EXAMINING THE SPATIAL VARITAION OF ECOSYSTEM SERVICES ON AGRICULTURAL LAND: A VITAL STEP FOR DECISION-MAKING

RICHARD J. STONE (Student No. 23779195)

ENVS6009 Advanced Research Project

Word Count: 6675

Centre for Environmental Sciences, Faculty of Engineering and the Environment, University of Southampton, Highfield, Southampton, SO17 1BJ, UK

Email: rjs2g10@soton.ac.uk

Abstract: There are increasing efforts being taken to mainstream the framework of ecosystem services in to policy at all levels of governance, the most recent of which being the Common Agricultural Policy of the EU. The implementation of these policies will require ecosystem assessment and guidance for implementation on the ground. Ecosystem assessments to date have been dominated by large-scale assessments but there is an increasing need to re-focus the approach to a much finer scale to achieve relevance to local decisions in landscape management and design. This paper draws on existing data to disaggregate information on ecosystem services and show the fine-scale at which spatial variation can occur. An ex-ante analysis of the CAP reform was used as the most likely scenario for agricultural land use change to 2020. It investigates how targeted landscape management can be used to maximise the potential increases in ecosystem services and minimise potentially detrimental effects in agricultural ecosystems. There is inconsistency in the current literature over which variables should be used to measure ecosystem services at different spatial scales. Not all the variables currently used can be disaggregated to assess services being provided at a fine-spatial scale, for example farmland bird species richness. Arable land produces significantly different services to both pasture and woodland and a clear trade-off between calorie production and water quality can be observed. Under future scenarios of land use change cultivated goods are projected to increase calorific output within the study site at the expense of decreased water quality. This paper suggests that targeted landscape management making use of fine-scale ecosystem service maps will help to achieve the desired increase in ecosystem services from policy implementation.

Key words: Agriculture; CAP; ecosystem services; biodiversity; water quality; cultivation.

1.0 Introduction

1.1 Background

Covering more than 70% of the land area in the UK, agriculture represents the largest managed ecosystem and the most important form of land use (FAO 2009; Angus et al. 2009). Historically, agricultural ecosystems have been managed to maximize the provision of food, fuel and fibre that human populations depend on but more recently there have been efforts to try to quantify and deliver other 'ecosystem services' (ES) that are not currently included in management decisions (Dale & Polasky, 2007; Swinton et al. 2007). Agriculture-related ES include water purification; climate and air regulation; soil and genetic diversity as well as many cultural ES and the failure to recognize the value of these services has led to their degradation (Firbank et al. 2008; Daily et al. 2009; UKNEA, 2011; Firbank et al., 2013). The UK is one of the countries that has committed to measuring its progress towards the Convention on Biological Diversity's strategy against the 20 Aichi Biodiversity Targets; target 14 relates directly to the safeguarding and restoration of ES (CBD, 2010).

The conversion of land from its natural state to an agricultural ecosystem and the increasingly intensive practices associated with agriculture have been implicated in the pollution of aquatic ecosystems and the observed declines in biodiversity in recent decades (Carpenter et al., 1998; Donald et al. 2001; Firbank et al. 2008; Kleijn et al. 2009). The biggest driver of land use change in the European Union is the Common Agricultural Policy (CAP) and since its inception at the Treaty of Rome in 1957 it has undergone several re-designs and worked to transform the agricultural landscape (Stoate et al. 2001; Rounsevell et al. 2007; Cooper et al. 2009; Grant, 2010). The most recent reform for the period 2014-2020 introduces a mandatory 'greening' component to the Direct Payments Regulation (EU 1307/2013) that attempts to mainstream ES by including various measures such as permanent grassland and Ecological Focus Areas (EFA). Farms will have to dedicate 5% and later 7% of farmland as EFA that will include features such as field margins, hedges and landscape features with the aim of increasing ES on site. In the UK biodiversity and water quality will be prioritised when the CAP reform is implemented in the UK (DEFRA, 2014). It has been projected that the implementation of the CAP reform will lead to an 18% decrease in the proportion of agricultural land shared by pasture and an 18% increase in agricultural land shared by arable land in the UK (Lavalle et al. 2011). Other projections focus on the economic impact of the reform but little work has been done to examine whether the changes will have the desired effect of increasing ES (Metzger et al., 2006;

Plieninger et al. 2012). In contrast to these existing studies, this paper examines the current provision of ES on agricultural land and investigates an approach to land use management that can maximise the potential benefits under the CAP reform.

1.2 Measuring agricultural ecosystem services

Since the landmark study by Costanza et al. (1998), there have been increasing efforts to quantify the natural capital of ecosystems and the ES that contribute to human well-being (MA, 2005; TEEB, 2010; UKNEA, 2011). These large-scale ES assessments, as well as papers such as Foley et al. (2005) have worked to re-frame the trends in declining ES as a global problem however the delivery of services is intrinsically linked with land use and management decisions being made on a much smaller scale (Zhang et al. 2013). Current assessment methods are expensive in terms of the time, data input and technical expertise that they require; they also measure services at a broad scale and are limited by the coarse resolution of the input data (de Groot et al. 2010; Bagstad et al. 2013; Peh et al. 2013). Criticisms have also been levelled at current methodologies for inconsistencies between the variables that are measured and the final service of interest (Boyd & Banzhaf, 2007; Balvanera et al. 2013). In order to inform local decision making, assessments of ES need to be done for individual sites but there are few methodologies that offer decision-support tools at this scale. The Toolkit for Ecosystem Service Site-based Assessment (TESSA) addresses this gap and offers a decision-support tool at a site-scale for the comparison of a 'current state' and an 'alternate state' which is generally converted land use (Peh et al. 2013). On agricultural land many services are provided by, or differ between, individual fields or farms (Zhang et al. 2007; Laterra et al. 2012) however there is no existing methodology that allows the spatial disaggregation of ES at a field-scale.

Many of the indicators currently used to assess ES are still applicable to field-scale measurement. Birds are considered a good indicator for overall farmland biodiversity (Gregory et al. 2005). Many bird species have been able to adapt to extensively managed ecosystems such as agricultural landscapes and the measurement of the Farmland Bird Index (FBI) has been adopted as a Sustainable Development Indicator across the EU (Vickery et al., 2004; Butler et al. 2010). The history of bird monitoring schemes has meant there is a relative wealth of data when compared to other species groups (Bibby, 1999; BTO, 2013). The Common Bird Census involved the mapping of specific bird locations enabling the disaggregation of biodiversity data (BTO, 2009).

Measurements of nutrient levels and other contaminants are often used as a direct indicator of water quality in stream and river systems (Carpenter et al. 1998; Wood et al 2005; Ulén

et al. 2007). More recently there has been a growing awareness of the importance of sediment as a pathway for nutrient pollution, particularly from agricultural landscapes (Walling, 2005). As well as being a pollutant in its own right, suspended sediment in river systems is closely related to the phosphorus load and can be seen as a good proxy for general aquatic pollution (Oliver et al. 1979; Cooper, 1993; Walling, 2005). Up to 90% of sediment load has been attributed to surface flow and field drains making agricultural soil erosion a dominant factor in sediment supply to freshwater systems (Russell et al. 2001; Walling et al. 2008). Topography is a key factor in soil erosion and slope angle and length can differentiate sediment load contributions at a field scale (Renard et al. 1991; Vandaele & Poesen, 1995). Water quality, biodiversity and food production are the three services in focus under the CAP reform in the UK (DEFRA, 2014).

1.3 Decision support tools

The reforms of the CAP within the EU have highlighted the changing role of farmers to environmental stewards as well as producers (Banks & Marsden, 2000; Latacz-Lohmann & Hodge, 2003). This changing expectation is part of a changing culture where closer integration between science and policy is expected (Perrings, 2011a) however it is thought that stakeholders may need guidance in order to maximise the potential benefits in ES (UNEP, 2011). In order to support stakeholders in agricultural-environmental policy decision making a minimum-data, spatially explicit method is needed (Antle & Valdivia, 2006). The provision of ES is thought to be a good framework through which mapping can provide an engaging approach in order to support EU policies and guide decision-making (Maes et al. 2012).

It is in this context that this paper is written. The methodology uses an ex-ante analysis of the CAP by Lavalle (2011) to investigate the impact of the CAP on ES. The broad aim is to highlight that spatial variation of ES occurs at a fine-scale and that targeted land use management can be used to maximise the potential increases in ES. The objective is not to determine the ‘optimal’ land use solutions, but rather to inform a comparison of land management options and support decision-making.

2.0 Methodology

2.1 Study Area

The study area (GB grid reference: SK791024) comprised *c.* 155 ha of mixed arable and livestock farmland situated to the north of Loddington Farm in Leicestershire (Figure 1). The area was delimited by the local drainage basin and is managed by the Game and Wildlife Conservation Trust's Allerton Project in collaboration with neighboring sheep and horse farms. It was chosen because bird census data, geographic information system (GIS) data coverage and water quality data were readily available. It also represents a microcosm of lowland England; land use is divided with 23% arable land, 49.6% pasture, 17.7% woodland, 2.9% built environment, 2.2% scrub & grassland, 0.5% wetland & riparian and 4% environmental stewardship. Current Environmental Stewardship practices (Natural England, 2014) qualify for the outlined EFA criteria (EC, 2013) and so areas of set-aside, field margins and beetle banks were combined to calculate the EFA extent (Figure 2). Hedges were not considered part of the total EFA as they could not be assigned to a particular field.

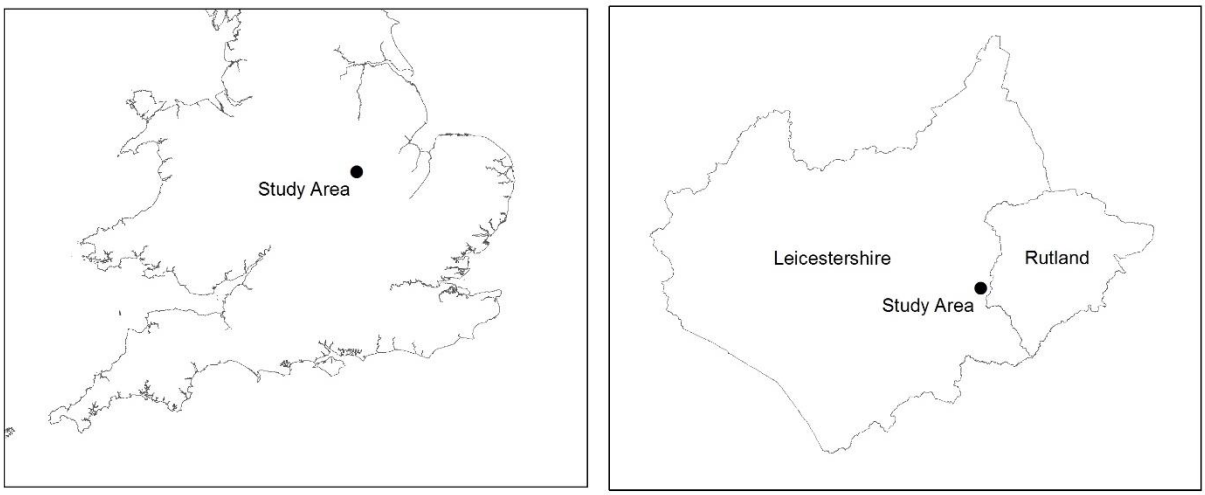


FIGURE 1: THE LOCATION OF THE STUDY AREA IN THE UK, WITHIN THE COUNTY OF LEICESTER

The analyses were confined to 21 'analysis zones' that together cover 128.9 ha and accounted for the three major land uses: arable, pasture and woodland (Appendix 1). Zones were aligned with field borders or clear land use divisions for relevance to management decisions and were chosen based on where data was available. Fields that had >25% of their area outside the study area or no available data were omitted from the analysis. The proportion of EFA was then calculated for each analysis zones and then categorised by percentage of total area into: 0%; <5%; 5-10%; and >10%.

