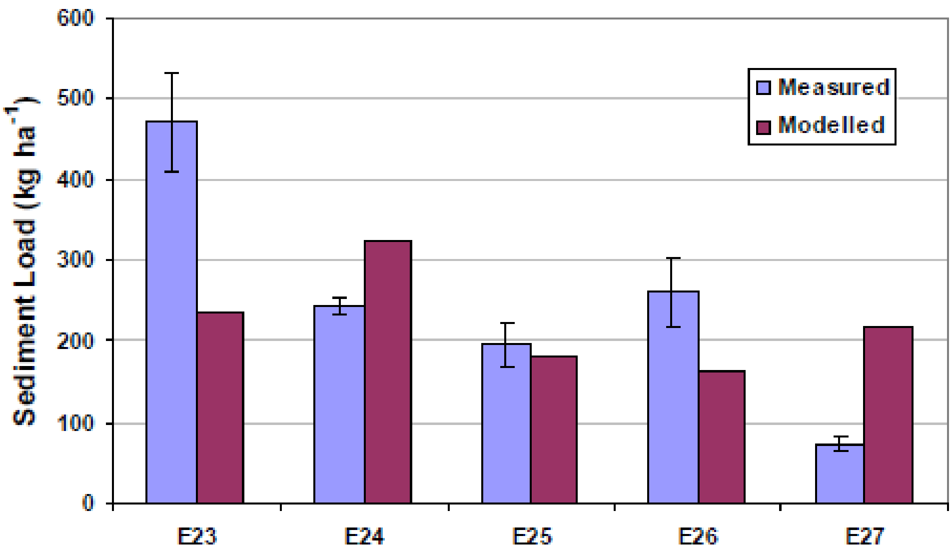


Figure 18. Comparison of modelled and measured sediment loads exported from the OSPAR-COM monitoring catchments located in Wales (Anthony et al., 2012).

a) b)

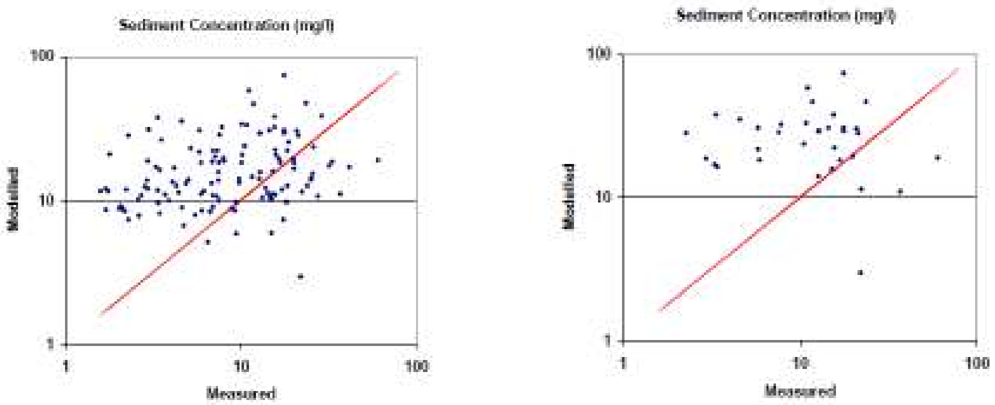


Figure 19. Comparison of measured average suspended solids concentrations and modelled average fine sediment concentrations for selected Environment Agency monitoring sites on rivers in Wales a) all catchments, b) only includes catchments where agriculture accounts for >75% of the total sediment load (Anthony et al., 2012).

Whilst the modelling for Wales does not seem to be particularly satisfactory, when data for English regions are included in the validation there is some consistency over a wide range of soil/climate conditions and relative contributions from point and diffuse sources (Figure 20), providing more confidence in the model outputs.

