**The Changing role of Land Use and subsequent biodiversity of hedgerows in the Knepp Castle Estate, West Sussex. From 1870 to present.**


## Department of Geography and Planning

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## Abstract

This study aims to assess the implications of land use change on the biodiversity of hedgerows by taking an environmental history perspective of The Knepp Castle Estate in West Sussex, from 1870 -2018. By combining data from fieldwork, historical and modern day maps and also aerial photographs, an indication of previous hedgerow structure has been determined, with overall increases in both hedgerow length and species richness per m2 of hedgerow observed from 1870 – 2018. Drivers of land use change have also been examined, and are predominantly reduced to external agricultural policy application dependant on social contexts. The most recent adoption of rewilding on The Knepp Castle Estate is more indicative of changing ecosystem service appreciation as well as economic climates, and appears to be the most appropriate land management strategy for the study area in terms of ecosystem service benefits.

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## List of abbreviations

- AES: Agri-environmental Scheme
- BAP: Biodiversity Action Plan
- CAP: Common Agricultural Policy
- CBD: Convention on Biodiversity
- CSR: Percentage change in species richness per m2 of hedge.
- EEC: European Economic Community
- EU: European Union
- TKCE: The Knepp Castle Estate

# 1.0 Introduction:

## 1.1 Biodiversity, Ecosystem Services and Land Use Change

The international importance of the state of biodiversity is not absent in the surrounding literature. Paoletti (1999) encourages the use of biodiversity as a tool for landscape observation, because reduced diversity is strongly correlated to environmental degradation. It is therefore important to monitor biodiversity levels as there is no way to impede its loss if it is not quantified (Buckland et al, 2015). In addition, the relevance and effectiveness of land use and planning policy cannot be realised without such studies (Van Vliet, 2015). Multilateral agreements such as The Convention on Biological Diversity (CBD) - signed by 150 countries at the 1992 Earth Summit - prove how biodiversity is invaluable to society. Its aims are too conserve levels of diversity in order to be able to equitably share the benefits created by a healthy ecosystem. However, the CBD warns that their current projections suggest that the 2020 Aichi targets for biodiversity conservation will not be met (CBD, 2014).

Ecosystem services are invaluable to society in a variety of ways, and high levels of biodiversity are in turn a benefit to ecosystem services (Faith et al 2010). Ecosystem services that are affected by a loss in biodiversity include provisioning services that provide vital resources such as food, water and medicine (Chaplin III et al, 2000). Due to the increase of land use intensity, regulatory ecosystem services have decreased by 60% in the last 50 years (Benagas et al, 2007), decreasing the resilience of ecosystems. Chaplin III et al (2000) put forward the idea that strong levels of biodiversity is a benefit to the resilience of ecosystems; for example, levels of primary production and nutrient retention are positively correlated with levels of species richness. It is clear from multiple studies that biodiversity loss has a negative effect on ecosystem properties (Godbold and Solan, 2009), and since the UK has suffered significantly higher losses of biodiversity compared to the world average (Hayhow et al 2016), there is significant incentives to investigate drivers of biodiversity change.

The most dramatic changes in land use have occured in recent decades as landowners are driven by various socio-economic pressures to maximise their ecosystem service efficiencies (Lambin and Meyfroidt, 2010). Subsequently, the UK's current rural landscape is '*unrecognisable from what our grandparents would have seen*' (Tree, 2018, pg. 3). There are numerous cases that prove that changes in land use is one of the biggest drivers of biodiversity change (Gerard et al, 2011, Petit et al, 2011, Falucci et al 2007). The BIOPRESS ('Linking Pan-European Land Cover Change to Pressures on Biodiversity') is a European Commission funded ' project (Gerard et al 2010) that concluded that the '*clearest indication of a change in the environment is when there is a change in land cover*' (pg. 185) through assessing historical land cover change for the purpose of measuring changes in biodiversity. The findings from Chaplin III et al (2010) support this, predicting that land use change will have the largest global impact on biodiversity by the year 2100. Environmental impacts of land use change are global - from the modification of atmospheric composition to soil quality degradation. The declines in biodiversity are one of the primary results of land use change through the loss and fragmentation of habitat and depletion of the ecosystem (Foley et al 2005). Since one-half to one-third of global ecosystem production is dedicated to human use, the maintenance of their service is vital to maintain the socio-economic benefits. Land use change may occur at the local scale, but negative impacts can be accounted for worldwide (Foley et al 2005).

## 1.2 Aims and Objectives

This project's main aim centres on the fluctuations in species richness levels from 1870 in relation to land use change in the study area. The aims and objectives covered are:

- To review land cover change within the study area between 1870 - 2018.
    - *Combining archival and digital data to assess land cover change in the study area and identify social, economic and political drivers behind this.*

- Obtaining an estimate of hedgerow length change between 1870 - 2018.
    - *Obtained through ArcGIS, the influence of land use changes on this aim will be considered.*

- Develop an understanding of the variance of Species Richness levels between 1870 - 2018 within the study area.

- *Collecting Species Richness levels of present day hedgerows and using this to observe biodiversity changes through hedgerow length changes from 1870 - 2018.*

- Identify relationships between land use change and species richness levels from 1870 - 2018 on The Knepp Castle Estate.

The use of archival, digital and fieldwork assessment to obtain an environmental history of the study area is a novel take on how land use change can affect biodiversity in the UK and similar ecological settings within Western Europe.

## 1.3 The study area

The Knepp Castle Estate (TKCE) covers 1,415 hectares and is situated due east of the hamlet of Shipley in West Sussex. The estate is part of a unique program in the UK, being the only large scale attempt to apply an emerging rewilding ethos that is unfolding in Europe.

Before the start of the rewilding project at the beginning of the millennium, the estate was intensively farmed - despite lying on heavy Low Weald Clay - increasing after the outbreak of the Second World War. In the 1940s, every part of the estate was put under cultivation to aid the war effort as an example to the rest of the parish by Sir John Burrell, including the 58 hectares of Repton style parkland which surrounds the castle (Tree, 2018). From 1939 - 1945 TKCE was requisitioned by the War Office, and used as an HQ for Canadian Infantry and Armoured divisions.

The Estate is divided into three blocks, with the Northern and Middle blocks separated by the A272, whilst the Southern block is divided by a small country lane. However, the blocks are also divisible by previous land use. The Northern block was predominantly used for dairy farming, accommodating for large grazers. In comparison, the Southern block has been intensively reworked to make it a suitable location for arable culture. The Middle block has predominantly been the site of parkland and well – kept pasture land. Today though, 796 acres have been taken out of arable cultivation due to tough external pressures (Greenaway, 2006).

TKCE was awarded a Higher Level Stewardship AES (Agri-Environmental Scheme) to support the creation of an extensive grazing system on its marginal land. In 2000 the farm's dairy herd was sold at the same time as the arable contract came to an end. However, TKCE is unusual in that its historic field system has largely been retained despite years of agricultural policy reform such as The Common Agricultural Policy (CAP). Gerard et al (2011) recognize that the protection of our ecosystems requires monitoring, a goal that coincides with the aims of the rewilding project on TKCE; to record and evaluate changes in the biodiversity and vegetation structure (Greenaway, 2006.) TKCE has experienced a variety of land cover change in its history, and so an insight into what effect this may have had on its immediate biodiversity as has similarly been undertaken by others (Gerard et al, 2011, Petit et al, 2011, Falucci et al, 2007) is of value.


*Figure 1.3.1:* Taken by author on TKCE.

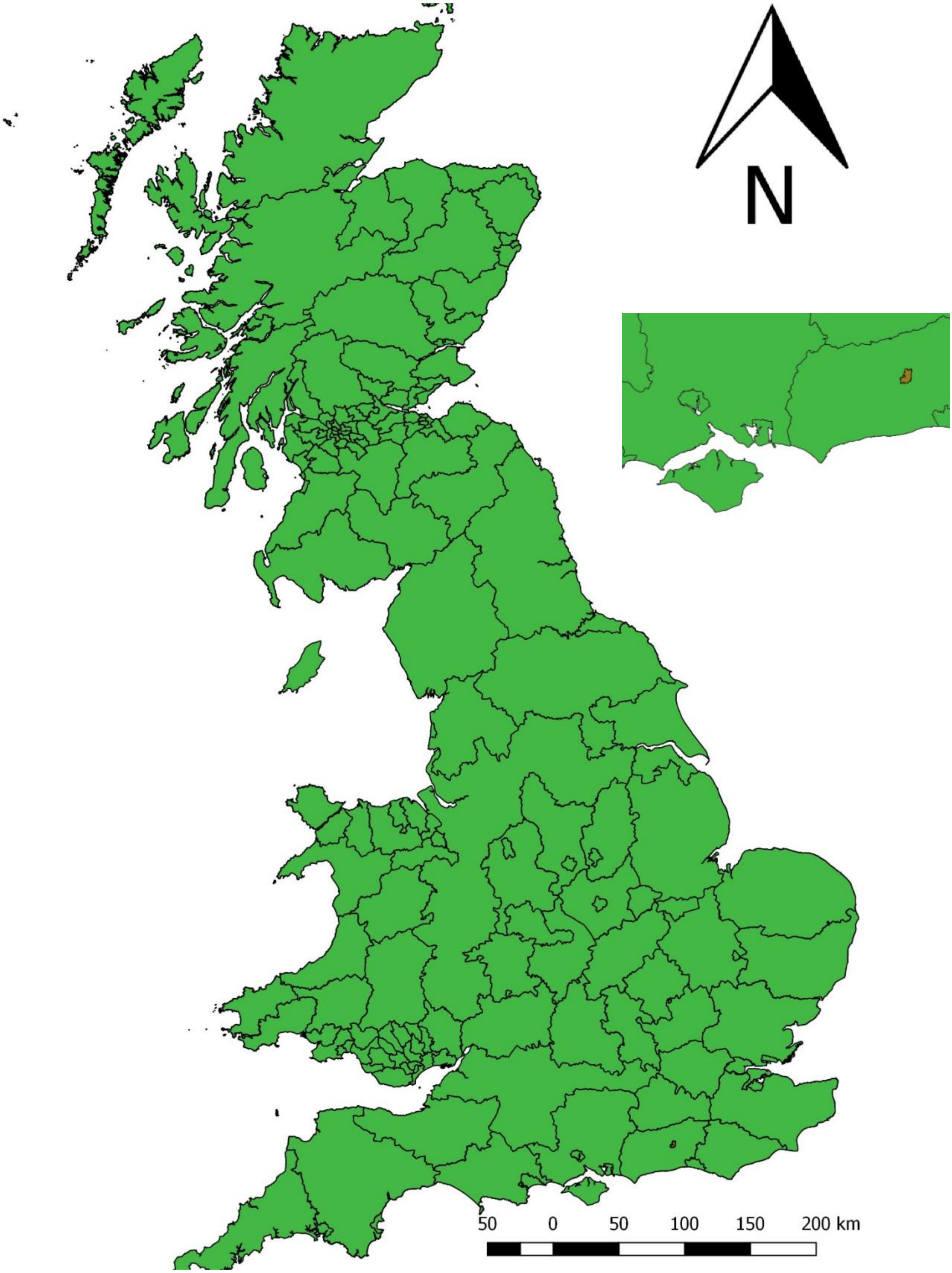



*Figure 1.3.2:* Map of study area, with zoomed in section presenting TKCE position within West Sussex. Composed in QGIS.

# 2.0 Literature Review

## 2.1 Land use change in the agricultural context

Land use change in the agricultural setting regarding biodiversity has been the subject of many reviews (Swetnam, 2007, Dallimer et al 2010, Klejn and Sutherland, 2003), but less common is relating these issues to the exogenous (external economic and political drivers of land transitions) and endogenous (the local and social drivers) from which they are born (Lambin and Meyfroidt, 2010). Assessing agricultural land use change from differing points in time gives a more dynamic idea of changing biodiversity levels (Dallimer et al, 2010) even though recording the state of ecology within the UK is not a new feature.