The year 2012 was selected as the base year as most data was available for this period.

2.2 GIS coverage

Previous work had been done on site digitizing Ordinance Survey land cover maps and land use data for the study area to create a base map using ArcMap software (ESRI, 2012; Aronsson, 2013). The data was separated into individual layers and combined into broad land use categories (Figure 2). The zones for analysis were digitized using the study area base map and subsequently ground-truthed during site visits.

Elevation data in the form of a Profile DTM was obtained from EDINA's digmap service (EDINA, 2014) at a 10m resolution. The DTM was used to create an elevation profile of the study area and to calculate water flow accumulation within the drainage basin.

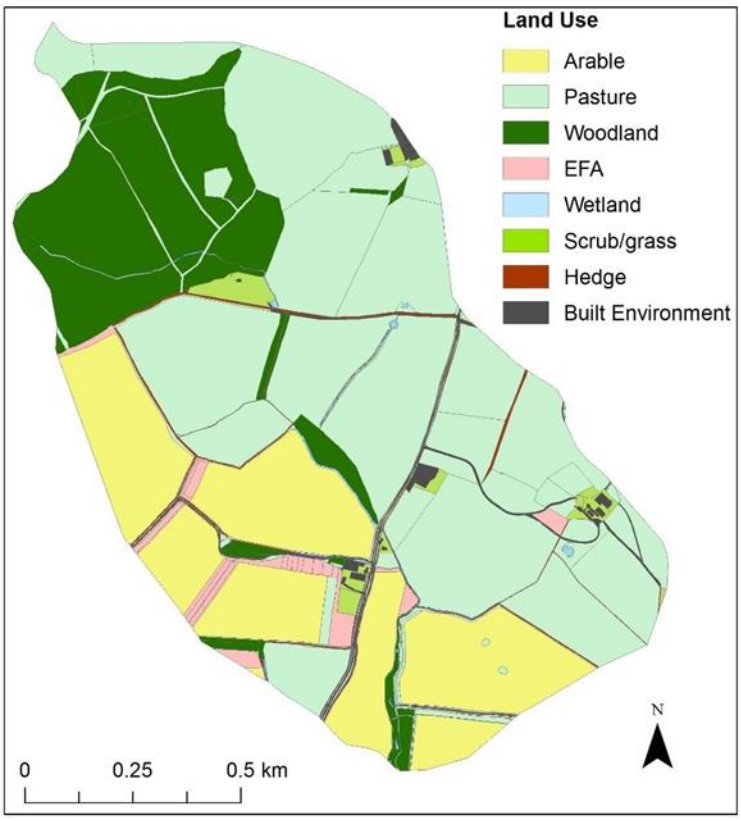


FIGURE 2: STUDY AREA WITH BROAD LAND USE CATEGORIES

2.3 Cultivation

Similar to Karali (2012), cultivation on all land uses was converted to calorific value in order to assess the final service as an energy value to human populations. Bread wheat was

selected as a proxy to represent general crops for human consumption and wheat crop yield data was used for the years 2008-2012. Each cropped field within the study area was cultivated with wheat for two of the years during the period under analysis. The mean average tonnage of crop was calculated over the two years and converted into calorific content. Calorific content was calculated using the value for soft white wheat taken from the United States Department for Agriculture (USDA, 2014).

The dead weight annual lamb production was used to represent cultivated goods for areas under pasture. The product of the average ewe stocking rate per hectare for the area, the average lamb carcass weight per ewe and field area were taken to calculate the total lamb yield. This average value was further adjusted using the supply of fodder to each field to account for variation in ewe stocking between fields. The total dry matter tonnage of fodder for the analysis area and the proportion of dry matter supplied to each field was calculated. The total lamb yield was then multiplied by the proportion of fodder supply to approximate the lamb yield per field and subsequently converted to calorific value using estimated calorific content as given by USDA (2014).

The quantity of wood harvested from areas of woodland in the study area was not recorded and so estimates were obtained from the Farm Manager. Estimates of the tonnage of timber removed and the approximate area of woodland where wood has been extracted was used to calculate the average wood yielded per hectare per year. This was then used to calculate the total yield for each woodland zone and an estimate of calorific content of timber as a wood fuel was used to convert the yield data to calorific value (Forestry Commission, 2010).

The calorific content of the yield per hectare within each zone was calculated for the final measure for cultivation to show the differences in yield between zones.

2.4 Biodiversity

The Common Bird Census methodology was employed to survey species richness and density estimates across the study area. Individual records were gathered by a trained observer on eight weekly visits from May to July 2012 and contacts with birds by sight or sound were recorded on visit maps and subsequently transposed to digitized species map using ArcGIS software (ESRI, 2012). Gamebirds and woodpigeons (*Columba palumbus*) were omitted from the survey due to existing management activities within and directly adjacent to the study area that would have affected population numbers. The survey route mainly followed field boundaries as these are important habitats for many farmland bird as

a source of cover and foraging opportunities (McMahon et al. 2005). Woodland survey routes followed woodland rides.

To analyse the contribution of land use and EFA to improving numbers of farmland bird species, the species richness of birds was measured, concentrating on bird species listed in Siriwardena et al (1998) and bird species included in the FBI. Nuthatch (*Sitta europaea*) and Spotted Flycatcher (*Muscicapa striata*) were excluded from the analysis. The bird locations were split by the 21 analysis zones. Birds included in the FBI were analysed as a separate guild in order to obtain an FBI species richness as well as the richness of all species under analysis (Appendix 2). Species richness (i.e. number of bird species present) was determined at the field level by the total number of species recorded within a field over all visits. This was then used for the comparison between different land use and EFA categories.

2.5 Water Quality

The study area represents a local drainage basin with a stream running down the centre of the basin that is sampled for turbidity (Figure 3). Turbidity values are measured in average mg/l and recorded every 15 minutes. Turbidity data from the meter was analysed for the year 2012 and restricted to readings for the two days after the filter is cleaned (Thursday and Friday) to improve accuracy. The mean turbidity was taken from this sample. At a nearby comparison catchment, maximum storm event suspended sediment measurements were recorded for the three land uses: arable, pasture and woodland (Table 1).

TABLE 1: MEAN SUSPENDED SEDIMENT FOR LAND USE AND 'CONTRIBUTION FACTOR'

Land use	Max. storm suspended sediment (mg/l)	Contribution
Arable	150.41	0.587
Pasture	29.92	0.928
Woodland	29.96	0.918

The suspended sediment values were taken as a proportion of the mean turbidity for the study area stream to give an approximation of the contribution to the maintenance of water quality. The reciprocal of this value was then taken so that the higher value represented an

improved ‘service’. The analysis zones were then rasterized and assigned this ‘contribution factor’ according to their land use.

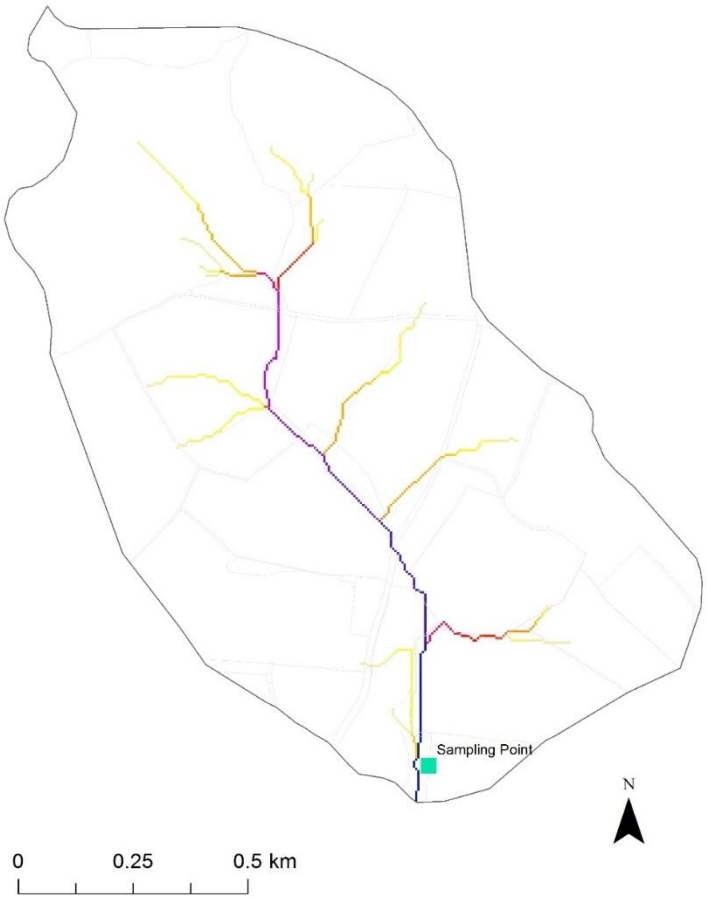


FIGURE 3: STUDY AREA WATER FLOW ACCUMULATION WITH STREAM SAMPLING POINT INDICATED

To account for the contribution of hillslope processes and soil erosion a ‘slope factor’ was calculated using ArcMap 10.1 that is equivalent to the ‘LS factor’ included in the Revised Universal Soil Loss Equation (RUSLE), combining slope length and steepness (Moore & Wilson 1992; Zhang et al. 2013; Griffin et al. 1988).

$$Slope\ factor = \left(\frac{A}{22.13}\right)^m \left(\frac{\sin\theta}{0.0896}\right)^n$$

Where:

A = Flow accumulation * DTM cell resolution

θ = slope degrees * 0.01745

m and *n* are parameters

Lower than average values were selected for parameters *m* and *n* [m=0.4 and n=1] as more dispersed flow is expected owing to the clay soils, the relatively high vegetation cover and work being done within the study area to reduce soil loss (McCool et al. 1997; GWCT 2013; Ockenden et al. 2014). The relative contribution of the slope factor to stream suspended sediment was assumed to be equivalent to the contribution of the LS factor in the RUSLE equation. The final value for water quality was therefore:

$$WQ = [Contribution\ factor] * [Slope\ factor]$$

A mean WQ factor was then taken for each zone under analysis.

2.6 Evaluated Scenarios

The ES provision values were all normalised to a range of between 0-1. Each zone within the study area was assigned a value for each of the ES and a mean average for each service was taken to represent current services provided across the study area. The statistical analyses were performed using IBM SPSS Statistics (IBM, 2012) and in all cases significance was taken at $\alpha = 0.05$ level. Data were checked for normality using the Shapiro-Wilk test and in some cases data did not conform to normality (Appendix 3).

Differences in the ES values within land use categories and between categories were revealed through use of one-way Analysis of Variance (ANOVA). Where this indicated a significant difference, pair-wise comparisons of samples were undertaken using the Holm-Sidak test.