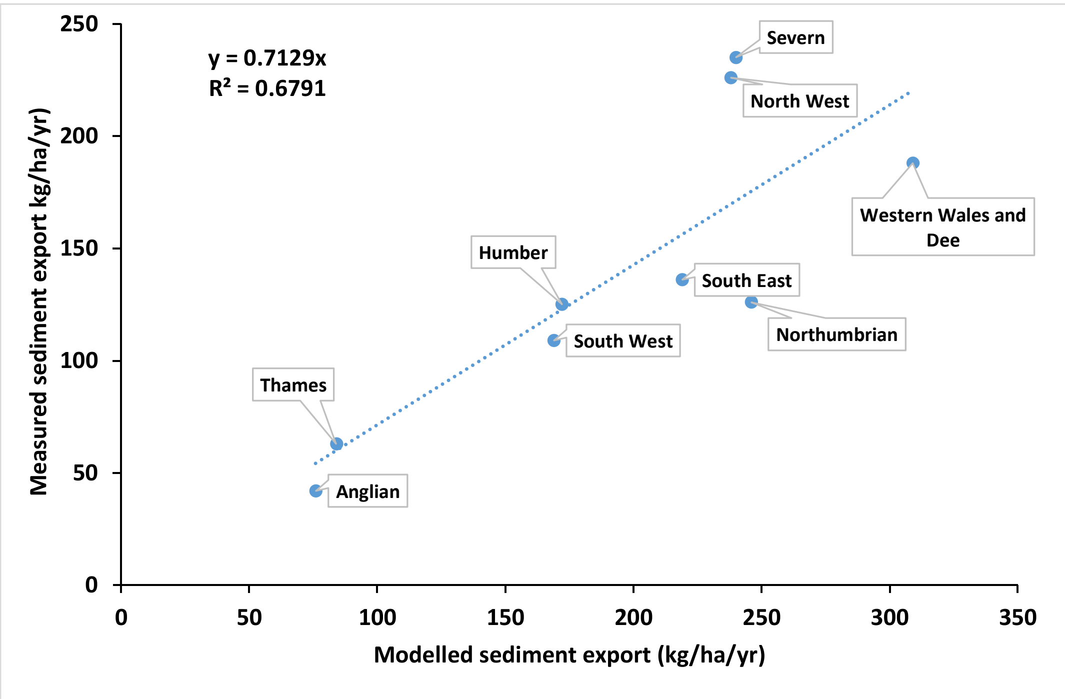


Figure 20. Measured and modelled annual total export of suspended sediment from PARCOM sample regions aggregated to approximate WFD basins for England and Wales (Based on data from Anthony & Collins, 2006).

5.6.2 Validation using water quality data

As Anthony & Collins (2006) noted, a major constraint in the source apportionment modelling studies was the lack of reliable suspended sediment data for rivers in England and Wales against which to validate the model predictions.

Suspended sediment data are available from some national water quality monitoring programmes such as the Harmonised Monitoring Scheme (HMS) and the General Quality Assessment (GQA) which has approximately 7000 monitoring sites across England and Wales. In the main these are operated by the regulatory agencies and are driven by legislation at national and European level, with a focus either on pollution prevention and control, or the maintenance of quality standards for particular water uses. This is reflected in the suites of measurements of the various schemes and the location of most monitoring sites in river reaches away from the headwaters.

National schemes provide better spatial coverage at national scale for larger rivers, but the sampling strategies do not focus on suspended sediment fluxes. Consequently, no special sediment sampling equipment is used and since the sampling is infrequent, it can fail to capture flood events when most suspended sediment transport occurs. As a result, these data provide limited information on the sediment response of individual rivers and estimates of suspended sediment fluxes obtained on the basis of such data are likely to underestimate the actual values. For large catchments it can be difficult to attribute signals in the data to specific sources, and soil may only be one of many possible candidates.

Nevertheless the data can be used to provide a broad overview of sediment concentrations in surface waters highlighting areas of concern (see for example Figure 21). Although additional work is required before this information can be interpreted directly in relation to specific land and soil management practices, the data could be used for targeted monitoring of problem areas (Emmett et al., 2006).

a) b)