Extensive overviews of rural land use was first established in the 1930s by Dudley L Stamp, with secondary land utilisation surveys appearing in the 1960s (Swetnam, 2007). More recently, Countryside Surveys serve as tools for monitoring and evaluating land use across the UK. As the majority of the UK's land cover falls under agricultural purpose, the changes that occur often dictate processes that occur in other environmental systems (Dallimer et al, 2010). Van Vliet et al (2015) express how agricultural land use change is less black and white than other land use changes; there are four predominant transformations including increases and decreases of agricultural intensity, and increases and decreases of absolute land area. TKCE has experienced both agricultural intensification and abandonment.

Petit et al (2007) found in their study that in five of 13 ecological regions, the intensification of farming impacts habitats more than any other anthropogenic pressure. Habitat depletion is catalyzed by pollutants from chemical inputs and the removal of small biotypes such as hedgerows. This is especially relevant as 50% of European wildlife species are found on farmland habitat (Hicks et al, 2013), and agriculture takes up 38% of ice free terrestrial areas (Querioz et al, 2014). International conditions will elicit different responses in farmers, and land use changes are also subject to individual characteristics; with younger farmers often moving towards efficiency increasing methods (Van Vliet et al, 2015). The increased agricultural market in Europe also catalysed the intensification of farming methods, with a nationwide suffering of natural systems (Bengagas et al, 2007). Burel and Boudry (1995) add that anthropogenic land use change is especially detrimental to biodiversity as no natural cyclical behavior is seen; events such as fire, wind and pests cease in an agricultural setting leading to monocultures and a subsequent drop in biodiversity. TKCE is a distinctive example of what can be allowed to unfold if natural succession is given free rein, and as land cover change is often dependent on exogenic factors such as income and family structure (Burel and Boudry, 1995) this is of a national importance in economically trying times for farming families across the country.

The termination of intensive agriculture on TKCE may be the most dramatic form of land use change in its history. In Europe, nearly 1 million hectares of land are released from small scale agriculture each year (Helmer et al, 2015) due to socio-economic pressures. For example, the number of farmers employed in agriculture is related to economies of scale. As farms extensify, less individual businesses operate and so the numbers of farmers drop (Metzger et al, 2006). CAP has also aided this process with the transformation of agricultural land to forest since the 1990s as it's focus shifts away from intensive food production (Bengagas et al, 2007). It is thought that ecosystem quality decreases quicker within an arable context due to higher levels of production intensity, caused by artificial input (Reidsma et al, 2006). In this way, less habitat for biodiversity is lost to intensive methods of agriculture but the overall negative effect on the ecosystem as a whole is greater. TKCE has experienced a wide range of landscape alteration such as this, and so a study focusing on biodiversity levels in relation to land use change is appropriate.

## 2.2 The concept of rewilding

The substitution of agricultural production in favour of natural succession is a concept more popular in vast areas of North America, where rewilding has gained attention as an alternative regime for marginal agricultural land (Carver 2007) as it may be more profitable than agricultural production. This notion has been heavily influenced by the work of Frans Vera. Vera (2009) proposes that due to thousands of years of cultivation, the 'biodiversity baseline' of Europe has dropped dramatically. Vera's theory of shifting baseline syndrome relates the current work of conservationists in maintaining the modern landscape despite its negative effects on ecosystem health. This has occurred due to the heavy management of rural areas where natural disturbance has not been allowed to unfold (Fuhlendorf et al, 2009). Subsequently, a degraded ecosystem is considered normal (Vera, 2009). Rewilding aims to mimic the ecological baseline before the agricultural revolution of Neolithic times (Hodder et al, 2014). TKCE has therefore tried to adopt Vera's proposal of encouraging a wood-pasture landscape, controlled by the grazing of large herbivores such as horses, deer, and cattle and not micro-managing for specific targets like most of the commonly applied conservation schemes (Tree, 2018). The differentiation in diet ensures that not one vegetation type becomes dominant, and so an even mosaic of plants develop in a park-woodland like landscape, with the hope of producing higher levels of biodiversity.

Landscape–scale management practices such as rewilding are increasing, with the rise of landscape ecology as a developing discipline (Turner, 2005). Stakeholder perceptions of conservation and land use are now influenced by meta-populations and ecosystems as a whole, with conservation efforts no longer focusing on a handful of vulnerable species (Hodder et al, 2014). Although the term rewilding can mean different things in different parts of the world, European directions sway towards an ecological state before the Holocene; before excessive land clearing and the extinction of keystone species due to human pressures (Jorgenson, 2015). The most notable case of success for a taxon replacement approach is the Oostvaardersplassen in Holland where a wood-pasture landscape supports many species that are under threat (Vera, 2010). Random disturbances caused by large herbivores ensure that disturbance dependent and disturbance sensitive species can both thrive in this environment and so biodiversity is increased (Fuhlendorf et al, 2009). Rewilding Europe is a non-profit organization that began in 2010 and aims to let nature take the lead in 1 million hectares of land by 2020 (Helmer et al, 2015). Hodder et al (2014) assessed how this type of landscape management on TKCE has an overwhelming benefit to ecosystem services (see Figure 2.2.1), and the projected increase in Biodiversity Action Plan (BAP) habitat area is expected to increase by 622%. Applying a novel method such a rewilding to a national problem that is seen within the current agricultural sector makes TKCE an ideal site of academic investigation in relation to land use change.

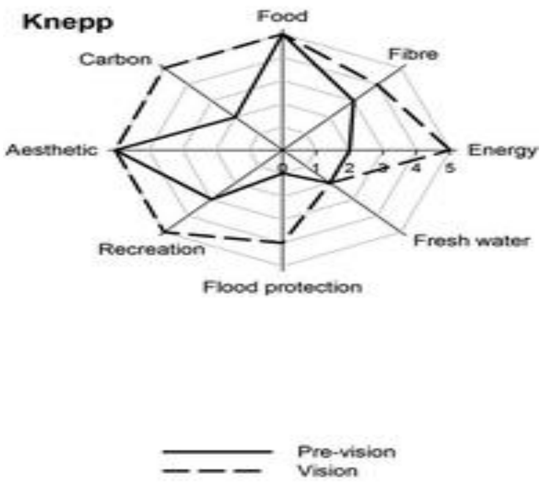



*Figure 2.2:* Taken from Hodder et al (2014). Depicts changes in ecosystem services on TKCE since the rewilding initiative.

## 2.3 Historical significance

Lunt and Spooner (2005) emphasize the importance of an historical grounding in environmental research as past anthropogenic activities are what shape our ecosystems today. In addition, changes to landscapes are often imperceptible under short term observation periods (Haase et al, 2007) and key drivers, trends and levels of equilibrium and disequilibrium may be overlooked (Petit and Lambin, 2002). Ernoult and Alard (2011) reveal that there may be a time lag between the cause and effects of non-equilibrium landscapes such as within the agricultural setting. Therefore, the biodiversity – reflected through species richness and distribution - may in fact reflect past landscape patterns. This is reiterated by Gonzalez (2013), who describes the extinction debt as something experienced by communities long after the initial habitat disturbance has occurred as populations decay over time. Approaching current states of the environment from a historical perspective requires a close look at the last 200 years due to the correlation between environmental degradation, industrial expansion, land use intensification and changing social contexts on TKCE (Felucci et al, 2007), and adheres to the methodological constraints associated with obtaining accurate conclusions from archival resources (Slenicka, 2009). Therefore, an overview of the environmental history can help to establish the current ecological situation at TKCE. The research in this topic is therefore of interest to historic and current environmentalists, particularly as it encases the emergent concept of rewilding. The historical findings of this project can be used to efficiently map the current state of nature elsewhere (Lunt and Spooner, 2005) and the effects of the recent developments will be a good indicator of the relative success of rewilding on the site's biodiversity compared to its past.

## 2.4 The role of hedgerows

The temporal nature of the investigation requires a bio-indicator that is consistent through time. Using plants as a biomonitor of environmental degradation is well accepted in the literature (Sawidis et al, 2011). Due to its agricultural history, TKCE has always had a strong network of hedgerows (100 – 120 km currently on the estate (Sussex Hedgerow Inventory Project, 2011)) and so the use of this type of vegetation as a passive sampler of environmental recording has benefits spatially and temporally. Investigating biodiversity around hedgerows can allow us to build a picture of how the changing land use affects biodiversity: Le Coeur et al's (2002) study revealed that adjacent land use change was the predominant factor in influencing the diversity of assemblages, even more so than direct intervention on the field boundary. Although many of Great Britain's hedgerows can be dated back to Neolithic agriculture (Le Coeur et al, 2002), and many appearing around the 18th and 19th century as a result of land enclosures (Marshall and Moonen, 2002), many were removed under the 1947 Agriculture Act after the Second World War food shortages (Robinson & Sutherland, 2002). Their presence (or absence) can be noted on archival maps and aerial photographs dating back to 1947, and this can aid in estimating past biological climates when combined with modern day hedgerow and biodiversity data.

It is well known that a loss in hedgerows often results in a loss of biodiversity (Macdonald and Jackson, 1999) because they provide an undisturbed and sheltered buffer from harsh agricultural practices (Merckx et al, 2012). Marshall and Moonen (2002) note how field margins can become ecotones; regions of marked ecological change and a difference in plant communities. The field margin ecotone will be susceptible to influences from the land use of the crop and the environment with the hedgerow boundary. Land use changes such as adopting rewilding may increase the scope of hedgerows, with some predictions estimating a thickening and succumbing of structure to scrubland. (Corby, 2010, Hicks et al, 2013). Therefore, the effect on land use change on the hedgerows themselves have a possibility of affecting the biodiversity on TKCE.

The site has attracted previous academic interest, and secondary data is available from an array of investigations including a study of the changing hedgerows from 2001 – 2015 (Eernisse, 2017). The Sussex Hedgerow Inventory Project (2007) ran a program using aerial photography to conclude an increase in hedgerow extent after the abandonment of agriculture. Studies assessing vegetation, insect and bird biodiversity in relation to the adopted grazing agenda have also been carried out (Corby 2010, Buttone 2016, Szota 2015) all of which provide valuable data and insight to the progress of the rewilding project thus far. In addition, Theresa Greenaway has conducted multiple baseline ecology reports on TKCE in order to monitor the changes that are unfolding as a result of the rewilding program.

# 3.0 Methodology

## 3.1 Historical data collection

Land cover change of the TKCE from the late 1800s was obtained through archival data collection. Two days were spent searching through the archives at the West Sussex County Records Office in Chichester in August 2018 to obtain this information. Obtained from this are aerial photographs capturing TKCE in 1947, 1971, 1981 and 1991. Land cover information for 1930, 1990, 2000, 2007 and 2015 was downloaded from EDINA Digimap, as well as data for County Series Maps for the 1870s, 1890s and 1910s. National Grid maps were used within the ranges of the 1960s and 1980s, all of which were composed using ArcGIS. Slenicka's (2009) study additionally utilised a mixture of historical maps, aerial photography and computer software to reconstruct landscapes of the past.

## 3.2 Biodiversity data collection

To gather data for present day levels of biodiversity on TKCE, fieldwork was undertaken over four days of September 2018. Species richness is a widely used and the most fundamental concept of diversity, being a count of the number of species in a given area (Peet, 1974), and mapping patterns of species richness can provide a basis for future land use planning (Gould, 2000). Quantifying species richness is important for comparisons between sites and to compare current and background rates of species extinction against this value (Gotelli and Colwell, 2001). Hedgerows have been used as a temporal reference point, and so species richness measurements were taken from hedgerows in order to build a picture of evolving biodiversity. Ernoult and Alard's (2011) study additionally used hedgerows to relate past levels of biodiversity, and combined the surveying of aerial photographs to aid with this.

Six hedges were selected at random from each of the Southern, Middle and Northern blocks of TKCE in order to gather an even spread of species richness throughout the whole study area. MacDonald and Jackson (1995) chose hedgerows to study in a similar way, and express how data from individual hedges can also be applied to a national context. The species richness of plants found within the herb layer of each hedge were recorded. Pereira and Cooper (2006) explain how the assessment of vascular plants within the herb layer of hedgerows will be an adequate indicator of biodiversity as they are the main primary producers in terrestrial ecosystems and are fundamental to ecosystem functioning. They even go as far to say that the diversity of plants are the best available predictors of diversity of other taxa. French and Cummins (2001) and Marshall and Moonen (2002) further this, explaining how the herb layer of hedgerows are subject to surrounding land use practices and should not be considered a separate ecotone, and so are a good indication of the influence of land use on biodiversity.