Plausible ‘alternate states’ were identified according to changing policy and conversations with stakeholders. Average values for biodiversity and cultivation were calculated for each land use category and used to analyse the impact of land conversion. To calculate the average cultivation values, the calorific content of the total yield from each zone was averaged. Values for the contribution to maintaining water quality were calculated by multiplying the ‘contribution factor’ for each land use by the ‘slope factor’ for the entire study area. An initial analysis examined the impact of a 100% conversion between the three land uses on each of the three services provided across the study area (6 scenarios). To examine the impact of the land use change scenario outlined by Lavalle (2011) on ES values, the analysis zone areas were calculated to 2 s.f and the different assemblages of zones that when combined, achieved an 18% conversion rate were included in the analysis.

3.0 Results

3.1 Ecosystem service assessment

The methodology is applied to construct ES provision maps for the year 2012 (Figure 4-6). Cultivated services is represented on the map as yield/ha. All services were normalized to a scale of between 0-1. Values approaching 1 represented by darker shaded colours can be interpreted as having an improved service when compared to values approaching 0, represented by a lighter shaded colour. Average values for each land use category are shown in Table 2. Figure 7 and Figure 9 show a contrast between arable land and the two other land uses. Table 2 indicates that arable land achieves a much higher value for cultivated services however the contribution to the maintenance of water quality is much lower.

TABLE 2: MEAN VALUES FOR EACH ECOSYSTEM SERVICE NORMALISED TO VALUES BETWEEN 0-1

Land Use	Biodiversity	Cultivation	Water quality
Arable	0.58	0.83	0.07
Pasture	0.33	0.01	0.68
Woodland	0.63	0.02	0.62

A one-way ANOVA of the 21 analysis zones revealed a significant difference between land use categories for the contribution to the maintenance of water quality ($F = 78.829$, $p= 0.000$) and cultivation ($F = 252.371$, $p=0.000$) but biodiversity did not differ significantly between categories. Post-hoc comparisons using the Holm-Sidak test indicated that arable land was significantly different ($p<0.05$) to the two other land use categories for both services. Pasture and woodland were not shown to be significantly different for any of the measured ES.

Values for cultivation on arable land were two orders of magnitude higher than pasture and woodland for zones when the total calorific content was measured (Appendix 4). Pasture zones had lower values for calorie cultivation than all land uses except for woodland within

Zone 1 [Launde Wood] where it was assigned a value of 0 after it was indicated that no wood from the Zone was used as wood fuel. All other woodland zones had the same value (0.02) as the original data was based on farm-scale estimates and could not be differentiated for calories/ha. Four pasture zones were also assigned a value of 0 as they were used for horse grazing and produced no product of calorific value.

Values for biodiversity were not found to be significantly different between groups however values were more variable within groups. A Shapiro-Wilk test showed the biodiversity values to be normally distributed and Figure 8 shows the distribution of zone values. Pasture zones were identified as having the lowest mean value for biodiversity (Table 2) and the lowest biodiversity value was calculated for Zone 7 [Middle Field], categorised as pasture. The highest value for biodiversity was found in Zone 1 [Launde Wood].