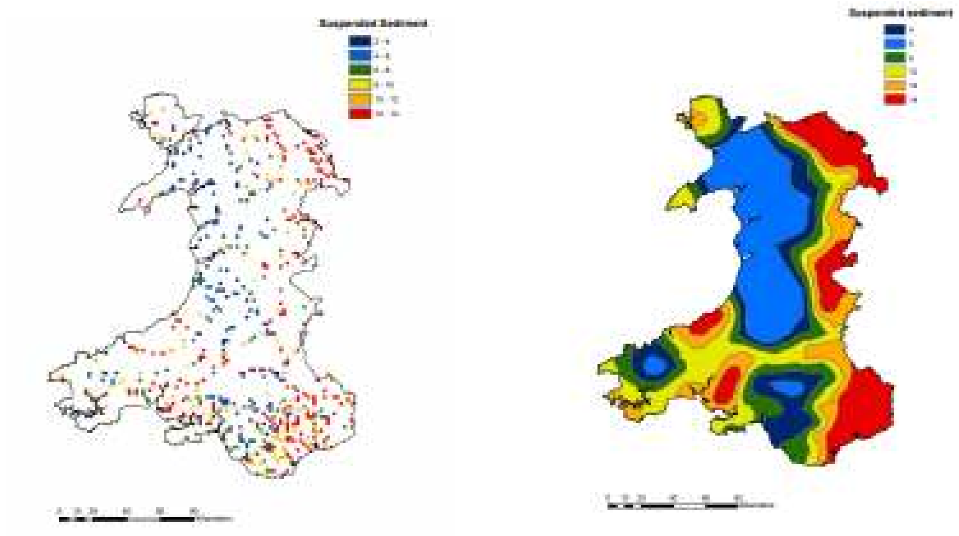


Figure 21. Maps of a) suspended sediment concentrations at individual sampling points in Welsh rivers and b) suspended sediment concentrations interpolated from the point data using geostatistics (Source: Emmett et al., 2006).

5.6.3 Comparison with other methods of source apportionment/sediment budgeting

Another potential means of model validation is to determine whether the source apportionment assessments are similar to those derived by other sediment budgeting methods such as sediment fingerprinting studies. As well as providing data to validate the modelled sediment losses via bank erosion (see Section 5.4), these could potentially be used a 'sense check' for the overall model outputs of the contributions of the various different sources to sediment loads.

Since sediment fingerprinting studies can only be undertaken for single or a few catchments, these studies are generally seen as tools to support catchment management (e.g. Walling & Collins, 2008). In order to use data from such studies for validation purposes, the outcomes from the modelling would need to be presented on a catchment basis (which the model is designed to do; see Davison et al., 2008), and validated against individual catchment data.

This approach was adopted by Anthony & Collins (2006) who compared modelled sediment losses with those obtained from sediment fingerprinting studies for 23 catchments in England (Figure 22). With the exception of two outlier under-predictions, there was a positive correlation between the modelled and measured estimates of total sediment yield (Figure 22a). The under prediction on two outlier catchments (on the River Exe) was due to an under-estimate of the yield from diffuse agricultural sources, whilst over- estimates (at locations on the rivers Avon and Wye) were probably due to poor characterisation of the presence and efficiency of agricultural drainage systems. Overall, the modelled and measured proportions of the diffuse sediment yield from cultivated land and pasture were in general agreement (Figure 22 c,d).