To keep with methodological consistency, species richness was recorded using a 1m x 1m quadrat on a parallel transect with the hedgerow, and measurements were taken at 10m intervals on this transect. In this way, multiple species richness recordings were obtained for one hedge. Species present within the quadrat at the herb layer level of the hedge were recorded at the 10m interval. Each side of the hedge was treated as a separate plot in order to establish the effects of the corresponding adjacent land use on species richness (Marshall and Moonen 2002). The combination of transect and quadrat in a systematic sampling method reduces bias when recording species richness in order to build a representative sample of the hedgerow. From this we can gather how the land use has affected the hedge itself as well as the surrounding biodiversity.

However, it is worth noting that obtaining a comprehensive species composition of a community is virtually impossible (Peet 1974), as most communities are too large for every individual to be identified. This results in sampling of communities and samples are subject to sampling error (Heltshe and Forrester, 1983). Furthermore, fixed, arbitrary samples are taken from a heterogeneous population that changes with time and space, yet samples are assumed representative of the whole population at one point in static time (Heltshe and Forrester, 1983).

In order to gauge how species richness has developed over time within TKCE, the average species richness of each hedge was measured against hedgerow length change. Six historical maps downloaded from EDINA Digimap were composed in ArcGIS, and the measuring tool was used to measure the same hedges through time using the six maps . The hedges measured were the same that were used to collect species richness values from, and were located using Google Maps at the time of fieldwork. The recorded hedge length measurement was multiplied by the average species richness or each hedge (see Appendix).

The percentage change in species richness (m2) (CSR) for whole hedgerows was calculated by multiplying the hedge length value obtained from ArcGIS for the different time periods recorded, by the average herb layer species richness value (Table 4.41) recorded during field work for each hedge. Rich et al (2000) concluded that it is acceptable to survey standardized sections of hedgerows and assume that biodiversity values can be applicable to the whole hedgerow.

The average species richness value for the herb layer of each hedge was multiplied by its length (which differed depending on the time period measured), resulting in a species richness (m2) value for the whole hedge. Subsequently, a value for species richness through the time periods is obtained. The equation was calculated for each hedgerow, depicting the species richness change through time by applying the following equation for differing years:

**Hedgerow Length (m) x Average Species Richness**

From this value, a percentage change for each hedgerow recorded is calculated from one period of time to another and is displayed in Figure (4.5).

## 3.3 Potential for inconsistencies and errors

Due to the temporal and spatial variance of the project, data has been obtained from multiple sources which are accompanied by different scales, resulting in a potential for inaccuracies (Petit and Lambin, 2002). There are errors within this type of data collection as they all have to be reduced to the same scale in order for an effective and significant comparison. Haase et al (2007) warns that processing historical maps are full of variance in survey techniques and levels of detail, for example some of the Dudley Stamp (1930) land cover maps were completed by school children and so inconsistencies are common. Differences in skill, scale and intention from multiple sources of data make historical and contemporary data comparisons difficult (Tingley and Beissinger, 2009). This is exacerbated through the fact that maps are an abstraction from reality (Robinson, 1978) and successive manipulation through GIS can create a false sense of accuracy. Integration of differences involves a loss of information caused by thematic aggregation of analysis (Petit and Lambin, 2002), exacerbated by the fact that there is no coherent land use mapping method in place, and the data collected thus far has been produced under different overall project aims and contexts (Dallimer et al, 2009). Due to unavailability of the data, the land cover maps do not show a completely linear progression through time from 1930 - 2018 but do provide useful information nonetheless. Despite these issues, manipulation of historical data can provide a valuable insight and an alternative lens through which to view changes in the natural world (Tingley and Beissinger, 2009) even though a trade-off between going back in time as far as possible and accuracy within comparisons is experienced (Petit and Lambin, 2002).

# 4.0 Results and Data Analysis

## 4.1 Land Cover Change

Figure 4.1 represents the land cover change in TKCE, covering 1930, 1990, 2000, 2007 and 2015. Data was downloaded from EDINA Digimap and composed in QGIS. Although the land classification categories from the Center for Hydrology and Ecology (2018) are not identical over the data sets, temporal deductions of land cover change can still be made. The most notable change is a reduction in arable cultivation over the years and an increase in improved grassland, with Figure 4.1(b) revealing the highest level of pasture land compared to other years. The 1930s Dudley Stamp data set was unavailable for download and so was not configured in QGIS.

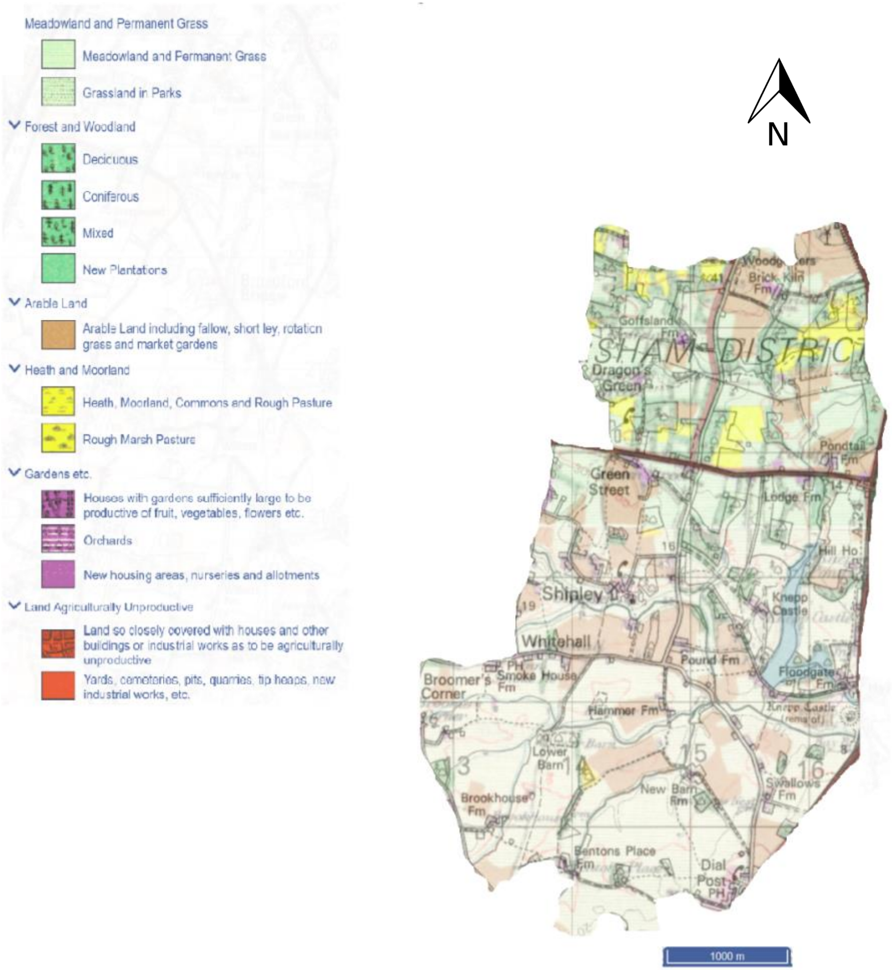
## 4.2 Aerial Photography 1947 - 2017

Aerial Photographs collected from the West Sussex County Records Office for the years 1947, 1971, 1981 and 1991 were georeferenced in a raster based system in ArcGIS (Figure 4.2). Transformations aimed to stay within a second - order polynomial in order to reduce distortions and warping (Cajthaml, 2013). A modern day satellite basemap was used to place control points for the aerial photographs and used to create a map for 2018, resulting in five additional satellite depictions through time of TKCE to visually aid the process of land cover change. As is always the case with compiling historical data, some photos were missing from the records and so a completely coherent coverage of the study area has not been obtained in this data set, however; enough has been reproduced to see a progression in the vegetation structure through time. Slenicka et al (2009) also used aerial photographs from the 1950s to decipher previous land cover change and categorize the land cover types.

## 4.3 Historical maps of The Knepp Castle Estate

Figures 4.3.1 - 4.3.4 represent the results of County Series, National Grid and Ordnance Survey maps composed in ArcGIS and reveal instances where hedgerows have increased or decreased in length over time. Figure 4.3.4 is a good indication of the vegetation succession witnessed on TKCE.

a)




b)

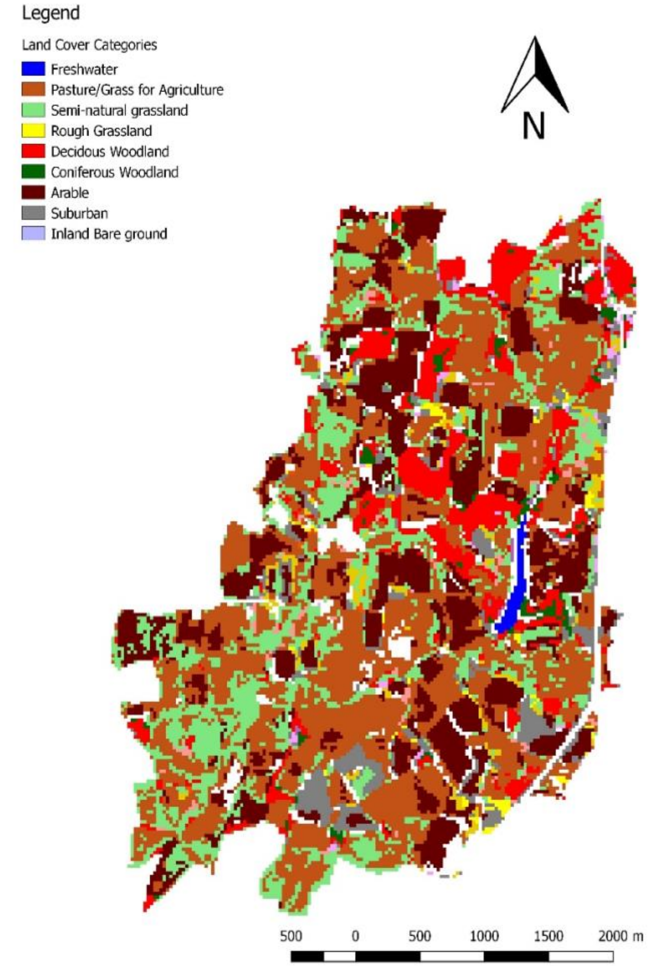
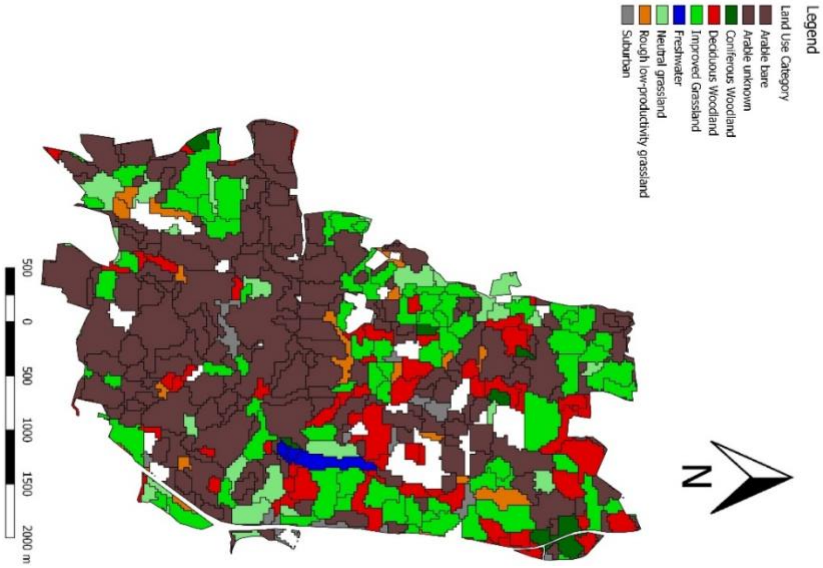
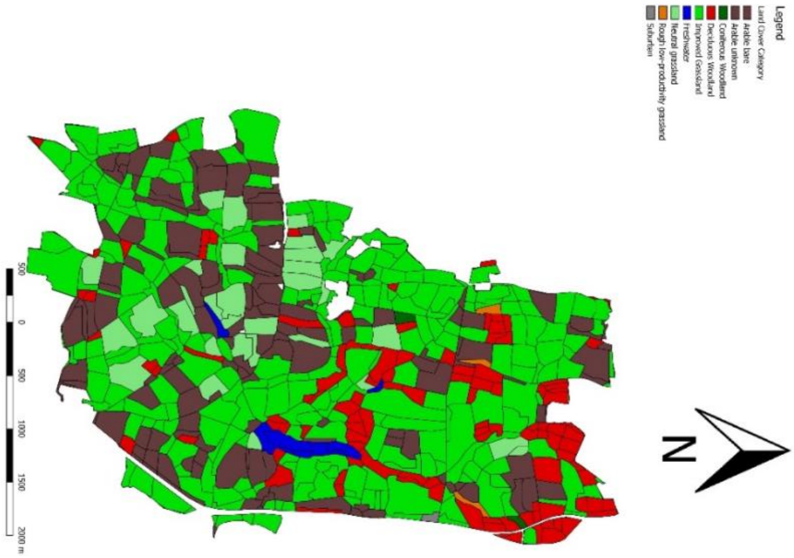