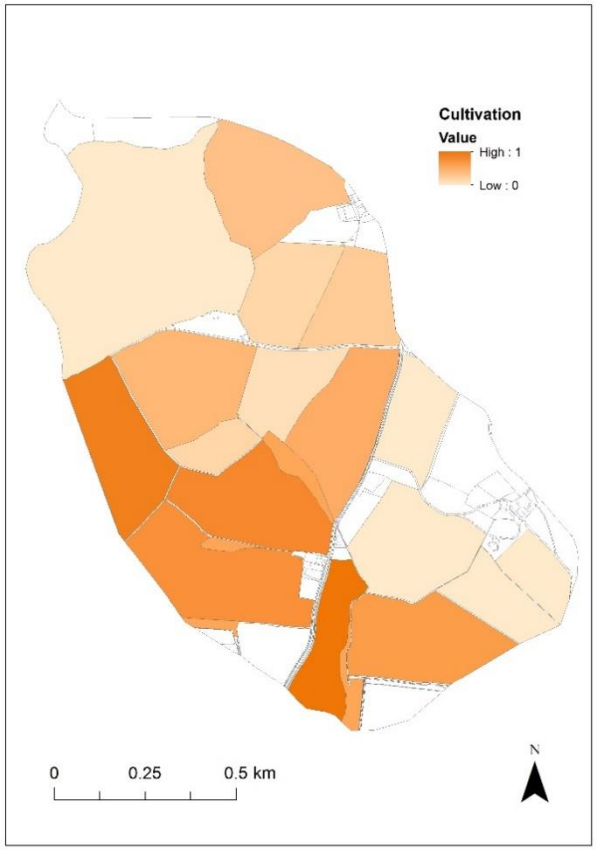


FIGURE 4: ES VALUE MAP FOR CULTIVATION

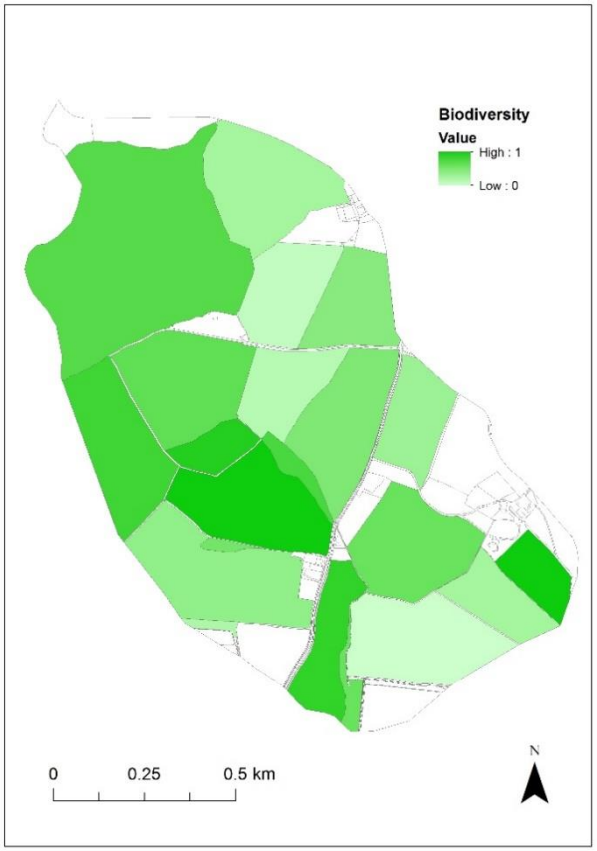


FIGURE 5: ES VALUE MAP FOR BIODIVERSITY

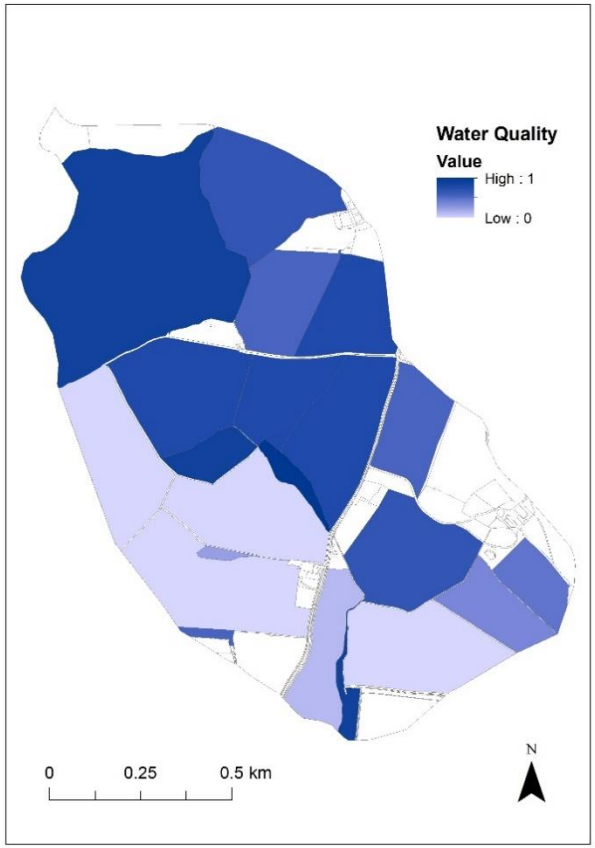


FIGURE 6: ES VALUE MAP FOR WATER QUALITY

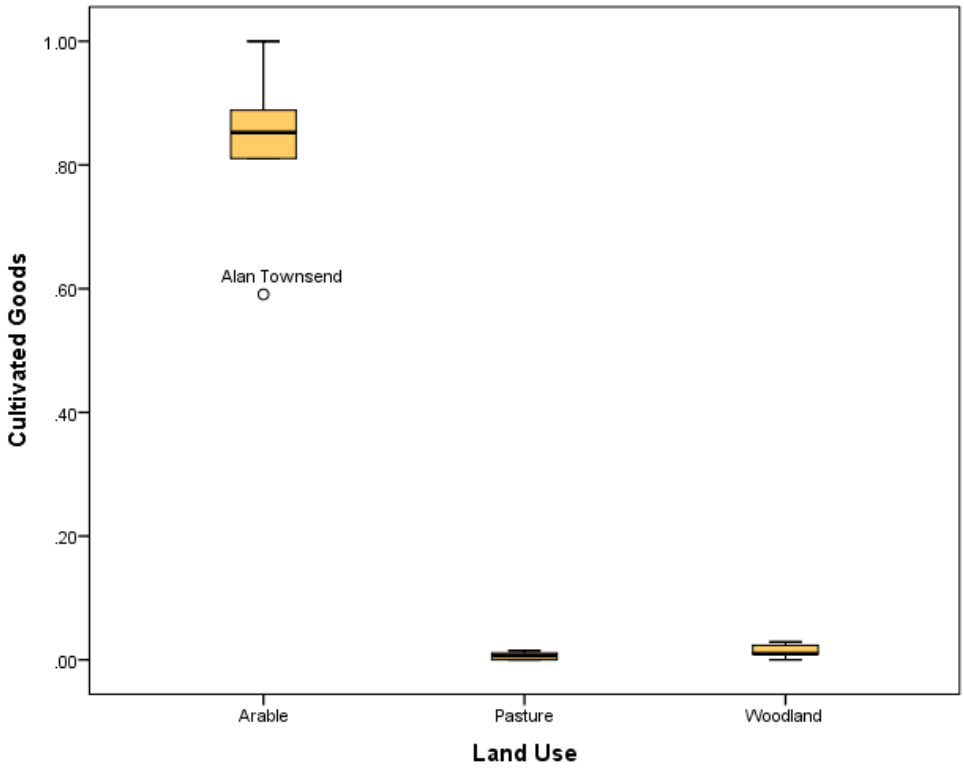


FIGURE 7: A BOX PLOT FOR CULTIVATION SHOWING THE VARIATION IN VALUES FOR LAND USE

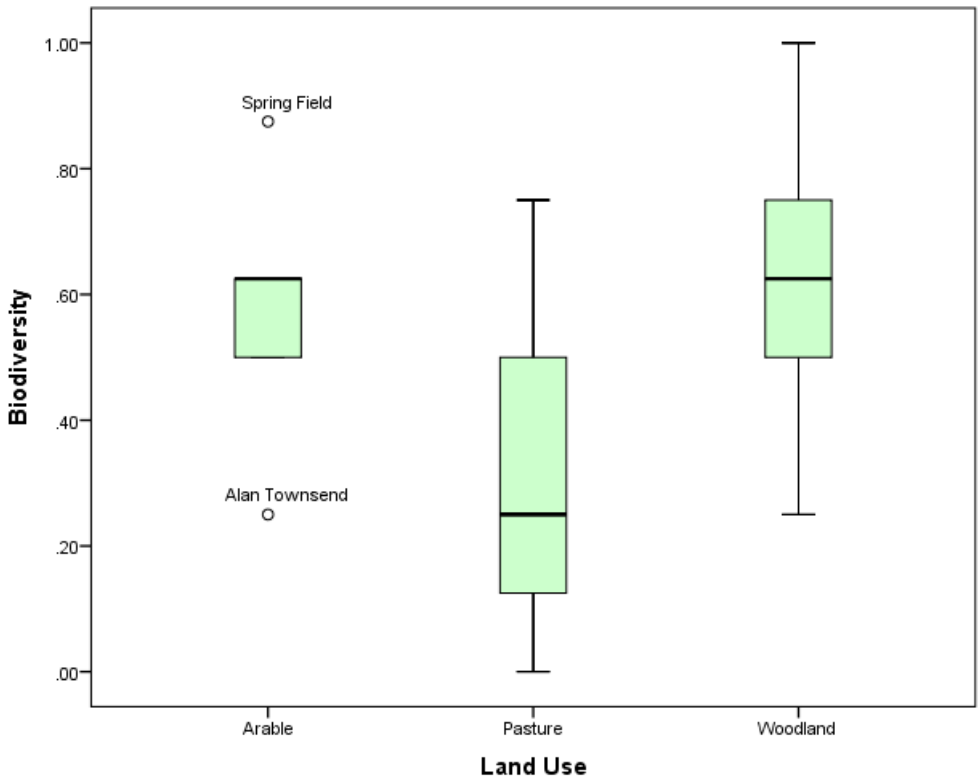


FIGURE 8: A BOX PLOT FOR BIODIVERSITY SHOWING THE VARIATION IN VALUES FOR LAND USE

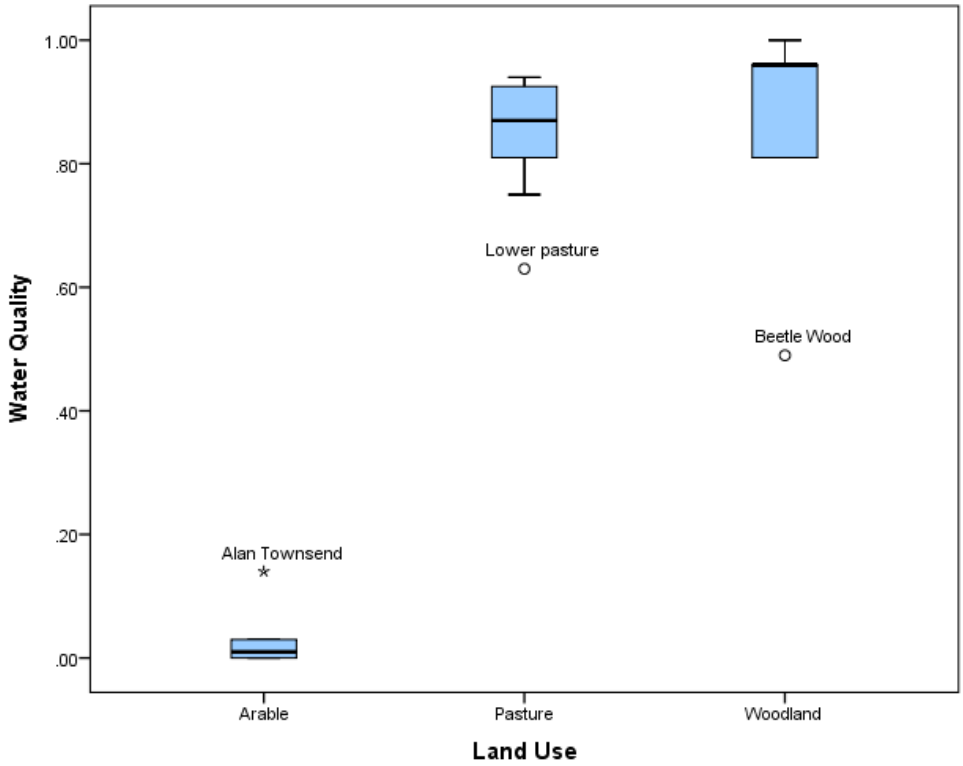


FIGURE 9: A BOX PLOT FOR WATER QUALITY SHOWING THE VARIATION IN VALUES FOR LAND USE

Arable land was shown to have a significantly lower ($p < 0.05$) contribution to the maintenance of water quality when compared to pasture and woodland. This is clearly shown in Figure 9 where the five arable zones are assigned the five lowest values for water quality. The highest value for water quality was observed for Zone 16 [Alan Townsend] and was shown to be an outlier when compared to the other arable zones (Figure 9). Before the adjustment for slope was applied to the 'contribution factor', woodland was shown to have a higher contribution to the maintenance of water quality than pasture. When the impact of hillslope processes are accounted for at a field-scale, pasture achieves a higher mean value than woodland (Table 2). Pearson's correlation indicated a strong negative correlation between cultivation and water quality variables ($r = -0.9407$, $p < 0.05$).

3.2 Ecological Focus Area

Pearson's correlation indicated that there was no significant correlation ($p > 0.05$) between the species richness of FBI birds and the species richness of all species. An analysis of FBI

species richness showed that of the zones that had above average species richness and number of individual sightings (Table 3), the highest value was found in Zone 13 [Cawthorn]. Of the four zones that were categorised as pasture, three of them were used for horse pasture.

Arable land had the highest average FBI species richness (2.4) followed by pasture (1.45). Only 1 farmland bird species was recorded within a zone categorised as woodland over all visits during the bird surveys. Woodland zones had the highest species richness for all bird species (7) followed by arable land (6.6).

TABLE 3: ZONES WITH ABOVE AVERAGE VALUES FOR FBI RICHNESS AND SIGHTINGS

ID	Land Use	EFA	FBI Species Richness	FBI Sightings
5	Pasture	0%	2	5
10	Pasture [H]	0%	2	5
14	Crop	5-10%	2	7
15	Crop	>10%	3	6
12	Pasture [H]	0%	3	6
9	Pasture [H]	0%	3	9
13	Crop	<5%	4	29

When examining only zones that contained EFA, no significant correlation ($p > 0.05$) was found between the proportion of land used as EFAs and any of the variables under analysis (Table 3).

TABLE 4: ES VALUES FOR ZONES WITH PROPORTION OF EFA

ID	EFA%	Biodiversity	FBI Species Richness	Yield/ha (tonne)
13	<5	0.63	1	0.95
14	5-10%	0.88	0.5	0.95
15	>10	0.63	0.75	0.92
16	5-10%	0.25	0.25	1.00
17	<5	0.50	0.5	0.77

3.3 Conversion scenarios

Under a 100% conversion from one land use to another, the biggest gains and losses are observed in values for cultivation (Figure 10). A conversion from pasture to arable results in a 144% increase in average value for cultivation across the whole site. The same conversion also resulted in a 20% decrease in the value for contribution to the maintenance of water quality. Biodiversity values increased the most when land was converted from pasture.

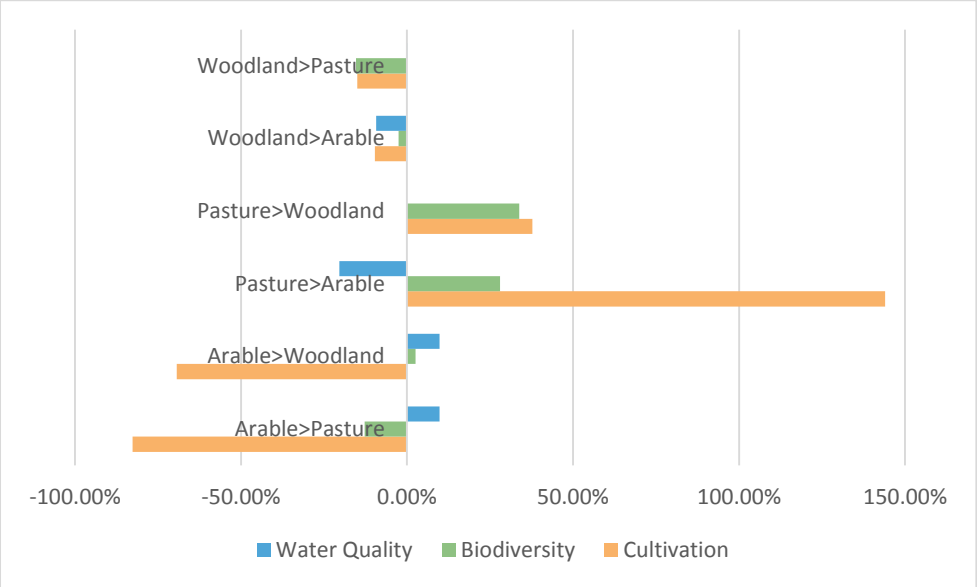


FIGURE 10: GAINS AND LOSSES FROM A 100% CONVERSION BETWEEN LAND USES

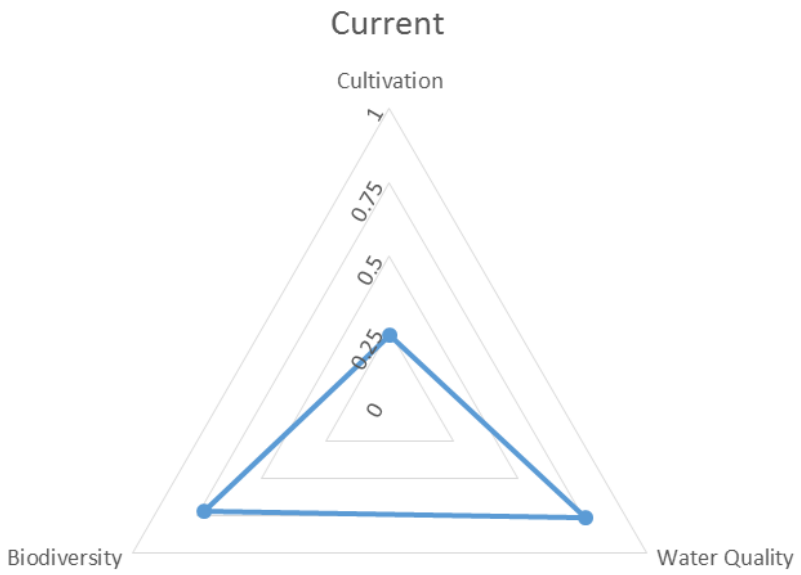
3.4 CAP Scenario

TABLE 5: GAINS/LOSSES FROM LAVALLE (2011) LAND USE CHANGE SCENARIO

	18% P>A Conversion	
	Gains/losses (%)	Range (%)
Cultivation	44.79	8.25
Water Quality	-15.96	3.17
Biodiversity	6.73	3.38