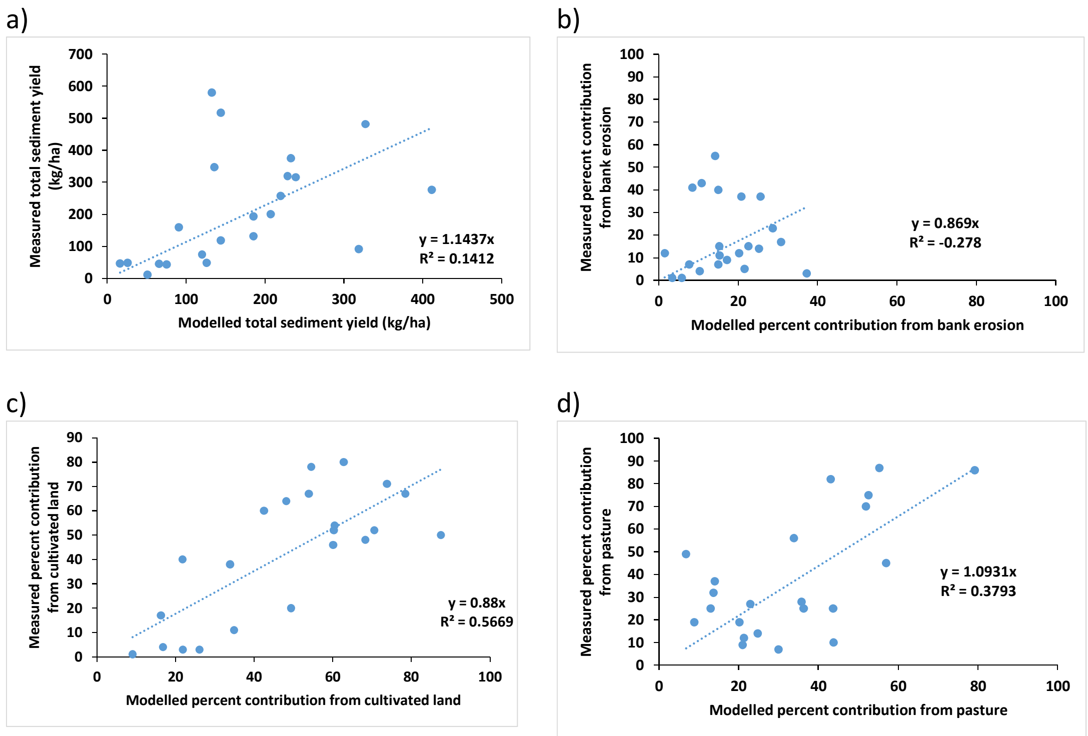


Figure 22. Measured and modelled a) total suspended sediment yields and percent contributions from b) bank erosion, c) cultivated land and c) pasture for study catchments used to provide source apportionment data on the basis of sediment source fingerprinting. (Based on data from Anthony & Collins, 2006).

As shown previously (Table 7) there have been relatively few recent sediment fingerprinting studies for Welsh river catchments so there is not currently sufficient data available for comprehensive validation of source apportionment models in Wales.

Moreover, there are potential issues with the sediment fingerprinting approach itself. For example, Parsons & Foster (2011) concluded that rates of soil erosion that are based upon the use of the ^{137}Cs technique are not reliable, and that the approach cannot be used to provide reliable information about rates of soil erosion. More recently, Evans et al. (2017) also suggested that fingerprinting (using ^{137}Cs) needs to be validated more rigorously and conversion models re-calibrated. They claimed that erosion rates based on this method "may well overstate the severity of the problem in lowland Britain and, therefore, are not a reliable indicator of water erosion rates".

Nevertheless, earlier findings from Walling & Collins (2005) provide some support for the relative importance of the different sources identified by Anthony et al. (2012). In an attempt to understand the suspended sediment budgets of British catchments, these authors collated data from source fingerprinting investigations and synthesised the findings from 48 catchments (including several in Wales). The results showed that, in the majority of cases, the relative contribution from forested areas was low (generally <10%), reflecting both the relatively small proportion of most catchments occupied by forest and woodland, and the role of trees in limiting runoff and erosion (see Section 3.1). This supports the modelled estimates from Anthony et al. (2012) for Wales where wood and forest accounted for 2% of total sediments losses (Figure 12). Walling & Collins (2005) reported that channel bank erosion accounted for 1-55% of the total sediment budget, reflecting the diversity of factors that influence losses from this source (see Section 5.4).

5.6.4 Observations of soil damage and erosion issues

Information and observations on the extent and consequences of localised soil damage could be important in terms of understanding the areas at risk of runoff and soil erosion.

Newell-Price et al. (2013) reported that only 8 to 12% of grass fields across England and Wales were in poor condition based on the Landcare (VSA) and Peerlkamp (ST) structural scoring systems. However, visual assessment of surface conditions has tended to report wider damage. For example, Anthony et al. (2012) reported observations of soil damage associated with soil compaction and structural degradation from 600 Welsh farmers. Soil compaction due to heavy machinery on both arable and grassland land (19%) and grassland poaching by livestock (43%) were the most cited observations of soil damage, but visual damage tends to be confined to only parts of a farm and information on proportion of land affected was not collected (Table 10). were the least degraded (Palmer & Smith, 2013).