*Figure 4.1*: Land cover maps of TKCE, data from EDINA Digimap, (a) compromised of screenshots and (b) to (d) composed in QGIS, and (e) in ArcGIS.
(a): 1930. (b): 1990. (c): 2000. (d): 2007. (e): 2015.

c


Legend
Land Use Category
Arable bare
Arable unknown
Coniferous Woodland
Deciduous Woodland
Improved Grassland
Freshwater
Neutral grassland
Rough low-productivity grassland
Suburban
N
500
0
500
1000
1500
2000 m

d)


Legend
Land Cover Category
Arable bare
Arable unknown
Coniferous Woodland
Deciduous Woodland
Improved Grassland
Freshwater
Neutral grassland
Rough low-productivity grassland
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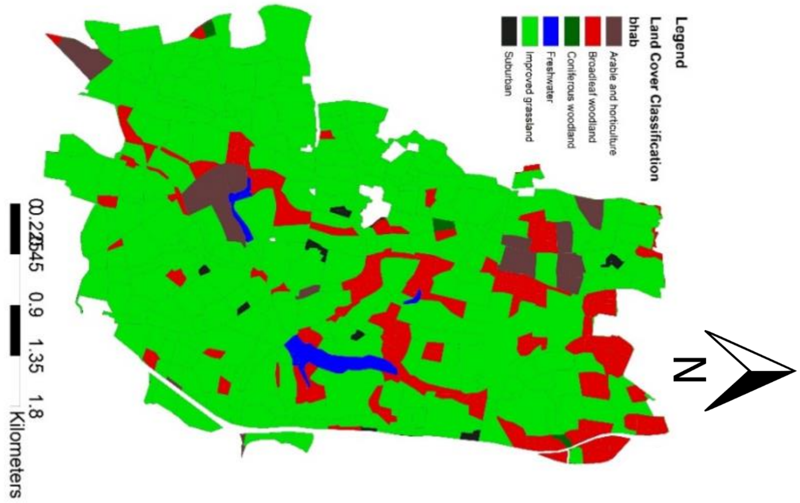

Legend
Land Cover Classification
bhab
Arable and horticulture
Broadleaf woodland
Coniferous woodland
Freshwater
Improved grassland
Suburban
N
0.9
1.35
1.8
Kilometers

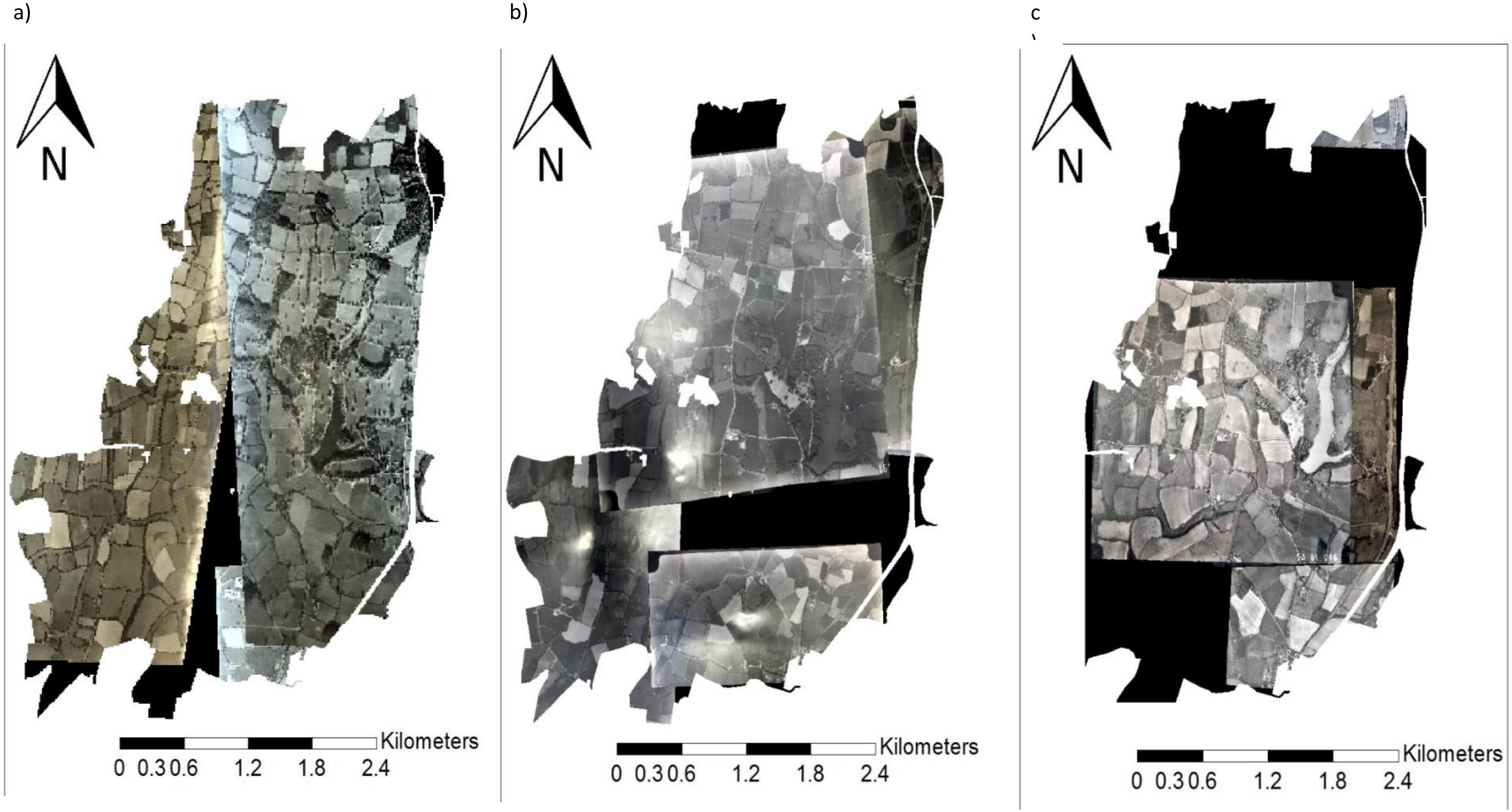



*Figure 4.2:* Aerial Photography of TKCE, georeferenced in ArcGIS. (a): 1947. (b): 1971. (c): 1981. (d):1991. (e):2017.

d)


e)

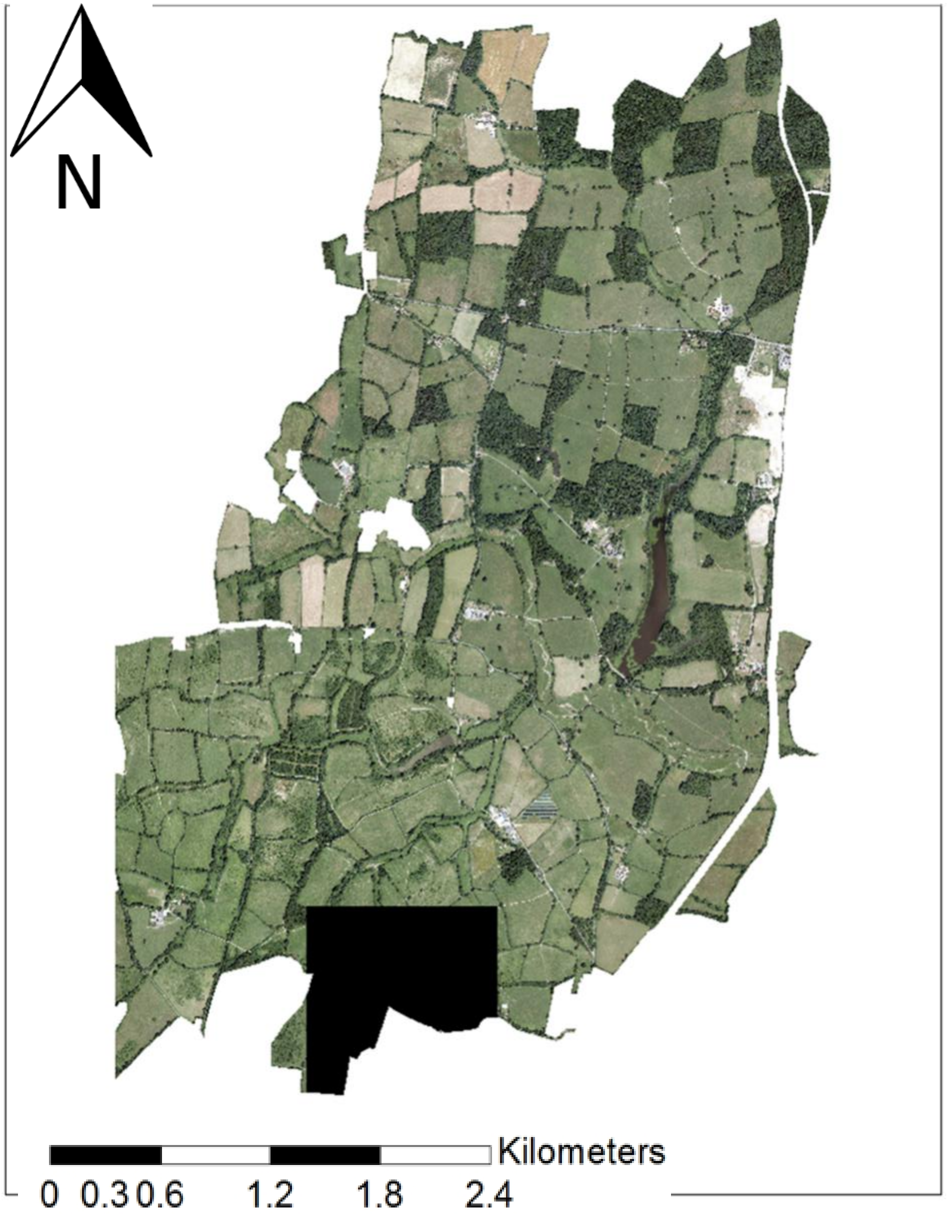



*Figure 4.3.1*: Maps of the Northern Block. © Crown Copyright Ordnance Survey (2019). (a):1890. (b):1960. (c):1980. (d) 2017. Red arrow in (d) indicates hedgerow length increase compared to (a).