The 18% conversion of proportion of agricultural land from pasture to arable outlined as a likely scenario under the CAP reform to 2020 could be achieved through 26 different combinations of zone conversions (Appendix 5). Each conversion scenario was analysed to investigate the variation in ES provision between zones. Figure 11 and 12 show the average

ES provision for the study area in the 'current' and 'alternate' state when the maximum service provision from an analysis zone is 1. The average gains and losses in ES provision for the 26 conversion scenarios are displayed in Table 5. The largest gains are expected for cultivation where a ~45% increase is the equivalent to ~2.5M calories or 75 tonnes of bread wheat. Cultivation also showed the largest range between the maximum and minimum



expected gains between conversion scenarios

FIGURE 11: AVERAGE CURRENT ES PROVISION VALUES ACROSS THE STUDY AREA

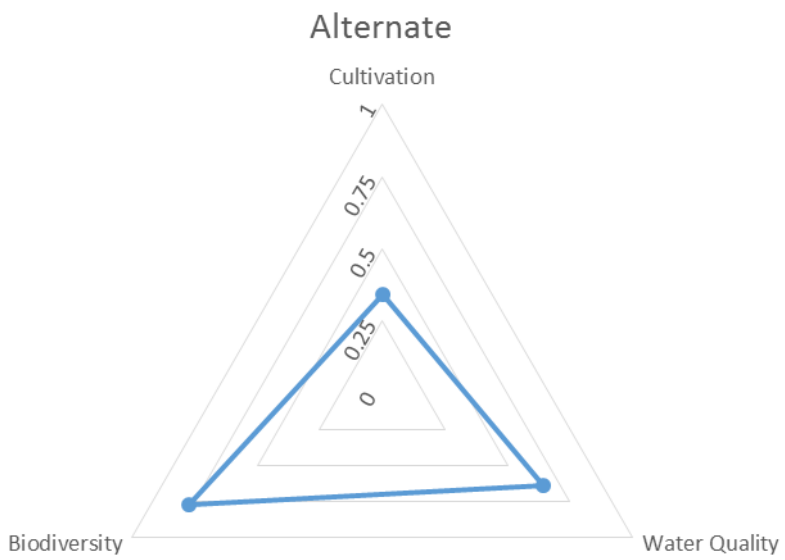


FIGURE 12: AVERAGE ES PROVISION VALUES UNDER LAVALLE (2011) LAND USE CHANGE SCENARIO

4.0 Discussion

4.1 Explanation of results

The methods employed by this paper enable the field-scale specific analysis and mapping of three ecosystem services that can be used to guide local decision making. Owing to the different variables that were used to calculate the different ES values, services are not directly comparable. Therefore a 20% increase in one variable may not be equivalent to a 20% increase in another, for example. Analysis of the results should only draw conclusions from within services.

Based on the analysis and the presented maps of ES provision it can be shown that calorific value is much higher on arable land than both woodland and land used for pasture. Though this may be an intuitive conclusion it is useful to display the scale of difference between arable and pasture land (two orders of magnitude) when the CAP reform is framed with increasing food production as one of the main objectives (EC, 2013). A large proportion (35%) of land was not used for the provision of food or fuel, including 30% of pasture. The pasture zones not used for the production of food were instead used for keeping and grazing horses. This activity results in a decrease in the average ES provided across the site when in fact recreational activities represent another service provided across agricultural landscapes that hasn't been quantified for this analysis. Other analysed scenarios of future land use within Europe have projected an increase in recreational areas such as this at the expense of areas of both arable and pasture land (Rounsevell et al. 2005). According to this analysis the proportion of arable land would have to increase relative to pasture in order to compensate for the decrease in calories produced.

The trade-off of services in agriculture is one that has been well documented in the literature (Zhang et al., 2007; Power, 2013) and the negative correlation between the values for the provision of cultivated goods and the contribution to the maintenance of water quality is a clear illustration of this. On a broad scale, the high values for cultivation observed on arable land are associated with the generally low values for water quality. This is a trend that has been observed and is attributed to the lack of crop cover and the tramlines from farm machinery (Chambers et al. 1992; Chambers et al. 2000). The correlation can be seen to hold true down to the field level. Zone 16 [Alan Townsend] was found to have the highest contribution to the maintenance of water quality largely owing to the slope factor that was applied as Zone 16 has the second lowest average slope of all the zones in the drainage basin. However the cultivation factor for Zone 16 was the lowest measured across all arable zones when measured as calorific value/ha. This measure is independent of

adjustments in the methodology and suggests that slope degree is positively correlated with crop yield. This is counter-intuitive as increasing slope steepness would normally result in increased soil erosion and reduced topsoil thickness (Julien & Simons, 1985; Al-Kaisi, 2001). It is also contrary to the results of previous analyses on the effect of slope on crop yield (Krevchenko, 2000). A Pearson's correlation did indicate a positive correlation between slope and crop yield for the five agricultural zones however with such a small sample size it was not thought to be significant. More research is needed to investigate this apparent anomaly.

The large variation in values for biodiversity observed across all land uses indicate that the use of the species richness of farmland birds may not be appropriate at such a fine scale. It is more commonly used as a tool for comparison at broader scales, between landscapes or countries (Butler, 2010; Wretenberg et al. 2010; Overmars, 2014) and this is most likely due to the relatively high-dispersal ability confounding any results from fine-spatial disaggregation of individual bird sightings (Tscharntke et al. 2005). The lack of correlation between the overall zonal species richness and the richness of FBI species is to be expected owing to the farmland specialisms of FBI species and shows that the inclusion of other species dilutes the contributions to boosting farmland bird numbers that EU policy focusses on. A considerable gap still exists for a widespread indicator of biodiversity (Feld et al. 2009) and progress may require closer integration of modelling, scenario and field-based monitoring in order to outline a common, useful indicator (Haines-Young, 2009).

The comparison between FBI species richness and land use draws few conclusions. Such a small sample size means that even where correlations between variables are indicated, extreme caution should be taken before extrapolating lessons from these results. The higher prevalence of FBI species on arable land with EFA is to be expected and falls in line with the study by Henderson et al. (2000), however the comparison in this analysis suffers from the lack of a control scenario. The effect of the proportion of EFA within a field on bird numbers is as yet unsure and other projections have to date only projected a "less pronounced decline" (Chiron et al. 2013).

The 18% conversion from pasture to arable land use projected by Lavalle (2011) is likely to lead to a large increase in the calorific value of agricultural land in the UK because of the higher calorie density of arable land. As mentioned before this is likely to have a trade-off of a decrease in water quality and this analysis projects an average 16% decrease in the contribution to the maintenance of water quality. The spatial variation of services across the study area mean that these increases and decreases have a range of possible values. The range is higher for cultivation values, mainly owing to the fact that under certain scenarios,

land not currently producing any goods with a calorific value would come into production. In order to maximize the potential benefits and minimize the potential declines, spatially targeted land management would be needed.

4.2 Use in decision making

The objective of this study was to provide an analysis of fine scale ES that can be used to inform local decision making. As previously mentioned, site-specific measures are useful to understand how particular management decision affect the delivery of ES but they cannot be used to understand the overall effects on the system (Dale & Polasky, 2007). Though there are considerable uncertainties involved in estimates of ES values, local maps that display approximations of services provide an engaging way of visualizing fine-scale variation in ES to aid management and decision making.

Decisions may include which field will be brought under cultivation as arable land, how to spatially arrange current crops or where to place EFA features that will increase ES provision. According to Firbank et al. (2013) land should be allocated to different ecosystem services depending on its suitability. This may mean the 'spatial redistribution of high sediment export risk land use to areas of the catchment with the greatest intrinsic sediment retention capacity', or, putting arable land on flatter fields (Johnes & Heathwaite, 1997). The position of field wetlands in areas of high soil erosion is another method to maximize ES potential (Hein et al. 2006; Ockenden et al. 2014). This targeted spatial management of ES is thought to be cheaper and less disruptive (Braden et al. 1989). Though the depiction of ecosystem services as being site bound on static maps has been criticized (Tallis et al. 2008), the production of fine-scale ES maps can be a vital tool in this spatial targeting and decision-support (Maes et al. 2012), clearly highlighting where the largest potential for gain is. Simple maps such as these (Figure 4-6) can act as an effective decision support tool for landscape managers.

4.3 Caveats to the methodology

Rapid measurements of ES using minimal data can only represent a simplification of more complex ecological processes and any interpretations should be treated with caution and acknowledge the underlying assumptions (Carpenter et al., 2009; Posthumus et al., 2010).

Through prioritising the use of existing data only, it often represents only a 'snapshot' in time that is limited in usefulness; the provision of ES varies temporally as well as spatially

however the nature of the data does not allow for this analysis. The analysis of bird location data from one year means that weather effects on either birds or detectability could confound other environmental influences. Inaccuracies will also enter through the use of datasets that have differing time periods. The water quality valuation used data for turbidity that had been collected across an entire calendar year however the values for cultivation on arable land were calculated using a winter wheat crop. Winter cereal is less susceptible to soil erosion and though the soil losses can be expected to be greater than pasture and woodland, they may be overestimated using annual data (Chambers & Garwood, 2000).

The reliability of the turbidity data is also subject to some scrutiny. The turbidity meters used to calculate the maximum storm event suspended sediment for the three land uses had been calibrated and their accuracies checked; the turbidity meter for the study area catchment had not been calibrated by the time this paper was produced however. Though efforts were made to increase the accuracy of the values and the readings are what might be expected for a comparable catchment, the reliability of the results has not yet been verified. The water quality index is also limited in its scope and what it can account for. The reductions in sediment transport that can be achieved through landscape features would work to further differentiate between fields however were not included in the index (Ockenden, 2014).

The study area used for the analysis is managed by the GWCT and practices on site are used with the intention of increasing the provision of ecosystem services and is not fully representative of other lowland enclosed farmland. GWCT's Loddington Farm accounts for almost one third of the study area and wheat and oat crops are grown to an environmental standard to be sold as Conservation Grade. Farms growing crops at Conservation Grade standard have noted a 41% increase in birds (Conservation Grade, 2014) and this decreases the relevance of the findings to farms that do not meet this standard.

5.0 Conclusions

After several global and national assessments of ecosystem services and the recognition that the global trend is of a wide-spread decline, the focus of the assessment needs to be reframed to a local-scale to gain relevance to land management decisions on the ground. It was the aim of this paper to demonstrate how the provision of ecosystem services differs even between agricultural fields and how targeted management can maximize service benefits. The results demonstrate the variability of ecosystem services at a fine scale but also the difficulties in assessing such variability. No methodology has been produced to

date that enables this kind of analysis however the potential use of this assessment can be demonstrated. With increasing efforts to mainstream the framework of ecosystem services in all levels of policy and expectations that this will be translated on the ground, ecosystem assessment toolkits will be required at all scales, particularly at finer resolutions. Current efforts to assess ecosystem services have been disjointed and a common set of variables are yet to be agreed upon. It is thought that the process would benefit in the future from closer integration of policy, modelling, and monitoring.

Agriculture is the most important form of land use in the UK policy has historically been the biggest driver of land use change. Much attention was given to the CAP policy reform and the integration of ecosystem services as part of its policy objectives. The CAP reform is projected to result in a change in the balance of agricultural land and with it, the ecosystem services it provides. Owing to the fine-scale variations in ecosystem services there is a range of potential impacts that policy implementation might have. In order to maximize benefits where possible, stakeholders and landscape managers need decision support tools and guidance on the best practicable option. Simple fine-scale ecosystem service maps can provide this tool.