Holman et al. (2003) combined a 'spade-test' with a visual scoring of features such as poached areas and standing water in surveys of the Severn, Yorkshire Ouse, Uck and Bourne catchments. Severe (2%), high (34%) and moderate (41%) degradation was observed widely on grassland fields (n = 121). Similarly, Palmer & Smith (2013) recorded severe (1%), high (9%) and moderate (67%) degradation on permanent grassland fields surveyed across south-west England (n = 1,154). Severe levels of degradation were visually associated with runoff generation across whole fields, whilst moderate levels of degradation led only to localised patches of enhanced runoff. These authors also reported a strong correlation between levels of degradation and soil type. Sandy soils and groundwater gley soils were most frequently degraded, whilst shallow soils over rock and under drained clay soils were the least degraded.

Table 10. Proportion of grassland farmers observing signs of soil damage on grass fields within the last 3 years, by farm type (Wales Farm Practice Survey, Anthony et al., 2012).

Soil Damage Type	Farm Type		
	Upland Cattle & Sheep	Lowland Cattle & Sheep	Dairy
Poaching of grassland	35±6	39±7	57±7
Erosion of stream banks	4±2	8±4	8±4
Discoloured water entering ditches or streams	6±3	6±3	14±4
Sediment deposited in ditches or streams	7±3	5±3	9±4
Gullies and rills formed in fields	3±2	3±2	3±2
Compaction or rutting due to heavy machinery	11±4	13±5	27±6
None	55±6	50±7	30±6

±Ninety-five percent confidence intervals

It should be noted that surveys of soil compaction and grassland degradation have found only weak correlations between measured status and the predicted risk of soil compaction or poaching based on national mapping of soil properties (Hallett et al., 2016 ; Cranfield University, 2007). The mapping predicts the intrinsic risk of compaction rather than its occurrence, which depends on the history of field management, and it is feasible that appropriate management such as controlled traffic, lower stocking densities or delayed turn-out dates will mitigate risk on the most vulnerable soils. The use of representative average rather than site specific soil horizon PSD and OC, and pedo-transfer functions in place of measurements of soil water content, also contributes to uncertainty in the mapping of intrinsic risk *per se*.

6 FUTURE RESEARCH REQUIREMENTS

6.1 Improvements to model outputs

Some areas where the data used to populate the modelling framework for source apportionment of sediment losses to water courses could be refined and validation of the model outputs improved are itemised below.

Climate data. The climate data used to produce the previous model predictions were for 1961-1990. There is clear evidence that rainfall patterns in Wales have changed between 1961 and 2006, and this may have an effect on soil erosion. The impact on sediment losses could be explored by running the model using more recent climate data for Wales, and/or projections of future rainfall amounts and patterns.

Sediment retention. Landscape and in-channel sediment retention are still rather crudely represented in the PSYCHIC model of diffuse sediment losses from agriculture. There is an opportunity to explore the possibility of modifying the landscape connectivity index using data on landscape features which are known to control retention such as riparian areas and hedgerows, although this is likely to require complex modelling and large amounts of data. An alternative approach exploring connectivity is outlined in Section 6.3.1.

Rough grazing. Modelled sediment concentrations generally exceeded measured concentrations, notably in upland areas of extensive rough grazing, implying that modelled loads from rough grazing areas were over-estimated. Rough grazing represents 24% of the agricultural land area of Wales (and 18% of estimated total sediment losses) so it is important that estimates of sediment losses are robust. More work is required to establish why the estimated sediment losses from this important source are being overestimated and what could be done to improve the modelled outputs.

Land drainage. The under-drainage of arable and grassland is a significant risk factor for the delivery of sediments to watercourses, and will be affected by the condition of the land drains themselves. This is not factored into the current modelling approach but it might be possible to use survey or observational data (e.g. from future WFPS) to inform future adjustments to the model to account for drain condition, as well as their simple presence/absence.

Urban runoff. The data used in the model for sediment concentrations in urban runoff was published in 2001 and is now relatively dated. It may be possible to obtain more recent data from the UK Chemicals Investigation Programme (CIP) monitoring programme to populate the model or provide supporting information.

Bank erosion. The modelled contribution of (river) channel bank erosion to the total suspended sediment load was a preliminary estimate. Wider spatial coverage of sediment source fingerprinting information is needed to improve the development and validation of a more robust model. Some such studies have been identified, although few of these have been conducted in Wales. There is still therefore a requirement to obtain data for catchment typologies representing Wales (see Section 6.3.2).

