Impacts of semi-natural grazing on vegetation biodiversity: A study of Knepp Castle Estate's re-wilding project

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Declaration of originality

'I hereby declare that this dissertation has been composed by me and is based on my own work'

Signature:________________________

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Abstract

Re-wilding is a revolutionary approach to conservation, one that aims to restore natural processes and keystone species to play a more prominent role in the landscape. In addition, re-wilding allows society to reconnect with nature. Knepp Castles Estate's re-wilding project is a unique experiment which endeavours to re-introduce large grazing herbivores into the landscape in the hope that this will eventually enable natural processes to take place and improve floral and faunal biodiversity. This study aims to assess the impacts of the semi-natural grazing at Knepp, as well as the ways in which it has affected the vegetation biodiversity. This was achieved by carrying out vegetation surveys and applying the data to calculate vegetation biodiversity. The results from each of Knepp's three experimental blocks highlight that different levels of human intervention were involved in the construction of these experiments. The different levels of intervention affected the species of grazer, intensity of grazing and land restoration strategies, which in turn had consequences for the vegetation biodiversity. Although there are numerous factors which can alter the dynamics of an ecosystem, this study found that high grazing intensity had the most influence on retarding vegetation biodiversity. As well as this, a fallow period (before the introduction of grazers) proved favourable to the formation or mosaic habitats, and subsequently, vegetation biodiversity. These findings provide suggestions for the future of Knepp's re-wilding experiments, namely that further and more controlled experimentation would be beneficial towards Knepp's success and the instruction of other similar re-wilding schemes.

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1. Introduction

Re-wilding is the mass restoration of ecosystems, which attempts to recover areas of wilderness. The term re-wilding was first used by the conservation activist Dave Foreman in the 1990's, but has since been refined by conservation biologist Michael Soulé (Fraser, 2009). He based re-wilding on three principles, firstly that ecosystems are maintained by 'top-down' predators, second that these animals require extensive space, and third that they need connectivity (Soulé and Noss, 1998). In short hand these are known as the three C's: cores, corridors and carnivores (Soulé and Noss, 1998). Since then, re-wilding has become popularised by writers such as British environmentalist George Monbiot, who argues that re-wilding is quintessentially based on the earth being a self-regulating and complex set of ecosystems (Monbiot, 2013). He explains the importance of the discovery of trophic cascading, and highlights the key role large animals play in many aspects of an ecosystem. In addition, he contends that removing these animals causes drastic changes to a habitat, both directly and indirectly (Monbiot, 2013). This makes for a powerful case to re-introduce large mammals and keystone species back into the environment, not in order to create a particular type of habitat, but instead, to minimise human intervention in natural processes.

Re-wilding has often been mistakenly described as an attempt to turn back time, when in fact, it is a way of using the past to inform the present (Lorimer et al., 2015). Knepp Castle Estate is an example of such a project, it is a site which uses large grazing herbivores to restore natural processes (Knepp, 2016). The ecologist Frans Vera's pioneering work in re-wilding has informed the project at Knepp (Greenway, 2006). His wood-pasture hypothesis determined the primordial baseline for the state of vegetation in Europe (Vera, 200).

This study assesses how the introduction of large grazing herbivores has influenced the vegetation biodiversity at Knepp Castle estates' re-wilding project. This is achieved by surveying the vegetation across three blocks of Knepp which have all undergone forms of re-wilding experiments. This study aims to fill in gaps of knowledge in Knepp's re-wilding project, no study has been carried out assessing the impacts of grazing on vegetation biodiversity. The data gathered during the survey period was then analysed, and upon examination, the southern block proved to be the most successful at 're-wilding'. This is because the southern block has progressed further in terms of vegetation succession due to an optimal grazing intensity of a variety of large grazing herbivores. The grazing herbivores have created intermediate patterns of vegetation disturbances, generating a mosaic habitat, and in turn increasing vegetation biodiversity. This is ultimately indicative of a habitat undergoing the process of re-wilding.

1.1 Aim and Objectives

The overall aim of this study is:

To determine the effect of Semi-natural Grazing on the vegetation biodiversity of Knepp Castle Estate's Re-wilding Project.

In order to achieve the above aim, this study has three objectives:

- Examine current vegetation biodiversity across the three blocks of Knepp Castle Estate.
- Assess the impact of grazing herbivores on vegetation biodiversity by comparing the three blocks of Knepp Castle Estate.
- Evaluate which block is most successful at achieving Knepp's re-wilding aims.

1.2 Background

1.2.1 Re-wilding

Broadly defined, re-wilding is a form of conservation aimed at protecting and restoring ecological processes and areas of wilderness by reintroducing keystone species and apex predators. The preservation of wilderness or 'wild places' was one of the main motivations of conservation pioneers like John Muir in the 19th century. However, re-wilding is a more modern take on such ideas. Re-wilding encompasses a broad range of projects globally with numerous approaches, the reintroduction of wolves to Yellowstone park is often seen as the flagship example. Other forms of re-wilding include plestoicene re-wilding and the use of megafauna analogues or island biogeography with the reintroduction of giant tortoises in the Golapogas (Hanson et al., 2010). In Europe re-wilding tends to focus on naturlistic grazing through the introduction of large herbivores. This has become popular due to concerns over rural depopulation and intensive farming (Macdonald et al., 2000). Arguably, some contention does exist when it comes to defining re-wilding, for example, Svenning (2015) argues that no part of Europe could be deemed 'natural' and untouched by humans, whereas Scottish Natural Heritage has defined and protected 19% of current Scotland under the label 'wild land areas' (Scottish Natural Heritage, 2016). An obstacle for re-wilding is the nescience that misconstrues re-wilding as something far-fetched, but the restoration of wild places is not a reverie of going back to the Ice Age (Macdonell, 2015). Nevertheless, although re-wilding has been applied to a diverse range of concepts and practices, they all share a common aim, to maintain, or improve, biodiversity, while reducing the impact of human interventions through the restoration of ecological processes (Lorimer et al., 2015).