5.0 Acknowledgements

Many thanks to C. Stoate from the GWCT who has helped with guidance on experimental design and allowed the use of GWCT data. I am grateful to all farmers and land owners who allowed access to their land. I am thankful for the initial advice provided by Dr P. Osborne to help guide the project. I would also like to thank Dr P. Shaw for helping me to overcome some of the difficulties of carrying out an advanced research project.

References

Al-Kaisi, M., (2001). Soil erosion and crop productivity: topsoil thickness. *Integrated crop management*. [Online] Available at: http://www.ipm.iastate.edu/ipm/icm/2001/1-29-2001/topsoilerosion.html. [Accessed 15/04/2014]

Angus, A., Burgess, P. J., Morris, J., & Lingard, J. (2009). Agriculture and land use: Demand for and supply of agricultural commodities, characteristics of the farming and food industries, and implications for land use in the UK. *Land Use Policy*, *26*, 230–S242.

Aronsson, L. (2013). *The School Farm demonstration catchment*. [Online] Available at http://www.gwct.org.uk/allerton/catchment-research/the-school-farm-demonstration-catchment/. [Accessed 15/06/2013].

Bagstad, K. J., Semmens, D. J., Waage, S., & Winthrop, R. (2013). A comparative assessment of decision-support tools for ecosystem services quantification and valuation. *Ecosystem Services*, *5*, 27–39.

Banks, J., & Marsden, T. (2000). Integrating Agri-Environment Policy, Farming Systems and Rural Development: Tir Cymen in Wales. *Sociologia Ruralis*, *40*(4), 466–480.

Balvanera, P., Siddique, I., Dee, L., Paquette, A., Isbell, F., Gonzalez, A., Byrnes, J., O'Connor, M. I., Hungate, B., & Griffin, J. N. (2014). Linking Biodiversity and Ecosystem Services: Current Uncertainties and the Necessary Next Steps. *BioScience*, *64* (1), 49–57.

Bibby, C. J. (1999). Making the most of birds as environmental indicators. *Ostrich*, *70*(1), 81–88.

Boyd, J., & Banzhaf, S. (2007). What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics*, *63*(2-3), 616–626.

Braden, J. B., Johnson, G. V, Bouzaher, A., & Miltz, D. (1989). Optimal Spatial Management of Agricultural Pollution. *American Journal of Agricultural Economics*, *71*(2), 404–413.

BTO (British Trust for Ornithology). (2009). *Common Bird Census*. [Online] Available at http://www.bto.org/survey/complete/cbc.htm. [Accessed 15/06/2013].

BTO (British Trust for Ornithology). (2013). *Bird Indicators: Indicator update – October 2013*. [Online] Available at http://www.bto.org/science/monitoring/developing-bird-indicators. [Accessed 28/02/2014].

Butler, S. J., Boccaccio, L., Gregory, R. D., Vorisek, P., & Norris, K. (2010). Quantifying the impact of land-use change to European farmland bird populations. *Agriculture, Ecosystems & Environment*, *137*(3-4), 348–357.

Carpenter, S. R., Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N., & Smith, V. H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications*, *8*(3), 559–568.

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Díaz, S., Dietz, T., Duraiappah, A. K., Oteng-Yeboah, A., Pereira, H. M., Perrings, C., Reid, W. V., Sarukhan, J., Scholes, R. J. & Whyte, A. (2009). Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences*, *106*(5), 1305–1312.

CBD (Convention on Biological Diversity). (2010). *COP 10 Decision X/2. Strategic Plan for Biodiversity 2011-2020*. [Online] Available from: http://www.cbd.int/decision/cop/?id=12268 [Accessed 28/02/2014]

Chambers, B. T., Davies, D. B., & Holmes, S. (1992). Monitoring of water erosion on arable farms in England and Wales, 1989–90. *Soil Use and Management*, *8*(4), 163–169.

Chambers, B. J., & Garwood, T. W. D. (2000). Monitoring of water erosion on arable farms in England and Wales. 1990–94. *Soil Use and Management*, *16*(2), 93–99.

Chiron, F., Princé, K., Paracchini, M. L., Bulgheroni, C., & Jiguet, F. (2013). Forecasting the potential impacts of CAP-associated land use changes on farmland birds at the national level. *Agriculture, Ecosystems & Environment*, *176*, 17–23.

Conservation Grade. (2014). *Nature friendly conservation farming*. [Online] Available at: http://www.conservationgrade.org/nature-friendly-farming/ [Accessed 02/04/2014]

Cooper, C. M. (1993). Biological Effects of Agriculturally Derived Surface Water Pollutants on Aquatic Systems—A Review. *Journal of Environmental Quality*, 22(3), 402-408.

Cooper, T., Hart, K., & Baldock, D. (2009). *Provision of public goods through agriculture in the European Union, Report prepard for DG Agriculture and Rural Development*. Institute for European Environmental Policy, London.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limberg, K., Naeem, S., O'neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., & van den Belt, M. (1998). The value of the world's ecosystem services and natural capital. *Ecological Economics*, *25*(1), 3–15.

Daily, G. C., Söderqvist, T., Aniyar, S., Arrow, K., Dasgupta, P., Ehrlich, P. R., Folke, C., Jansson, A., Jansson, B., Kautsky, N., Levin, S., Lubchenco, J., Mäler, K., Simpson, D., Starrett, D., Tilman, D., & Walker, B. (2000). The Value of Nature and the Nature of Value. *Science*, *289* (5478), 395–396.

Daily, G. C., Polasky, S., Goldstein, J., Kareiva, P. M., Mooney, H. A., Pejchar, L., Ricketts, T. H., Salzman, J., & Shallenberger, R. (2009). Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment*, *7*(1), 21–28.

Dale, V. H., & Polasky, S. (2007). Measures of the effects of agricultural practices on ecosystem services. *Ecological Economics*, *64*(2), 286–296.

DEFRA (Department for Environment, Food and Rural Affairs). (2014). *New environmental scheme for farmers to prioritise biodiversity*. [Online] Available at https://www.gov.uk/government/news/new-environmental-scheme-for-farmers-to-prioritise-biodiversity. [Accessed 28/02/2014].

De Groot, R. S., Alkemade, R., Braat, L., Hein, L., & Willemen, L. (2010). Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity*, *7*(3), 260–272.

EC (European Commission). (2011). *Our life insurance, our natural capital: an EU biodiversity strategy to 2020*, COM(2011) 244, Brussels, European Commission.

EC (European Commission). (2013). *Overview of the CAP Reform 2014-2020*. [Online]. Available at http://ec.europa.eu/agriculture/cap-post-2013/. [Accessed 05/01/2014]

EDINA, (2014). *Digimap*. [Online] Available at http://digimap.edina.ac.uk/digimap/home. [Accessed 16/08/2013].

Espinosa, M., Psaltopoulos, D., Santini, F., Phimister, E., Roberts, D., Mary, S., Ratinger, T., Skuras, D., Balamou, E., Cardenete, M. A., & Gomez y Paloma, S. (2013). Ex-Ante Analysis of the Regional Impacts of the Common Agricultural Policy: A Rural–Urban Recursive Dynamic CGE Model Approach. *European Planning Studies*, 1–26.

ESRI, (2012). ArcMap. Environmental Systems Research Institute. Redlands, CA.

EU Regulation 1307/2013 of 17 December 2013 establishing rules for direct payments to farmers under support schemes within the framework of the common agricultural policy and repealing Council Regulation (EC) No 637/2008 and Council Regulation (EC) No 73/2009 [2013] OJ L347/608.

Evans, R. (2002). An alternative way to assess water erosion of cultivated land – field-based measurements: and analysis of some results. *Applied Geography*, *22*(2), 187–207.

FAO (Food and Agricultural Organisation of the United Nations). (2009). The Agricultural Statistics of the Food and Agricultural Organisation of the United Nations. *FAOSTAT*. Available at: http://faostat.fao.org/ [Accessed 28/02/2014]

Feld, C. K., da Silva, P. M., Sousa, J. P., de Bello, F., Bugter, R., Grandin, U., Hering, D., Lavorel, S., Mountford, O., Pardo, I., Partel, M., Rombke, J., Sandin, L., Jones, K. B., & Harrison, P. (2009). Indicators of biodiversity and ecosystem services: a synthesis across ecosystems and spatial scales. *Oikos*, *118*, 1862–1871.

Firbank, L. G., Petit, S., Smart, S., Blain, A., & Fuller, R. J. (2008). Assessing the impacts of agricultural intensification on biodiversity: a British perspective. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *363*(1492), 777–787.

Firbank, L., Bradbury, R. B., McCracken, D. I., & Stoate, C. (2013). Delivering multiple ecosystem services from Enclosed Farmland in the UK. *Agriculture, Ecosystems & Environment*, *166*, 65–75.

Foley, J. A., Defries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. A., Kucharik, C. J., Monfreda, C., Patz, J. A., Prentice, C., Rumankutty, N., & Snyder, P. K. (2005). Global consequences of land use. *Science*, *309*, 570–574.

Forestry Commission, (2010). *Wood as fuel: technical supplement for fuel suppliers*. Bristol, Forestry Commission.

GWCT (Game and Wildlife Conservation Trust). (2013). Farming at Allerton. [Online] Available at: http://www.gwct.org.uk/allerton/farming-at-allerton/ [Accessed 15/06/2013]

Grant, W. (2010). Policy Instruments in the Common Agricultural Policy. *West European Politics*. 33(1), 22-38.

Gregory, R. D., Noble, D., & Custance, J. (2004). The state of play of farmland birds: population trends and conservation status of lowland farmland birds in the United Kingdom. *Ibis*, *146*, 1–13.

Griffin, M. L., Beasley, D. B., Fletcher, J. J., & Foster, G. R. (1988). Estimating soil loss on topographically non-uniform field and farm units. *Journal of Soil and Water Conservation*, *43*(4), 326–331.

Haines-Young, R. (2009). Land use and biodiversity relationships. *Land Use Policy*, *26*, 178–S186.

Hein, L., van Koppen, K., de Groot, R. S., & van Ierland, E. C. (2006). Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics*, *57*(2), 209–228.

Henderson, I. G., Cooper, J., Fuller, R. J., & Vickery, J. (2000). The relative abundance of birds on set-aside and neighbouring fields in summer. *Journal of Applied Ecology*, *37*(2), 335–347.

IBM (IBM Corporation). (2012). SPSS Statistics. Portsmouth, UK.

Johnes, P. J., & Heathwaite, A. L. (1997). Modelling the impact of land use change on water quality in agricultural catchments. *Hydrological Processes*, *11*(3), 269–286.

Julien, P. Y., & Simons, D. B. (1985). Sediment transport capacity of overland flow. *Transactions of the American Society of Agricultural Engineers*, 28(3), 755-762.

Karali, E., (2012). *Investigating the effect of farmer land-use decisions on rural landscapes using an agent-based model approach*. Ph.D Thesis, The University of Edinburgh.

Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E. D., Clough, Y., Díaz, M., Gabriel, D., Holzschuh, A., Knop, E., Kovács, A., Marshall, E. J. P., Tscharntke, T., & Verhulst, J. (2009). On the relationship between farmland biodiversity and land-use intensity in Europe. *Proceedings of the Royal Society B: Biological Sciences*, *276*(1658), 903–909.

Kravchenko, A. N., Bullock, D. G., & Boast, C. W. (2000). Joint Multifractal Analysis of Crop Yield and Terrain Slope. *Agronomy Journal*, 92(6), 1279-1290.

Laterra, P., Orúe, M. E., & Booman, G. C. (2012). Spatial complexity and ecosystem services in rural landscapes. Agriculture, Ecosystems & Environment, 154, 56–67.

Lavalle, C., Baranzelli, C., Mubareka, S., Gomes, C. R., Hiederer, R., Batista e Silva, F., & Estreguil, C. (2011). *Implementation of the CAP policy options with the land use modelling platform: A first indicator based analysis*. Luxembourg, European Commission.

MA (Millennium Ecosystem Assessment). (2005). Ecosystems and human well-being: synthesis. World Resources Institute, Washington DC.

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J. P., Grizzetti, B., Drakou, E. G., Notte, A. L., Zulian, G., Bouraoui, F., Paracchini, M., Braat, L., & Bidoglio, G. (2012). Mapping ecosystem services for policy support and decision making in the European Union. *Ecosystem Services*, *1*(1), 31–39.

McCool, D., Forster, G., & Weesies, G. (1997). Slope length and steepness factors (LS). In K. Renard, G. Foster, G. Weesies, D. McCool, & D. Yoder (Eds.), *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE).* (101–141). Washington DC: United States Department of Agriculture.

McMahon, B. J., Whelan, J., Kirwan, L., & Collier, M. (2005). Farmland birds and the field boundary evaluation and grading system in Ireland. *The Irish Journal of Agri-Environmental Research*, *4*(7), 67–77.

Metzger, M. J., Rounsevell, M. D. A., Acosta-Michlik, L., Leemans, R., & Schröter, D. (2006). The vulnerability of ecosystem services to land use change. *Agriculture, Ecosystems & Environment*, *114*, 69–85.

Moore, I. D., & Wilson, J. P. (1992). Length-slope factors for the Revised Universal Soil Loss Equation: Simplified method of estimation. *Journal of Soil and Water Conservation*, *47*(5), 423–428.

Natural England, (2014). *Environmental Stewardship*. [Online] Available at http://www.naturalengland.org.uk/ourwork/farming/funding/es/. [Accessed 15/04/2014].

Ockenden, M. C., Deasy, C., Quinton, J. N., Surridge, B., & Stoate, C. (2014). Keeping agricultural soil out of rivers: Evidence of sediment and nutrient accumulation within field wetlands in the UK. *Journal of Environmental Management*, *135*, 54–62.

Oliver, B. G., Cosgrove, E. G., & Carey, J. H. (1979). Effect of suspended sediments on the photolysis of organics in water. *Environmental Science & Technology*, *13*(9), 1075–1077.

Overmars, K. P., Schulp, C. J. E., Alkemade, R., Verburg, P. H., Temme, A. J. A. M., Omtzigt, N., & Schaminée, J. H. J. (2014). Developing a methodology for a species-based and spatially explicit indicator for biodiversity on agricultural land in the EU. *Ecological Indicators*, *37*, 186–198.

Peh, K. S.-H., Balmford, A., Bradbury, R. B., Brown, C., Butchart, S. H. M., Hughes, F. M. R., Stattersfield, A., Thomas, D. H. L., Walpole, M., Bayliss, J., Gowing, D., Jones, J. P. G., Lewis, S. L., Mulligan, M., Pandeya, B., Stratford, C., Thompson, J. R., Turner, K., Vira, B., Wilcock, S., Birch, J. C. (2013). TESSA: A toolkit for rapid assessment of ecosystem services at sites of biodiversity conservation importance. *Ecosystem Services*, *5*, 51–57.

Perrings, C., Duraiappah, A., Larigauderie, A., & Mooney, H. (2011a). The Biodiversity and Ecosystem Services Science-Policy Interface. *Science*, *331* (6021), 1139–1140.

Perrings, C., Naeem, S., Ahrestani, F. S., Bunker, D. E., Burkill, P., Canziani, G., Elmqvist, T., Fuhrman, J. A., Jaksic, F. M., Kawabata, Z., Kanzig, A., Mace, G. M., Mooney, H., Prieur-Richard, A., Tschirhart, J., & Weisser, W. (2011b). Ecosystem services, targets, and indicators for the conservation and sustainable use of biodiversity. *Frontiers in Ecology and the Environment*, *9*(9), 512–520.

Plieninger, T., Schleyer, C., Schaich, H., Ohnesorge, B., Gerdes, H., Hernández-Morcillo, M., & Bieling, C. (2012). Mainstreaming ecosystem services through reformed European agricultural policies. *Conservation Letters*, *5*(4), 281–288.

Posthumus, H., Rouquette, J. R., Morris, J., Gowing, D. J. G., & Hess, T. M. (2010). A framework for the assessment of ecosystem goods and services; a case study on lowland floodplains in England. *Ecological Economics*, *69*(7), 1510–1523.

Power, A. G. (2010). Ecosystem services and agriculture: tradeoffs and synergies. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *365*, 2959–2971.

Renard, K. G., Foster, G. R., Weesies, G. A., & Porter, J. P. (1991). RUSLE: Revised universal soil loss equation. *Journal of Soil and Water Conservation*, *46* (1), 30–33.

Rounsevell, M. D. A., Ewert, F., Reginster, I., Leemans, R., & Carter, T. R. (2005). Future scenarios of European agricultural land use. *Agriculture, Ecosystems & Environment*, *107*(2-3), 117–135.

Rounsevell, M. D. A., Audsley, E., & Mortimer, D. J. R. (2007). *The impact of the Common Agricultural Policy on land use in Europe*. *Vol. IV*. In: Verheye, W. H. (2007). *Land use, land cover and soil sciences*. [Online]. Available at http://www.eolss.net/Land-Use-Land-Cover-Soil-Sciences.aspx. [Accessed 28/02/2014].

Russell, M., Walling, D., & Hodgkinson, R. (2001). Suspended sediment sources in two small lowland agricultural catchments in the UK. *Journal of Hydrology*, *252*(1-4), 1–24.

Siriwardena, G. M., Baillie, S. R., Buckland, S. T., Fewster, R. M., Marchant, J. H., & Wilson, J. D. (1998). Trends in the abundance of farmland birds: a quantitative comparison of smoothed Common Birds Census indices. *Journal of Applied Ecology*, *35*(1), 24–43.

Swinton, S. M., Lupi, F., Robertson, G. P., & Hamilton, S. K. (2007). Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits. *Ecological Economics*, *64*, 245–252.

Tallis, H., Kareiva, P., Marvier, M., & Chang, A., (2008). An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences*, 105(28), 9457-9464.

TEEB (The Economics of Ecosystems and Biodiversity). (2010). The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundation. Earthscan, Cambridge.

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., & Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. *Ecology Letters*, *8*(8), 857–874.

UKNEA (UK National Ecosystem Assessment). (2011). *The UK National Ecosystem Assessment: Technical Report*. Cambridge.

Ulén, B., Bechmann, M., Fölster, J., Jarvie, H. P., & Tunney, H. (2007). Agriculture as a phosphorus source for eutrophication in the north-west European countries, Norway, Sweden, United Kingdom and Ireland: a review. *Soil Use and Management*, *23*, 5–15.

UNEP (United Nations Environment Programme). (2011) *Ecosystems for water and food security*. Nairobi, UNEP.

USDA (United States Department of Agriculture). (2014). *Data and statistics*. [Online] Available at http://www.usda.gov/wps/portal/usda/usdahome?navid=DATA_STATISTICS. [Accessed 08/10/2013].

Vandaele, K., & Poesen, J. (1995). Spatial and temporal patterns of soil erosion rates in an agricultural catchment, central Belgium. *CATENA*, *25*(1-4), 213–226.

Vickery, J. A., Bradbury, R. B., Henderson, I. G., Eaton, M. A., & Grice, P. V. (2004). The role of agri-environment schemes and farm management practices in reversing the decline of farmland birds in England. *Biological Conservation*, *119*(1), 19–39.

Walling, D. E. (2005). Tracing suspended sediment sources in catchments and river systems. *The Science of the Total Environment*, *344*(1-3), 159–84.

Walling, D. E., Collins, A. L., & Stroud, R. W. (2008). Tracing suspended sediment and particulate phosphorus sources in catchments. *Journal of Hydrology*, *350*(3-4), 274–289.

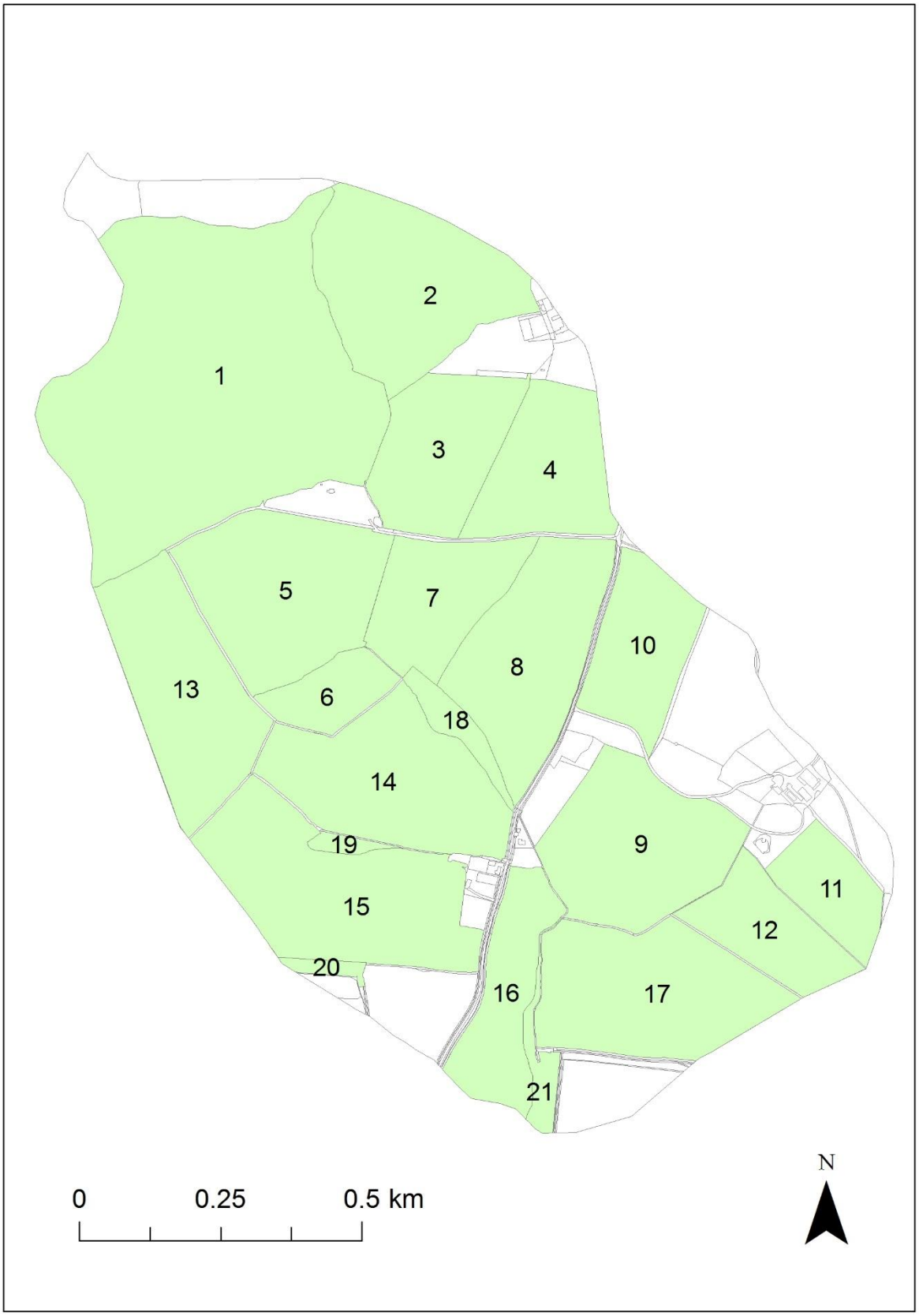
Wood, F. L., Heathwaite, A. L., & Haygarth, P. M. (2005). Evaluating diffuse and point phosphorus contributions to river transfers at different scales in the Taw catchment, Devon, UK. *Journal of Hydrology*, *304*(1-4), 118–138.