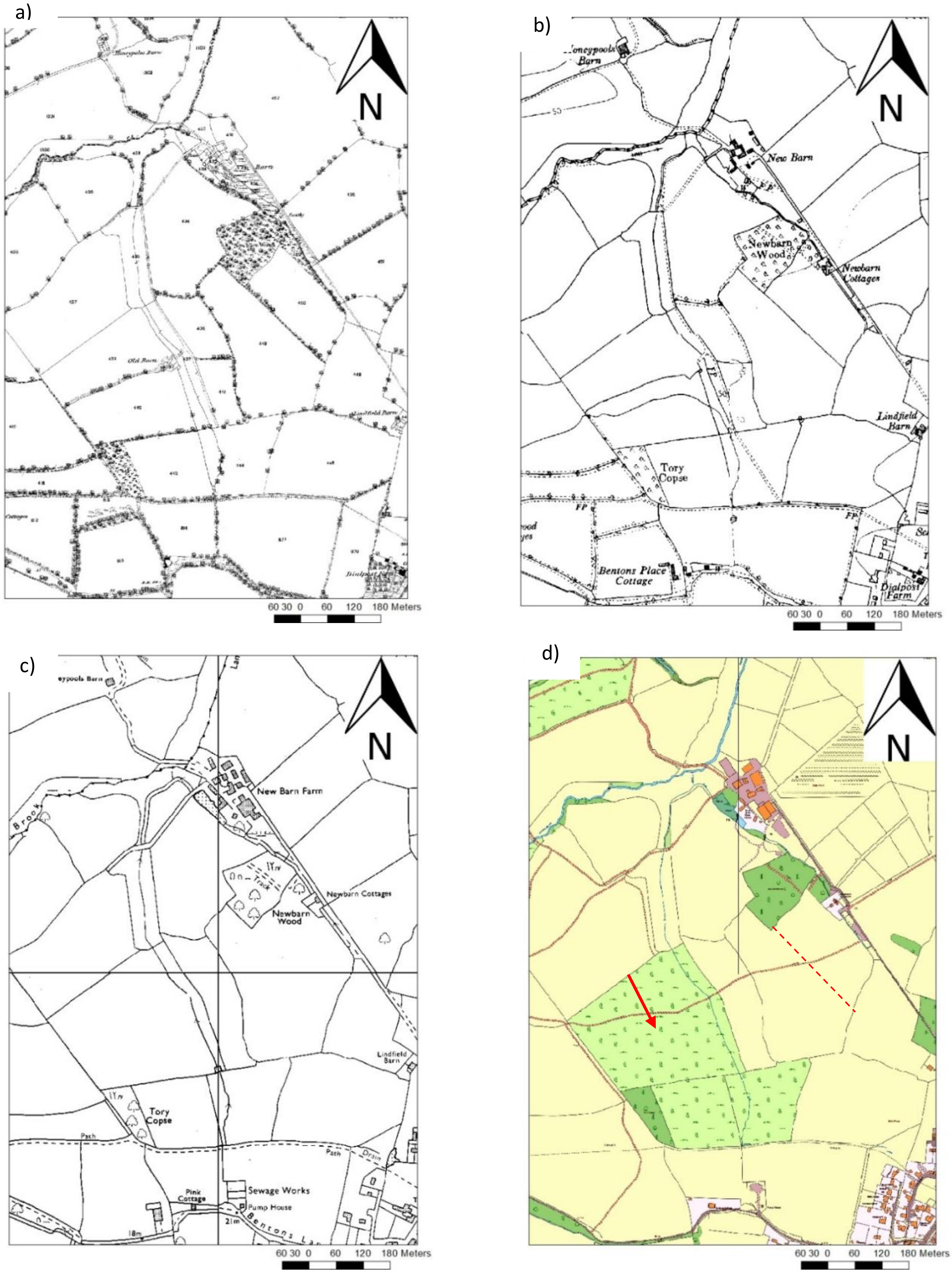
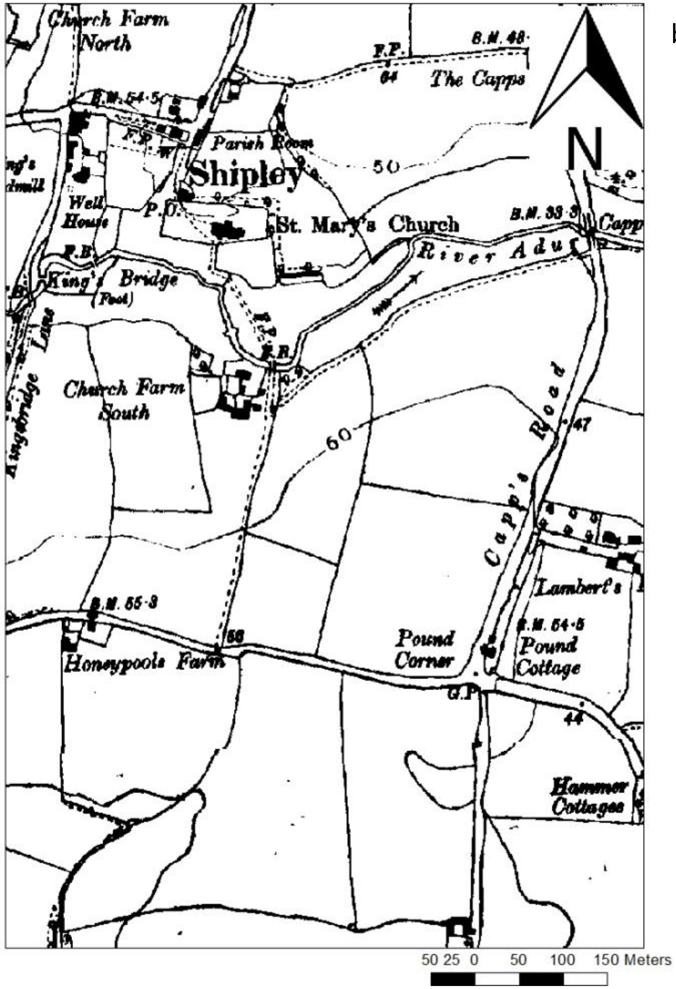



*Figure 4.3.2:* Maps of the Southern Block. © Crown Copyright Ordnance Survey (2019). (a):1890. (b):1960. (c):1980. (d) 2017. Red arrow in (d) indicates hedgerow lost to vegetation and dashed line represents hedgerow loss.

a)




b)

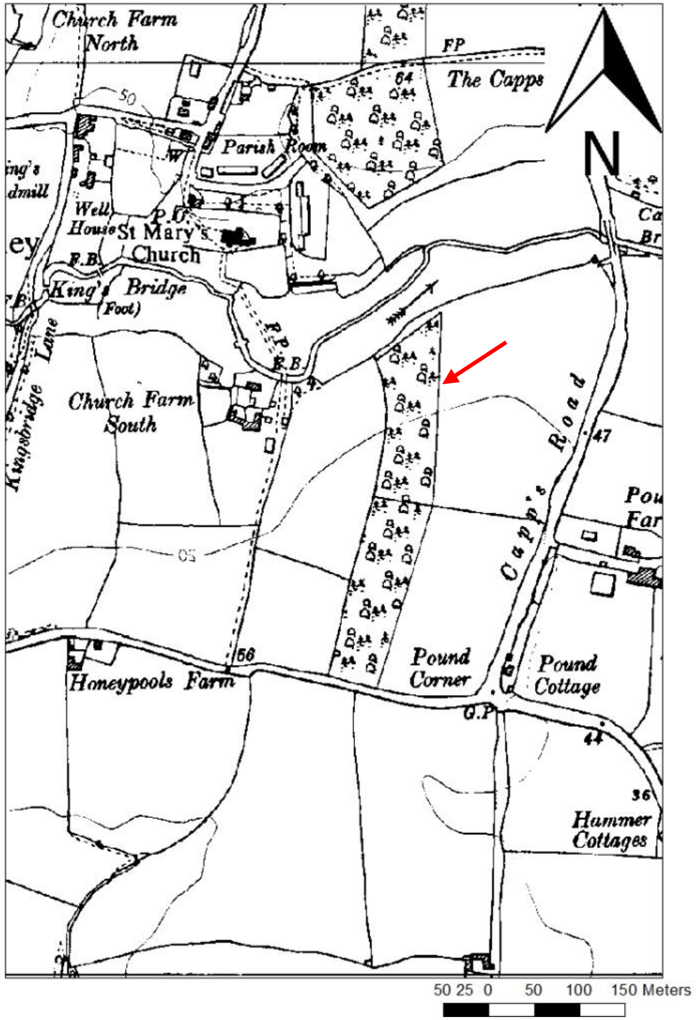



*Figure 4.3.3*: Historical maps from 1910 (a) and 1960 (b) showing the appearance of Hedge 6 from the Middle block (red arrow).

## 4.4. Hedge length change through time

Figures 4.4 and Table 4.4 reveal the hedge length change throughout the different time periods. 1910 - 1960 contains the largest interval in time as suitable maps for the interval were not available. All blocks saw an overall increase in hedge length from 1870 - 2018, with the largest overall gains occurring in the Northern Block with a sum gain of 770.23m compared to the lowest overall increases in the Southern block, where total gains over the same time period measured at just 38.98m (Table 4.4). The Northern block's most significant increases occurred between 1960 -1980, whilst both the Middle and Southern blocks experienced their greatest increases between 1910 - 1960. The largest overall losses in hedge length were witnessed in the Southern block between 1960-1980, with a loss of 12.17m (Table 4.4). The Middle block contains the hedge with the single largest increase at any one time, with a value of 421.86m between 1910 -1960 (Appendix 5), whilst the Northern block contains the opposite, with one hedge losing 75.26m between 1980 and 2018.

*Table 4.4:* Sum Hedge length change (m)

| | 1870-1890 | 1890-1910 | 1910-1960 | 1960-1980 | 1980-2018 | 1870-2018 |
|---|---|---|---|---|---|---|
| Northern | 44.07 | -7.75 | 12.86 | 605.01 | 116.03 | 770.23 |
| Middle | -0.4 | 11.21 | 435.32 | 26.56 | 18.11 | 490.8 |
| Southern | 5.729 | -11.49 | 34.09 | -12.17 | 22.82 | 38.98 |

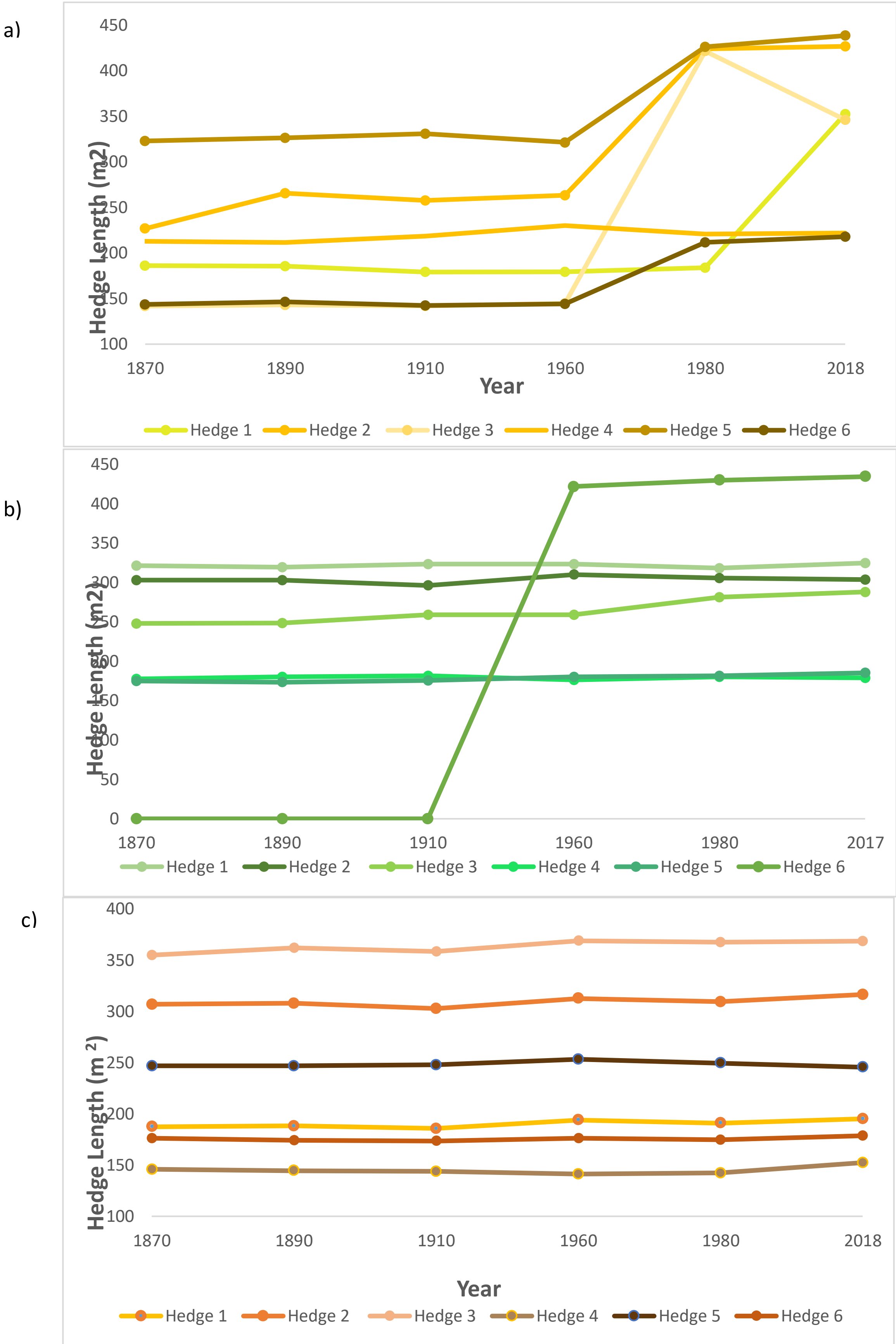



*Figure 4.4:* Hedge length values through time on TKCE: (a) Northern block (b) Middle block (c) Southern block