Sewage treatment works. More recent (2016) data on consented discharges are now available which could be used to assess whether there have been any changes in sediment loads from STWs since the previous modelling work for Wales was undertaken. Suspended solid inputs to rivers from STWs were fully updated during the creation of the SEPARATE database.

Mining discharges. Whilst the previous modelling work would have accounted for some part of the inputs from mines, specific high inputs from spoil heaps and below-ground discharges were not considered, although they could be added to the model as additional point sources and treated in a similar way to STWs (see Section 6.3.2).

6.2 Novel technologies

In 2019 a report was produced by BGS and CEH as part of the ERAMMP programme (Tye & Robinson, 2019). This report reviewed "new and existing technologies that can be used for capturing evidence on the extent of soil erosion (with a particular emphasis on monitoring of landslips, peatland and bankside erosion)". The tasks it addressed were to:

- Assess the range of techniques available to measure soil erosion and describe the scale over which they may operate along with their major advantages and disadvantages.
- Provide some estimates of costs associated with selected methods.
- Identify the extent that new technologies can be used within a soil erosion monitoring programme.

Within the report the authors looked at both direct and indirect physical measurements, as well as remote sending technologies, and provided costings for the different methods. For example, the cost of a ^{137}Cs survey of fields for soil and tillage erosion was estimated at £2.4 million.

One outcome from the report was that Welsh Government and CEH have now agreed on a programme to collect general information on soil condition as part of the next ERAMMP survey. In the coming year, CEH and BGS staff will use aerial photography and remote sensing to assess the extent of large erosion features within each of the *c.* 200 one-kilometre-squares distributed across Wales, and field staff will also make a visual assessment for individual fields when they are visited for botanical survey.

However, we consider that there are opportunities to complement the planned work described above by adopting the alternative approaches outlined in the next section.

6.3 Alternative approaches

6.3.1 A scoping study to explore connectivity

Connectivity between sediment source (e.g. an agricultural field) and sink (e.g. rivers) has a very large effect on the amount of sediment retained in the landscape and how much reaches watercourses. Indeed, Boardman et al. (2019) consider than connectivity is more important than erosion rates in determining the off-site impacts of soil erosion and runoff. The current models described earlier in this report tend to use a simple 'landscape retention factor' to deal with connectivity (see Section 5.2.2). However, this is a crude representation of the complex processes involved and there remains a great deal of uncertainty associated with these factors.

The need to obtain information on connectivity was not directly addressed by Tye & Robinson (2019) and is not planned as part of current sampling or measurement programmes. Nevertheless, we consider that there would be great value in a scoping study to explore how aerial photography or satellite images (Google Earth) could be used to identify and map the frequency of erosion features, such as rills, that are successful in traversing field margins and breaking through field boundaries for delivery to rivers or road drainage systems.

This would be a relatively cost-effective way to provide data which could then be used to enhance existing models that attempt to quantify the landscape connectivity and sediment delivery to watercourses for different climate change scenarios. It would also provide an extremely useful baseline for characterising the frequency and site risk factors associated with 'muddy floods' that impact on roads. Each year this places a heavy burden on local councils in terms of clean-up costs; costs which may rise even further in an environment where the frequency of high intensity rainfall events is likely to increase (see for example Boardman & Vandaele, 2020; Hansel et al., 2019; Farewell et al., 2020).

6.3.2 Targeted measurements of specific sediment sources

Bankside erosion is a major source of sediment in Welsh watercourses, with modelling work estimating that it contributes around 30% of the total load (see Figure 12). Tye & Robinson (2019) concluded that remote sensing (costing *c.* £40K) or 'citizen science' (costing *c.* £10K) could be used identify bank erosion 'hotspots' as part of a targeted monitoring approach. More realistically, erosion pins or sediment source finger-printing would be required to quantify rates and confirm the effect of management changes, such as fencing off livestock.