Wretenberg, J., Pärt, T., & Berg, Å. (2010). Changes in local species richness of farmland birds in relation to land-use changes and landscape structure. *Biological Conservation*, *143*(2), 375–381.

Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., & Swinton, S. M. (2007). Ecosystem services and dis-services to agriculture. *Ecological Economics*, *64*, 253–260.

Zhang, H., Yang, Q., Li, R., Liu, Q., Moore, D., He, P., Ritsema, C. J., & Geissen, V. (2013). Extension of a GIS procedure for calculating the RUSLE equation LS factor. *Computers & Geosciences*, *52*, 177–188.

Appendix 1 – Analysis Zones within the study area



Zone reference

ID	Field	Area (ha)	Land Use	EFA
1	Launde wood	25.71	Woodland	0%
2	Back Park	8.93	Pasture	0%
3	Mill field	5.64	Pasture	0%
4	Home Slope	5.21	Pasture	0%
5	Birchberries	8.51	Pasture	0%
6	Six Acres	2.17	Pasture	0%
7	Middle field	4.74	Pasture	0%
8	Roadside	7.67	Pasture	0%
9	Big pasture	7.73	Pasture	0%
10	Upper pasture	4.58	Pasture	0%
11	Small pasture	2.78	Pasture	0%
12	Lower pasture	3.74	Pasture	0%
13	Cawthorn	7.79	Crop	4.9%
14	Spring field	8.28	Crop	7.2%
15	School field	9.87	Crop	21.6%
16	Alan Townsend	4.36	Crop	8.1%
17	Copthill	8.43	Crop	3.3%
18	Spring wood	1.10	Woodland	0%
19	Beetle wood	0.36	Woodland	0%
20	Base wood	0.38	Woodland	0%
21	Bug wood	0.90	Woodland	0%

Appendix 2 – Species richness category birds

ID	Name	Species	No. of sightings
1	Blackbird	*Turdus merula*	28
2	Blackcap	*Sylvia atricapilla*	80
3	Blue Tit	*Parus caeruleus*	10
4	Chaffinch	*Fringilla coelebs*	64
5	Chiffchaff	*Phylloscopus collybita*	48
6	Dunnock	*Prunella modularis*	32
7	Great Tit	*Parus major*	12
8	Robin	*Erithacus rubecula*	51
9	**Skylark**	***Alauda arvensis***	**30**
10	Song Thrush	*Turdus philomelos*	24
11	**Whitethroat**	***Sylvia communis***	**13**
12	Wren	*Troglodytes troglodytes*	60
13	**Yellowhammer**	***Emberiza citrinella***	**14**
14	**Yellow Wagtail**	***Motacilla flava***	**28**

Birds in bold feature on the UK Farmland Bird Index (FBI) and were analysed separately.

Appendix 3 – Tests for normality and One-way ANOVA statistical analysis

Tests of Normality

	Land Use	Kolmogorov-Smirnov[a]			Shapiro-Wilk		
		Statistic	df	Sig.	Statistic	df	Sig.
Cultivated Goods	Arable	.253	5	.200*	.935	5	.631
	Pasture	.211	11	.187	.838	11	.030
	Woodland	.249	5	.200*	.936	5	.635

*. This is a lower bound of the true significance.

a. Lilliefors Significance Correction

Tests of Normality

	Land Use	Kolmogorov-Smirnov[a]			Shapiro-Wilk		
		Statistic	df	Sig.	Statistic	df	Sig.
Water Quality	Arable	.340	5	.059	.712	5	.013
	Pasture	.167	11	.200*	.873	11	.085
	Woodland	.309	5	.134	.792	5	.070

*. This is a lower bound of the true significance.

a. Lilliefors Significance Correction

Tests of Normality

	Land Use	Kolmogorov-Smirnov[a]			Shapiro-Wilk		
		Statistic	df	Sig.	Statistic	df	Sig.
Biodiversity	Arable	.213	5	.200*	.963	5	.826
	Pasture	.176	11	.200*	.941	11	.531
	Woodland	.127	5	.200*	.999	5	1.000

*. This is a lower bound of the true significance.

a. Lilliefors Significance Correction

ANOVA

Cultivated Goods

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	2.560	2	1.280	252.371	.000
Within Groups	.091	18	.005		
Total	2.651	20			

Multiple Comparisons

Dependent Variable: Cultivated Goods

Sidak

(I) Land Use	(J) Land Use	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Lower Bound	Upper Bound
Arable	Pasture	.82227*	.03841	.000	.7212	.9233
	Woodland	.81415*	.04504	.000	.6956	.9327
Pasture	Arable	-.82227*	.03841	.000	-.9233	-.7212
	Woodland	-.00812	.03841	.995	-.1092	.0929
Woodland	Arable	-.81415*	.04504	.000	-.9327	-.6956
	Pasture	.00812	.03841	.995	-.0929	.1092

*. The mean difference is significant at the 0.05 level.

ANOVA

Biodiversity

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	.389	2	.195	3.216	.064
Within Groups	1.090	18	.061		
Total	1.479	20			

ANOVA

Water Quality

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	2.505	2	1.252	79.829	.000
Within Groups	.282	18	.016		
Total	2.787	20			

Multiple Comparisons

Dependent Variable: Water Quality

Sidak

(I) Land Use	(J) Land Use	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Lower Bound	Upper Bound
Arable	Pasture	-.81218*	.06756	.000	-.9899	-.6344
	Woodland	-.80800*	.07922	.000	-1.0164	-.5996
Pasture	Arable	.81218*	.06756	.000	.6344	.9899
	Woodland	.00418	.06756	1.000	-.1736	.1819
Woodland	Arable	.80800*	.07922	.000	.5996	1.0164
	Pasture	-.00418	.06756	1.000	-.1819	.1736

*. The mean difference is significant at the 0.05 level.

Appendix 4 – Final ecosystem service values for analysis zones

ID	Field	Area (%)	Land Use	EFA	Biodiversity	Cultivation/ha	Water quality
1	Launde wood	20%	Woodland	0%	1.00	0.00	0.96
2	Back Park	7%	Pasture	0%	0.25	0.01	0.87
3	Mill field	4%	Pasture	0%	0.13	0.01	0.82
4	Home Slope	4%	Pasture	0%	0.63	0.01	0.90
5	Birchberries	7%	Pasture	0%	0.50	0.02	0.93
6	Six Acres	2%	Pasture	0%	0.13	0.01	0.94
7	Middle field	4%	Pasture	0%	0.00	0.01	0.92
8	Roadside	6%	Pasture	0%	0.75	0.02	0.93
9	Big pasture	6%	Pasture	0%	0.50	0.00	0.84
10	Upper pasture	4%	Pasture	0%	0.38	0.00	0.80
11	Small pasture	2%	Pasture	0%	0.13	0.00	0.75
12	Lower pasture	3%	Pasture	0%	0.25	0.00	0.63
13	Cawthorn	6%	Arable	<5%	0.63	0.95	0.03
14	Spring field	6%	Arable	5-10%	0.88	0.95	0.00
15	School field	8%	Arable	>10%	0.63	0.92	0.01
16	Alan Townsend	3%	Arable	5-10%	0.25	1.00	0.14
17	Copthill	7%	Arable	<5%	0.50	0.77	0.00
18	Spring wood	1%	Woodland	0%	0.75	0.25	1.00
19	Beetle wood	0%	Woodland	0%	0.50	0.25	0.49
20	Base wood	0%	Woodland	0%	0.25	0.25	0.81
21	Bug wood	1%	Woodland	0%	0.63	0.25	0.96

Appendix 5 – Conversion scenarios to achieve the Lavalle (2011) land use change scenario.

ID	Conversion scenario	**Area%**	WQ	BD	CG	WQ_Conversion	BD_Conversion	CG_Conversion
1	---	20%	0.855878	0.859031236	**185882304.6**			
2	1,2,4,5,7,8,13,21,24	**7%**	0.823566	**0.72**	3912709.328	0.458134	0.777016454	239741755.2
3	1,2,3,10,11,16,19,20	**4%**	0.806301	**0.611111111**	2252124.349	0.432056	0.777016454	151519090.7
4	4,5,6,10,12,17,19,20	**4%**	0.835778	**0.823529412**	2107414.298	0.470802	0.777016454	139803476.7
5	1,3,4,6,7,9,14,22,25	**7%**	0.844637	**0.78200692**	3987970.036	0.484765	0.777016454	228586212.9
6	10,11,12,16,17,18,21,22,23,24,25,26	**2%**	0.843332	**0.444444444**	1729301.215	0.481352	0.777016454	58191763.5
7	7,8,9,11,12,18,19,20	**4%**	0.844915	**0.8515625**	3929567.797	0.484587	0.777016454	127270942.7
8	16,17,18,19,21,22,23	**6%**	0.850195	**0.625**	864941.0904	0.491101	0.777016454	205832430.8
9	---	6%	**0.53539**	0.67768595	234362000			
10	---	4%	**0.570504**	0.72	162404400			
11	---	2%	**0.523853**	0.850661626	222812200			
12	---	3%	**0.526555**	0.836734694	274873000			
13	---	6%	**0.523681**	0.8	244188000			
14	16,17,18,20,24,25,26	**6%**	0.812856	**0.808888889**	0	0.442805	0.777016454	222451444.7
15	10,11,12,13,14,15	**8%**	0.73992	**0.716049383**	0	0.360833	0.777016454	265010019.3
16	13,14,15,21,22,23,24,25,26	**3%**	0.783	**0.59375**	0	0.402461	0.777016454	116975231.4
17	2,3,5,6,8,9,15,23,26	**7%**	0.800806	**0.765432099**	0	0.427268	0.777016454	226468670.9
18	---	1%	0.869528	0.816608997	**7931489.953**			
19	---	0%	0.691948	0.8046875	**2586019.427**			
20	---	0%	0.803855	0.666666667	**2712339.507**			
21	---	1%	0.855061	0.845993757	**6473387.253**			