## 4.5: Percentage Change of Species Richness (m2) of hedgerow through time

Figure 4.5 displays the results of the CSR over time. No obvious trends across the data sets are clear, but notable changes are apparent within each block. 1870-1890 observed majority increases in CSR across the blocks (Figure 4.5), and 1910 - 1960 increases were marginal in all but the Southern block which contains relatively consistent growth. The Northern block witnessed significant CSR increases between 1960 - 1980 amongst the majority of its hedges compared to the Middle and Southern blocks which only saw anomalously large increases within one hedge only. Conversely, 1980 - 2018 saw a higher trend of CSR increases in the Southern block whilst the Middle and Northern blocks saw smaller increases and even a large decrease in the Northern block. The largest single species richness per $m^2$ of hedgerow for the Northern block lies in Hedge 5 in 1980 with a value of 1534.88, whilst the Middle blocks largest single value is 1719.68 in 1980, and the Southern block's is 1770.58 in 1960. (See Appendix 3). The largest sum value of species richness per m2 of hedgerow across the time periods was seen in 2018 (See Appendix 3).

a)
b)
c)


*Figure 4.5:* CSR through time. (a): Northern Block. (b). Middle Block. (c): Southern Block.

# 5.0 Discussion:

## 5.1 Land use change within The Knepp Castle Estate

For an Estate of its size, TKCE has witnessed a multitude of land use changes throughout its history that are dependent on external economic, societal and political policy pressures. Lambin and Meyfroidt (2010) elaborate that many land use transitions are a linear as well as a reversible dynamic, driven by maximum outputs from ecosystem services. Exogenous drivers of land use transitions are external from the ecological system, tied with policy and economic drivers from the state (Lambin and Meyfroidt, 2010), and early land use changes on TKCE highlight the significance of the relationship between the state and the farmer (Martin, 2000).

The Dudley Stamp land cover map of TKCE (Figure 4.1a) reveals how the majority of TCKE was comprised of arable land surrounded by permanent grassland and some deciduous woodland, or, as seen in the Middle Block; Grassland in parks. Martin (2000) supports these findings, observing that in the 1930s around 20% of national land was arable, predominantly bordered by some form of grassland. It is interesting to note how Figure 4.1a represents a time of agricultural depression (Swetnam, 2006), and the obvious changes in land cover in agriculturally productive times (Figure 4.1b-c) confirms this, with higher coverages of arable and pastureland.

Exogenous societal and policy pressures such as the advent of World War Two saw one of the biggest agricultural policy overhauls in the UK's history, with farmers obtaining £4.93 per hectare to plough up permanent pasture, a subsidy that rewards twice the price of the labour required (Martin, 2000). Transformations observed as a result of similar policies can be seen in a comparison between the aerial photography presented in Figure 4.2a and Figure 4.1a. Sir Merrik Burrell resided in TKCE at the time and was Chairman of the West Sussex War Agricultural Executive Committee. He led by example, pushing others in the Parish to plough up as much of their land as possible. In addition, land girls drafted in to farm the land during the War doubled West Sussex's agricultural area to 8.1 million hectares in just five years (Tree, 2018). After WW2, many large estates were broken amongst a culture of production maximisation on a national scale (Dallimer et al 2009). Much of the current agricultural landscape within the UK today are due to the 1947 Agricultural Act that promoted self-sufficiency in food production post war by fixing production prices no matter the output level (Martin, 2000). The Act included financial subsidies such as price reductions on large farm machinery to aid production (Robinson and Sutherland, 2002). Lowland farms in particular followed trends of expansion, specialisation, hedgerow removal and an increase in pesticides and fertilizer (Boatman et al, 2007) to aid productivity, explaining the increase in arable land witnessed on TKCE from the middle of the 20th century (Figure 4.2). As rationing continued nine years after Victory for Europe day, increases in production did not cease when the war ended, with The Agricultural Act of 1947 securing prices and helping TKCE to maintain pasture in the Northern Block in order to resume cattle rearing.

The era of agricultural escalation peaked during the 1960s and 70s, accompanied by increasing populations and the rise of new technologies that maximised agricultural production to unprecedented levels. The use of pesticides are an example of this; between 1960 -1980, 400 new forms of pesticide were in regular circulation saving £25 million a year in losses (Martin, 2000). Additionally observable in Figures 4.2a-b is the trend away from nitrogen fixing grassland cultivation to more soil intensive cereals which required government subsidised fertiliser (Tree, 2018). The Green Revolution was not just restricted to the UK; it is thought that without these innovative measures, China would need three times as much agricultural land area than is currently under cultivation (Khush, 2001). The technological advances were needed to match the increases in the UK population (46 million in 1930 to over 66 million in 2017 (ONS 2018)) driving the increases in arable cultivation witnessed in Figure 4.2a-c. The UK's accession into the EEC (European Economic Community) in 1972 exacerbated overproduction due to the higher market value of products compared to previous domestic prices, catalysing higher levels of production nationwide (Martin, 2000).

Despite lying on some of the worst soils in the UK, the Low Weald clay has experienced a lot of chemical and labour input in order to become productive arable land, particularly seen in the Southern block (Figure 4.1b-e) (Greenaway, 2011). The CAP contributed to this and by the 1970s, the subsidies had caused a surplus of food production in the UK as excess could not easily be sold on the cheaper world markets. The effects of this policy can be seen in findings from Haines-Young et al (2003), who evaluated land use change across the different national Environmental Zones. TKCE sits within Environment Zone 1 which characterises the south and southern eastern parts of England. The study found that from 1980 - 1990, increases in arable land cover coupled with a decrease in grassland dominated Environmental Zone 1's land cover change. Changes such as these are addressed in Figure

4.2 where the aerial viewpoint enables a qualitative view of this assessment, which also coincides with the current Burrell family takeover of the working farm in 1987. As is common practice, young farmers often look to transform farming practices in favour of more efficient methods (Van Vliet, 2015) and TKCE is not exempt in this - the Burrell family tried to achieve an extra 906 hectares of cultivation from 1987 to the early 21st century through modernisation, although this was not quite achieved (Tree, 2018:34) observable in Figure 4.2(c-e).

Figure 4.1(c-e) represents land cover changes that are more influenced by the individual endogenous pressures rather than policy driven, exogenic matters (Lambin and Meyfroidt, 2010). The overall trend of decreasing arable cultivation and simultaneous increase in improved grassland from 2000 - 2015 reflects the uptake of the rewilding project on TKCE (Figure 4.1). Endogenous land transitions such as these are related to the socio-economic feedbacks within the ecological system, influenced more by the local and individual decision maker relating to the maintenance of ecosystem services (Lambin and Meyfroidt, 2010). The CAP began to shift away from production in 1992 with the Macsharry reforms (Patterson, 1997), essentially decoupling agricultural production and financial incentive. Set-aside land policies and environmental management incentives grew in replacement (Martin, 2000). At the beginning of the century TKCE diverged away from agriculture by restoring the historical deer park in the Middle block of the estate, and this included sewing a top layer of native wildflower seed mix which was cropped for three successive years to reduce nitrate and phosphate levels within the soil (Tree, 2018). Combined with the withdrawal of chemical fertilisers, management practices such as this met the criteria for the land cover to be considered Improved Grassland and therefore accounts for the loss of arable and increases in improved grassland seen during this time period (Figure 4.1c-d). Within a national context, the deceleration of agricultural output - and loss of arable land cover - followed the UK’s agri-environmental focuses of CAP around wildlife conservation (Kleijn and Sutherland, 2003).

Complimentary socio-economic drivers also attributed to the uptake of rewilding on TKCE: ‘due to an amateurish love for wildlife and because we would have lost an impossible amount of money if we had continued to farm’ (Tree, 2018:9). Martin (2000) reiterates this within the national context, with real farming incomes falling by 47% in 1997. Despite their best efforts to intensify over 15 years, the Burrell family only saw two years of cash surpluses (Tree, 2018), encouraging the turn to rewilding and subsequent increase in improved grassland (Figures 4.1 and 4.2). Hodder et al (2014) investigated the future projections for land cover within TKCE, and the trend of decreasing arable cover and an increase in Improved Grassland and woodland was found to continue to increase to 2060.

## 5.2 Influencers of hedgerow length change over time and subsequent impacts on Species Richness

Hedgerow length and CSR are inextricably tied through the calculation of CSR which reveal trends in biodiversity. It is clear from Figures 4.1-4.2 that land use has varied from low and high intensity agriculture to a complete turnaround to conservation promotion. Hedgerow lengths are affected by land use changes and the external contexts that condition them. The influence of land cover change on biodiversity is evident through CSR variations between the blocks (Figure 4.5). French and Cummins (2001) have also found links between the species richness of the herb layer of hedgerows and land use change in Britain, as well as emphasising the importance of floristic diversity as an indicator of the surrounding ecosystem health. All blocks witnessed hedge length increases from 1870 - 2018, but variances within the blocks are apparent based on differing land use changes (Figure 4.4). Drivers of land use change across the decades will induce different management techniques and influences on hedgerow length (Schmitz et al 2007). This is especially relevant to TKCE as the separate blocks originated from different land uses.

Declines between 1870 -1910 can be attributed to enclosures of land between the 16th and 19th century, generating the structural mosaic of hedgerow networks across the country (Leonard and Cobham, 1977, Marshall and Moonen, 2002), and although appearing counterintuitive, the planting of hedges at this time can result in an overall hedge length decrease (Figure 4.3.1) as converted open field systems were divided by legal enclosures, with the first Enclosure Act passed in 1606 (Neeson, 1993). A decline of 7.88m for one hedge in the Northern block (Appendix 5) can be attributed to this. As well as enclosure, the increases in the value of livestock experienced at this time compared to cereal production would favour field systems with enclosed spaces in order to hold livestock (Pretty, 1991), which echoes declines in hedge lengths seen in the pasture lands of the Northern block. However, changes in landscape structures are multifactorial and other motivations should be considered; the hedgerow loss experienced by the arable Southern block from 1870-1910 can be related to the increases in technology and the obsoleting of hedgerows that only interfered with food production (Pretty, 1991). These losses also coincide with the production of the combine harvester which would need larger field boundaries, and the uptake of a horse drawn plough which also requires more field space - and subsequent hedgerow removal - than more traditional methods (Robinson and Sutherland, 2002). Appendix 5 reveals a loss of 5.09m in the Southern Block that relates to these land use changes, also presented in Figure 4.4.

The CSR for 1870 - 1910 (Figure 4.5) reflects the changes in hedgerow length. Declines predominantly seen in the Southern and Northern Blocks correlate to hedge length decreases witnessed in this time, as the potential for hedgerows to act as refuges for woodland plants decreases with hedgerow loss and so biodiversity levels (CSR) also decline (Staley et al, 2013). Agricultural depressions and the abandonment of government policy also dominated from 1870 - 1914,  and in 1901 only 12% of the male population was employed in agriculture, reflecting the lull in agricultural activity (NFU, 2014). A fall in the drive of policy could also contribute to the relatively stable readings for both hedge length and CSR between 1870 - 1910 across the Northern and Middle blocks, and the decreases observed in the Southern Block (Figure 4.5c) tie in with the transition of farming practices during this period.