A larger project focusing upon channel bank erosion across Wales could also be used to examine bank sediment loss more directly in relation to channel characteristics. There is scope for assembling a national database comprising important channel characteristics including, gradient, bank height, steepness and composition, stream power, poaching, vegetation cover and sinuosity. In addition, bank protection could be taken into account. Up to 10% of channel banks in the UK may be protected so that some of the most vulnerable zones are no longer being eroded. The database could be used to explore relationships between the bank erosion estimates provided by additional sediment source fingerprinting and key channel controls. Such work might investigate contrasts between cohesive and non-cohesive channel banks and the role of thresholds for erosion.

Tye & Robinson (2019) did not consider other potential 'hotspots' of soil erosion which we have identified in this report, including footpath (Section 5.5.5) and roadside verge (Section 5.5.6) erosion. There is currently little information specific to Wales on these sediment sources, and this report has relied largely on published data from England. Given how different the topographical and climatic conditions are in Wales, it would be extremely useful to collect targeted data on erosion rates from these sources, particularly as there are continuing efforts to increase access to the Welsh countryside, and the pressure on both footpaths and rural roads is likely to increase.

Mining discharges are monitored regularly in Wales, although the measurements available on the NRW website are from abandoned metal mines and report heavy metal concentrations rather than sediment loads. It should be possible to obtain data on the locations of discharges from abandoned metal and coal mines, together with flow rate data. In this way, mining discharges could be added to the model as additional point sources and treated in a similar way to STWs. Whilst there is little data available on sediment concentrations in mining discharges (from metal or coal mines), it may be possible to obtain this data or incorporate sediment measurements into existing monitoring schemes at a relatively low cost.

6.3.3 Farmer participation scheme

In an era of smart technology, an increasingly popular way of collecting data is via 'stakeholder engagement' using phone apps or other similar tools. We propose that a scoping study be undertaken to investigate how farmers could be encouraged to provide feedback on soil health and condition (via regular soil health checks) perhaps as part of Agri-Environment Scheme compliance. This might involve workshops with small group of farmers to explore how a combination of technology and regulation could be used to engage the farming community with issues surrounding soil erosion and loss, and incentivise them to provide data and implement mitigation measures.

It would be important to involve social scientists in these developments, so that we can draw on previous experience where participatory research approaches have been used to meet multiple economic and environmental objectives (e.g. Stoate et al., 2019).

7 SUMMARY

Wales differs from England in its topography, climate, soils and land use. It is a largely mountainous country with high rainfall (average *c*.1500 mm/yr), and with around 80% of the agricultural land area managed as grassland or rough grazing. This means that key factors controlling soil erosion rates (and hence sediment delivery to watercourses) such as slope, rainfall volume and intensity and vegetation cover are very different from those found in England.

A considerable amount of Defra and WG funding has already been devoted to developing a process-based modelling framework for the source apportionment of sediment (and nutrient) losses to water courses at both the national (England and Wales) and catchment scale. This approach has been used previously to support policy studies in Wales which have assessed the effectiveness of WAES. Estimates from these previous Welsh studies have indicated that, at the national scale, agriculture was the dominant source of sediment losses to water courses (62%), with river bank erosion contributing 27%. In comparison, the other sediment sources considered in the model (forest and woodland, urban runoff, discharges from sewage treatment works) provided a relatively minor contribution (<10% each) to the overall losses.

In view of the fact that the modelling framework has received considerable investment and has already been widely adopted for policy support studies by both Defra and WG, we do not consider that the development of a new modelling approach is justified. However, we have identified a number of areas where the data used to populate the model could be refined and validation of the model outputs improved.

Previous work has reviewed new and existing technologies that could be used for capturing evidence on the extent of soil erosion in Wales, including direct and indirect physical measurements, as well as remote sensing technologies. As a result, a programme to collect general information on soil condition using aerial photography and remote sensing, together with visual assessments is underway. However, there are opportunities for complementary approaches including:

- An exploration of ‘source-to-sink’ connectivity using aerial photography or satellite images to identify and map the frequency of erosion features, such as rills, that traverse field margins and field boundaries delivering sediment to rivers or road drainage systems.
- Targeted measurements of specific sediment sources including bankside, footpath and roadside verge erosion, and mining discharges, although the latter are unlikely to affect the overall balance of losses at the national scale.
- An investigation involving social scientists of how stakeholder engagement could be used to encourage farmers to provide feedback on soil health and condition, and engage with issues surrounding soil erosion and loss.

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