The changes in hedgerow length are relatively marginal in the 1910 - 1960 despite intensive national and European level agricultural policy application (this excludes the appearance of Hedge 6 in the Middle block, see Figure 4.3.4.) The agricultural endeavours accompanying the outbreak of two World Wars amongst labour, supply and equipment shortages led to astronomical efforts to increase arable productivity. 'Ploughing up' from 1917 created an extra 1 million hectares of land (NFU 2014). Ploughing up as a policy would favour areas of uninterrupted field for arable cultivation, inducing hedgerow loss. Over the war time period, hedges lengthen as fields amalgamate and divisible boundaries are removed, effectively reversing some of the effects of the enclosure acts of the previous centuries (Figure 4.3.2a-d). Examples of  slight hedge length increase during this time that could echo this policy can be found in all blocks (Table 1), and the anomalously large value of 421.86m for one hedge represents the appearance of a hedge within the 50 year gap, perhaps used to contain livestock in the Middle Block. Every block was turned over to ploughable space, including the private gardens of the Estate and the deer park landscape, prompting one of only two time periods analysed that saw positive overall increases in hedge lengths within every block (Table 4.4). The decreases observed, for example in the Northern Block, can also be attributed to the same wartime hedgerow removal policy. Modern machinery subsidised by the government through a £2.5 million investment in the Ford Motor Company led to the number of tractors trebling between 1942 - 1950 (Martin 2000) which would require wider entrance ways and subsequent hedge length decreases (Appendix 5, Figure 4.4).

The CSR between 1910 -1960 reflect the application of national policies discussed; Figure 4.5 reveals the largest change thus far in the Southern block, relating to the lengthening of hedgerows to favour arable production, increasing habitat for woodland species (Merckx et al 2012). Similarly, the anomalous result derived from Hedge 6 in the Middle block accounts for the sudden increase in habitat potential with the appearance of this hedgerow. The Northern and Middle blocks saw relatively little CSR during this time. In addition, the interval of 1910 - 1960 presents the most challenging explanation as it is the largest temporal gap between observations compared to other hedge length recordings, which increases the chance of more nuanced changes within the time period being missed, a common problem associated with historical investigation (Haase et al, 2007).

The nationwide loss of hedgerows peaked in the 1960s, after grants for their removal became available (Martin, 2000), and so TKCE is unique in that much of its historic hedgerow system has in fact between retained compared to nationwide levels (Burrell and Greenaway, 2011). This is be reflected in the hedgerow length values between 1960 - 1980 for the Middle and Southern blocks (Figure 4.2 (b-c)) which contain no significant changes in hedge length. The increases in hedge lengths observed in the Northern block during the same time period, however, could be indicative of hedgerow removal policy which would in turn create longer field boundaries (Figure 4.2a), with one hedge seeing a 277.05m increase (Figure 4.4a). Similar to the pro - production policy seen during war time Britain, EEC membership and the CAP produced similar effects in land use change, differing only in funding origin. This transition would have been particularly influenced by the security of higher arable prices within the European market, deterring a reliance on livestock production and thus altering hedge network structure in the Northern block (Figure 4.3.2b-c) (Martin, 2000).

Since hedge networks in the Middle and Southern blocks largely suited this production policy, their hedge lengths during this time saw less of a change apart from one large hedge length each. (Appendix 5).

As a consequence, the period between 1960 - 1980 is representative of the hedge lengths and CAP aims; CSR remained similar for the Southern and Middle Blocks as hedgerow structure was mainly retained from previous policy. The Northern Block, however observed the most dramatic overall increases in CSR thus far, with increases of 65.7% for one hedge (Figure 4.5a) again relating to the increased conversion of pasture land to arable cultivation during this time. Decreases in CSR in the Southern Block between 1960 - 1980 (Figure 4.5c) could be attributed to eutrophication from the input of fertilisers that were heavily introduced during this time period, reducing biodiversity by encouraging the growth of homogenous plant communities, a factor also encouraged by the monoculture favouring markets found in the EC (Staley et al 2013, Martin, 2000).

Across Europe, hedgerows have overwhelmingly disappeared over the last two decades – England witnessed a 21% decrease between 1984 – 1990 - with drivers predominantly associated with land abandonment or land use intensification (Sklenicka et al, 2009). Slenicka et al's (2009) study revealed that land use is a significant factor to hedgerow length change; for example, hedgerows are more likely to disappear when situated next to arable land over grassland. There are indications from the data collected that arable cultivation does influence hedgerow loss (Slenicka et al, 2009). The intensively farmed Southern block and parts of the Middle block under intensive cultivation showed the smallest overall increases in hedgerow length during this time compared to the low input, pasture based Northern block (Table 4.4, Figure 4.4). Robinson and Sutherland (2002) further support these findings, describing how arable areas may only have 20-30% of hedgerow extent compared to pastoral areas. This is due to arable areas not needing the stock – containing characteristic that hedgerows provide. Hedges in this area may also not be as long as they were previously cut every year after harvest, supported by (FIgure 4.4c) In addition, the introduction of roe and fallow deer into the Middle block as part of the rewilding project has thought to disturb hedgerow succession through foraging, represented through the lowest overall increases in hedge length out of all the blocks between 1980 - 2018 (Table 4.4) (Greenaway, 2011).

Despite some evidence supporting national agricultural policy of the time, the hedgerows on TKCE do not seem to completely align with the norm. The Low Weald Clay proved large scale farming to be difficult, and the mass removal of hedgerows would have proved counterproductive on the estate, as the hedgerow network also aligned with the historic drainage system of ditches that enabled farming to be at all possible (Tree, 2018). Therefore, despite national trends and practices saying otherwise, the retaining of, and in some cases, the increases in hedgerow length on TKCE from the 1980s onwards can be related to the retaining of this drainage mechanism. Increases in hedgerow length from 1980 – 2018 can also be related to reformation of production based CAP (Boatman, 2007). Agri-environmental schemes slowed down hedgerow loss with financial incentives for their protection (Robinson and Sutherland, 2002) as well as local council permission requests in order to remove hedgerows after 1997 (Martin, 2000). This is proved in the increases in hedges on TKCE (Figure 4.4, Table 4.4) and findings from Barr et al (1991) for the Department of the Environment which found 39,000 km of hedgerow gained during the 1990s compared to 124.800km lost during the 1980s. The niche, exogenic rewilding project on TKCE has additionally contributed to hedge length increases as hedgerows owe their existence to human intervention within the agricultural setting and without intervention, hedgerows often disappear into treelines (Forman and Baudry, 1984), creating a wood-pasture landscape envisaged by Vera (2009) (Figure 4.2(a-c), 4.3.4.).

CSR between 1980 - 2018 mirror the land use change experienced. Although the period does include a time of intensive farming on TKCE, this interval also importantly reflects the impacts of rewilding on the species richness in hedgerows, and the coupling of the largest species richness values (See Appendix 3) - impacted by keystone grazers increasing overall levels of diversity (Vera, 2009) - at the same time as the approach was adopted should not be regarded as a coincidence. Boutin et al (2007) confirm how the presence of natural or even semi-natural habitats increases the species pool - especially compared to those found within agricultural settings - and so biodiversity levels are benefitted, shown in Appendices 3 and 4. This point may be contradicted by the fact that this recent period in the Southern Block did not witness the largest CSR over the whole time period despite the most wild area of TKCE (Tree, 2018).; both the Middle and Northern blocks saw higher CSR (Figure 4.5). Arguments against the actual increase in biodiversity have been made which could reflect the lower levels of species richness change over time - introducing animals into previously heavily cultivated land does not mimic the ecological baseline that Vera (2000) and other conservationists are trying to mimic before heavy anthropogenic influence (Kirby, 2003). Therefore, species richness levels may never reach expectations, perhaps creating the CSR dynamic seen in Figure. Eriksson (2004) further contributes that the presence of the past habitats in surrounding fields also contribute to the plant diversity among sites, and so the differing land cover type witnessed in Figure 4.2 will influence the CSR (Figures 4.5). In this way, plant species diversity patterns in the present day landscape have been cultivated under ecosystems that no longer exist, and so CSR may not be as high in recent times despite the rewilding attempts across the blocks (Ernoult and Alard, 2011), driving the potential for an extinction debt (Eriksson, 2004). Schmitz et al (2007) additionally found that in Spain, for example, hedges that were mechanically flailed - as was the case in the Southern Block - experienced lower levels of species richness. In a similar way, cattle grazing of the herb layer around hedges may decrease the levels of species richness, accounting for lower increases in CSR in the Northern Block compared to the Middle which mainly contained low impact sheep or deer grazers who have less of an impact on herb layer diversity (Forman and Baudry, 1984, Schmitz et al, (2007)) (Figure 4.5). Despite the variability in CSR and the slowing down in percentage increases in recent years, it is undeniable that the increases in the last 40 years have been impressive and the species richness per $m^2$ of hedge was highest during the period that rewilding began (Appendix 3).

# 6.0 Conclusions

Anthropogenic land cover alterations have occurred since the post glacial period, yet the past 200 years evaluated in this study have seen the most dramatic ecosystem disturbances to date. Conservation biology requires analysis on a spatial and temporal time scale that is large enough to evaluate longer term trends and influencers of biodiversity fluctuations (Greenaway, 2011), and in this way human - environmental interactions can be elucidated by both geographers, ecologists and historians (Forman and Baudry, 1984).

Further studies on TKCE will continue to monitor and reflect on the change that rewilding has had on the landscape, reflective of the project's aims (Greenaway, 2006). In addition, improvements in the data collection section of this study would lead to a more comprehensive idea of past landscapes. This could be undertaken by collecting data from higher numbers of hedgerows, for example, or by finding further archival data on land cover types for the time periods between 1870 – 1890, and between 1930 and the 21st century. This was unfortunately limited by a restricted access to historical resources. Future measures such as these will help reduce inconsistencies encountered by aggregating multiple forms of data from different time periods.

Nonetheless, this study has met its aims by providing an insight into the effect of land use change on species richness levels of hedgerows through time. Land use transitions on TKCE have seen a varied history reflective of evolving economic, political and social contexts of the period, with major turning points including wartime policies, production maximising Agriculture Acts and the accession into the EEC (later the EU). Land cover change between 1870 - 2018 on TKCE has found to evolve from low-input farming through to intensive agriculture as a result of exogenous factors such as war time and European Union policy and economic markets. The 21st century, however, saw endogenous drivers of land change take the reins (Lambin and Meyfroidt, 2010), as forward thinking conservation initiatives unfold through rewilding TKCE instead of forcing agricultural output on this marginal land type.

The tailored endogenous factors that encouraged the uptake of rewilding on TKCE explain why land cover change and subsequent hedgerow lengths do not mimic the national trends, as the decisions made here are unique to the study area. The ethos of rewilding is reflected in the hedgerow lengths and species richness per m2 of hedgerow increasing from 1870 – 2018, and is a testament to the project's success. The benefits of the most recent land conversion are evident in the CSR for 1980 – 2018, revealing the largest increases tested which sit with the current literature. For example, Schmitz et al (2007) make apparent how diversity responds to the energy flow of the ecosystem, and the mosaic of open grass parkland that is unfolding currently on TKCE benefits biodiversity; unlike its previous bounded agrarian landscape. Landscape scale management schemes accompanied by endogenous land use change drivers will be an important consideration for agricultural policy in the post-Brexit period (Loth et al, 2018), especially as the results from this study suggest that species richness and hedge lengths benefit from policies such as AES and rewilding. Sustainable landscape development is impossible without the commitment of land users who always have, and continue to, alter the landscape (Pleninger et al, 2006). For many farming families failing on marginal land, rewilding and a return to less invasive forms of management may be the answer for improved social, economic and ecosystem service benefits (Merckx and Pereira, 2015).

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# Appendices:

- **Appendix 1:** Average Species Richness values for each recorded hedge.

| Northern block | Hedge | Average species richness |
|---|---|---|
| | 1 | 3.2 |
| | 2 | 3 |
| | 3 | 3.6 |
| | 4 | 3.6 |
| | 5 | 3.6 |
| | 6 | 4.2 |
| Middle block | | Average species richness |
| | 1 | 4.6 |
| | 2 | 3.2 |
| | 3 | 2 |
| | 4 | 2.6 |
| | 5 | 4 |
| | 6 | 4 |
| Southern block | | Average species richness |
| | 1 | 3 |
| | 2 | 3.2 |
| | 3 | 4.8 |
| | 4 | 4.4 |
| | 5 | 4.4 |
| | 6 | 4.8 |

- **Appendix 2**: Raw hedge length data for each hedge, obtained from ArcGIS.

| Northern | 1870 | 1890 | 1910 | 1960 | 1980 | 2017 |
|---|---|---|---|---|---|---|
| Hedge 1 | 321.72 | 319.61 | 323.1 | 323.4 | 318.35 | 324.76 |
| Hedge 2 | 302.74 | 302.92 | 296.23 | 310.04 | 305.49 | 303.84 |
| Hedge 3 | 248.07 | 248.27 | 258.74 | 259.27 | 281.37 | 288.1 |
| Hedge 4 | 177.55 | 180.28 | 181.62 | 176.34 | 180.54 | 179.23 |
| Hedge 5 | 174.75 | 173.35 | 175.95 | 180.05 | 181.85 | 185.27 |
| Hedge 6 | 0 | 0 | 0 | 421.86 | 429.92 | 434.43 |
| Middle | 1870 | 1890 | 1910 | 1960 | 1980 | 2018 |
| Hedge 1 | 187.57 | 188.11 | 185.79 | 193.89 | 190.92 | 195.155 |
| Hedge 2 | 306.85 | 307.85 | 302.759 | 312.56 | 309.37 | 316.45 |
| Hedge 3 | 354.91 | 362.09 | 358.66 | 368.87 | 367.46 | 368.77 |
| Hedge 4 | 145.72 | 144.63 | 143.71 | 141.25 | 142.33 | 152.38 |
| Hedge 5 | 246.7 | 246.71 | 247.77 | 253.24 | 249.49 | 245.51 |

| | | | | | | |
|---|---|---|---|---|---|---|
| Hedge 6 | 176.151 | 174.24 | 173.45 | 176.42 | 174.49 | 178.62 |
| Southern | 1870 | 1890 | 1910 | 1960 | 1980 | 2018 |
| Hedge 1 | 186.21 | 185.44 | 179.18 | 179.6 | 183.99 | 352.763 |
| Hedge 2 | 226.89 | 265.89 | 258.01 | 263.59 | 424.14 | 427.03 |
| Hedge 3 | 141.86 | 142.85 | 142.18 | 144.66 | 421.71 | 346.45 |
| Hedge 4 | 212.92 | 211.65 | 218.39 | 230.21 | 221.06 | 222.05 |
| Hedge 5 | 323.34 | 326.75 | 330.94 | 321.7 | 426.356 | 438.79 |
| Hedge 6 | 143.62 | 146.33 | 142.46 | 144.26 | 211.78 | 217.99 |

- **Appendix 3**: Species Richness per m2 oh hedgerow (Average species richness * hedge length).

| Northern Block SR/m2 | | | | | | |
|---|---|---|---|---|---|---|
| | 1870 - 1890 | 1890 - 1910 | 1910-1960 | 1960-1980 | 1980-2018 | 1870 - 2018 |
| Hedge 1 | 595.872 | 593.408 | 573.376 | 574.72 | 588.768 | 1128.842 |
| Hedge 2 | 680.67 | 797.67 | 774.03 | 790.77 | 1272.42 | 1281.09 |
| Hedge 3 | 510.696 | 514.26 | 511.848 | 520.776 | 1518.156 | 1247.22 |
| Hedge 4 | 766.512 | 761.94 | 786.204 | 828.756 | 795.816 | 799.38 |
| Hedge 5 | 1164.024 | 1176.3 | 1191.384 | 1158.12 | 1534.882 | 1579.644 |
| Hedge 6 | 603.204 | 614.586 | 598.332 | 605.892 | 889.476 | 915.558 |
| | | | | | | |
| Middle Block SR/m2 | | | | | | |
| | 1870 - 1890 | 1890 - 1910 | 1910-1960 | 1960-1980 | 1980-2018 | 1870 - 2018 |
| Hedge 1 | 1479.912 | 1470.206 | 1486.26 | 1487.64 | 1464.41 | 1493.896 |
| Hedge 2 | 968.768 | 969.344 | 947.936 | 992.128 | 977.568 | 972.288 |
| Hedge 3 | 496.14 | 496.54 | 517.48 | 518.54 | 562.74 | 576.2 |
| Hedge 4 | 426.12 | 468.728 | 472.212 | 458.484 | 469.404 | 465.998 |
| Hedge 5 | 699 | 693.4 | 703.8 | 720.2 | 727.4 | 741.08 |
| Hedge 6 | 0 | 0 | 0 | 1687.44 | 1719.68 | 1737.72 |
| | | | | | | |
| Southern Block SR/m2 | | | | | | |

| | 1870 - 1890 | 1890 - 1910 | 1910-1960 | 1960-1980 | 1980-2018 | 1870 - 2018 |
|---|---|---|---|---|---|---|
| Hedge 1 | 562.71 | 564.33 | 557.37 | 581.67 | 572.76 | 585.465 |
| Hedge 2 | 981.92 | 985.12 | 968.8288 | 1000.192 | 989.984 | 1012.64 |
| Hedge 3 | 1703.568 | 1738.032 | 1721.568 | 1770.576 | 1763.808 | 1770.096 |
| Hedge 4 | 641.168 | 636.372 | 632.324 | 621.5 | 626.252 | 670.472 |
| Hedge 5 | 1085.48 | 1085.524 | 1090.188 | 1114.256 | 1097.756 | 1080.244 |
| Hedge 6 | 689.376 | 702.384 | 683.808 | 692.448 | 1016.544 | 1046.352 |

- **Appendix 4**: Percentage Change in Species Richness through time

| | 1870 - 1890 | 1890 - 1910 | 1910-1960 | 1960-1980 | 1980-2018 | 1870 - 2018 |
|---|---|---|---|---|---|---|
| Hedge 1 | -0.42 | -3.49 | 0.23 | 2.39 | 47.84 | 47.21 |
| Hedge 2 | 14.67 | -3.05 | 2.12 | 37.85 | 0.68 | 46.87 |
| Hedge 3 | 0.69 | -0.47 | 1.71 | 65.70 | -21.72 | 59.05 |
| Hedge 4 | -0.60 | 3.09 | 5.13 | -4.14 | 0.45 | 4.11 |
| Hedge 5 | 1.04 | 1.27 | -2.87 | 24.55 | 2.83 | 26.31 |
| Hedge 6 | 1.85 | -2.72 | 1.25 | 31.88 | 2.85 | 34.12 |

| | 1870 - 1890 | 1890 - 1910 | 1910-1960 | 1960-1980 | 1980-2018 | 1870-2018 |
|---|---|---|---|---|---|---|
| Hedge 1 | -0.66 | 1.08 | 0.09 | -1.59 | 1.97 | 133.45 |
| Hedge 2 | 0.06 | -2.26 | 4.45 | -1.49 | -0.54 | 110.94 |
| Hedge 3 | 0.08 | 4.05 | 0.20 | 7.85 | 2.34 | 96.55 |
| Hedge 4 | 9.09 | 0.74 | -2.99 | 2.33 | -0.73 | 1343.68 |
| Hedge 5 | -0.81 | 1.48 | 2.28 | 0.99 | 1.85 | 143.75 |
| Hedge 6 | 0.00 | 0.00 | 100.00 | 1.87 | 1.04 | 100.00 |

| | 1870 - 1890 | 1890 - 1910 | 1910-1960 | 1960-1980 | 1980-2018 | 1870 - 2018 |
|---|---|---|---|---|---|---|
| Hedge 1 | 0.29 | -1.25 | 4.18 | -1.56 | 2.17 | 86.77 |
| Hedge 2 | 0.32 | -1.68 | 3.14 | -1.03 | 2.24 | 85.48 |
| Hedge 3 | 1.98 | -0.96 | 2.77 | -0.38 | 0.36 | -458.20 |
| Hedge 4 | -0.75 | -0.64 | -1.74 | 0.76 | 6.60 | 111.43 |
| Hedge 5 | 0.00 | 0.43 | 2.16 | -1.50 | -1.62 | 100.25 |
| Hedge 6 | 1.85 | -2.72 | 1.25 | 31.88 | 2.85 | 34.99 |

- **Appendix 5*:*** Changes in (m) of hedge between the years. (a) Northern block (b) Middle block (c) Southern block

| Hedge Number | 1870-1890 | 1890-1910 | 1910-1960 | 1960-1980 | 1980-2018 | 1870-2018 |
|---|---|---|---|---|---|---|
| Hedge 1 | -0.77 | -6.26 | 0.42 | 4.39 | 168.77 | 166.55 |
| Hedge 2 | 39 | -7.88 | 5.58 | 160.55 | 2.89 | 200.14 |
| Hedge 3 | 0.99 | -0.67 | 2.48 | 277.05 | -75.26 | 204.59 |
| Hedge 4 | -1.27 | 6.74 | 11.82 | -9.15 | 0.99 | 9.13 |
| Hedge 5 | 3.41 | 4.19 | -9.24 | 104.65 | 12.434 | 115.45 |
| Hedge 6 | 2.71 | -3.87 | 1.8 | 67.52 | 6.21 | 74.37 |
| | | | | | | |
| Hedge 1 | -2.11 | 3.49 | 0.3 | -5.05 | 6.41 | 3.04 |
| Hedge 2 | 0.18 | -6.69 | 13.81 | -4.55 | -1.65 | 1.1 |
| Hedge 3 | 0.2 | 10.47 | 0.53 | 22.1 | 6.73 | 40.03 |
| Hedge 4 | 2.73 | 1.34 | -5.28 | 4.2 | -1.31 | 1.68 |
| Hedge 5 | -1.4 | 2.6 | 4.1 | 1.8 | 3.42 | 10.52 |
| Hedge 6 | 0 | 0 | 421.86 | 8.06 | 4.51 | 434.43 |
| | | | | | | |
| Hedge 1 | 0.54 | -2.32 | 8.1 | -2.97 | 4.235 | 7.58 |
| Hedge 2 | 1 | -5.09 | 9.80 | -3.19 | 7.08 | 9.6 |
| Hedge 3 | 7.18 | -3.43 | 10.21 | -1.41 | 1.31 | 13.86 |
| Hedge 4 | -1.09 | -0.92 | -2.46 | 1.08 | 10.05 | 6.66 |
| Hedge 5 | 0.01 | 1.06 | 5.47 | -3.75 | -3.98 | -1.19 |
| Hedge 6 | -1.91 | -0.79 | 2.97 | -1.93 | 4.13 | 2.469 |

# Skills Summary

| SKILLS | **How used/enhanced during dissertation** |
|---|---|
| **1. Subject-specific Knowledge** | The wider reading used during the project has given a greater insight into the global importance of land use change and the state of biodiversity. Anthropogenic influences have been a particular focus of this and many other studies. |
| **2. Subject-specific Skills** | Fieldwork planning and recording skills such as planning the equipment needed and necessary risk assessment knowledge has been solidified. Furthermore, proficiency in both Arc and QGIS has been greatly improved through creating maps through geo-referencing photographs and using external vector data. |
| **3. General Skills and Attributes** | |
| a) **Self-management** | Discipline to stick to the Gantt chart and be aware of deadlines and personal aptitude to achieve them have been realised successfully. |
| b) **Positive attitude** | A genuine interest in the project and enthusiasm to meet deadline has aided my attitude during this study. |
| c) **Problem solving** | Problem solving skills were particularly practised when using ArcGIS through trial and error and research, as well as help from staff members. |
| d) **Communication and literacy** | Communication with both my supervisor and staff members on TKCE has been vital throughout the project for clarity on what direction to take. |
| e) **Application of numeracy** | Numerical skills have been enhanced through calculation of both hedgerow change over time and species richness per m2 over time. |
| f) **Application of information technology** | IT has been essential to this project in terms of using software such as Digimap, ArcGIS and QGIS, Excel and Word processors. |
| g) **Business and customer awareness** | The project was carried out on a privately owned estate that houses business ventures such as Glamping and wildlife safaris, and so permission was granted and respect paid to the business of TKCE. |