Although a vast array of literature assesses the contention of what is considered 'wild' and 'natural' (Cronon 1995), this study does not discuss the theories of the social construction of nature. This study will refer to the term natural based on the mid-holocene landscape (around 6000 BP). This definition is chosen in parallel with Frans Vera, who defines a natural landscape as pre-agricultural, since it involved little human intervention.

1.2.3 The role of grazers

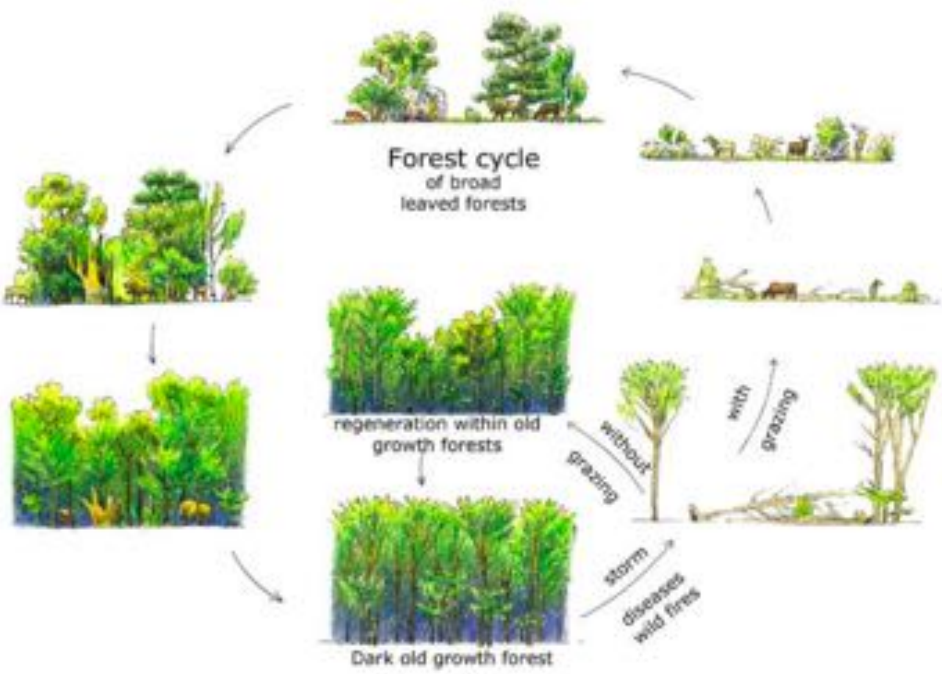


Figure 1. The forest cycle of vegetation succession occurring with grazing using Vera's theory (2000) creating a mosaic pattern. Figure also shows what happens without grazing (Vermeulen, 2015)

Frans Vera's study on the effects of grazing on forest history provoked a discussion in ecology over the structure of pre-human landscapes, one that could be used as a criterion for re-wilding. Vera (2000) argues that the primeval woodland landscapes of North West Europe were an open canopy, mosaic habitat. His wood pasture hypothesis argues that the landscape and vegetation structure was driven by large herbivores and a cyclical grazing dynamic

Figure 2. Photos showing two species of grazer at Knepp. *Bos primigenius taurus L* (Long-horn cattle) (top) in the Middle block, site 4 and Equus ferus caballus L (Exmoor Ponies) (bottom), in the Middle block, site 6. Knepp, July 2015.

(Figure 1). Vera (2000) uses paleoecology evidence to form the foundation of his argument; he proposes that the abundance of *Quercus sp* (oak), and *Corylus avellana,* (hazel) in the palynological data suggests an open canopy woodland. He explains that if the landscape had been a closed canopy woodland, oak and hazel would not be so well represented in the pollen data since they are light dependent species. Instead, he suggests that they thrived as a result of grazing animals (such as Figure 2) reducing the competition of more shade tolerant species (Hodder et al, 2009).

Before Vera's hypothesis was proposed, the closed canopy hypothesis was widely accepted by ecologists and conservationists (Birks, 2005). Paleontologists that champion the closed canopy hypothesis, such as Mitchell (2005), argue that the pollen diagrams are too static, therefore do not provide evidence of a cyclical succession. This suggests that the forest structure influenced herbivore abundance, rather than the other way around. Mitchell (2005) used primeval pollen records from Ireland to oppose Vera's theory. He compared *Quercus* and *C. avellana* pollen counts from Ireland to those in Europe (Mitchell used Ireland as a comparison due to the country's lack of large herbivores during the early Holocene). Mitchell's results show little difference between Europe and Ireland's pollen data; implying that the presence of large herbivores has little effect on the abundance of *Quercus* and *C. avellana.*

In rebuttal to Mitchell's argument, Birks (2005) points out that Mitchell's comparison data comes from lakes and bogs in a very large pollen source area. This could potentially hide mosaic vegetation patterns and the small changes that Vera describes in his theory (Sugita et al, 1997). Other literature suggests the landscape could be the result of a combination of factors, including fire (Hodder et al 2009, Birks 2005). For example, Buckland (2005) assessed primeval fossil insect data and compared it to the relevant pollen data. He reviewed the frequency of pyrophilic insect species associated with dead wood. The evidence shows that pyrophilic species were present during the early Holocene which would indicate there were

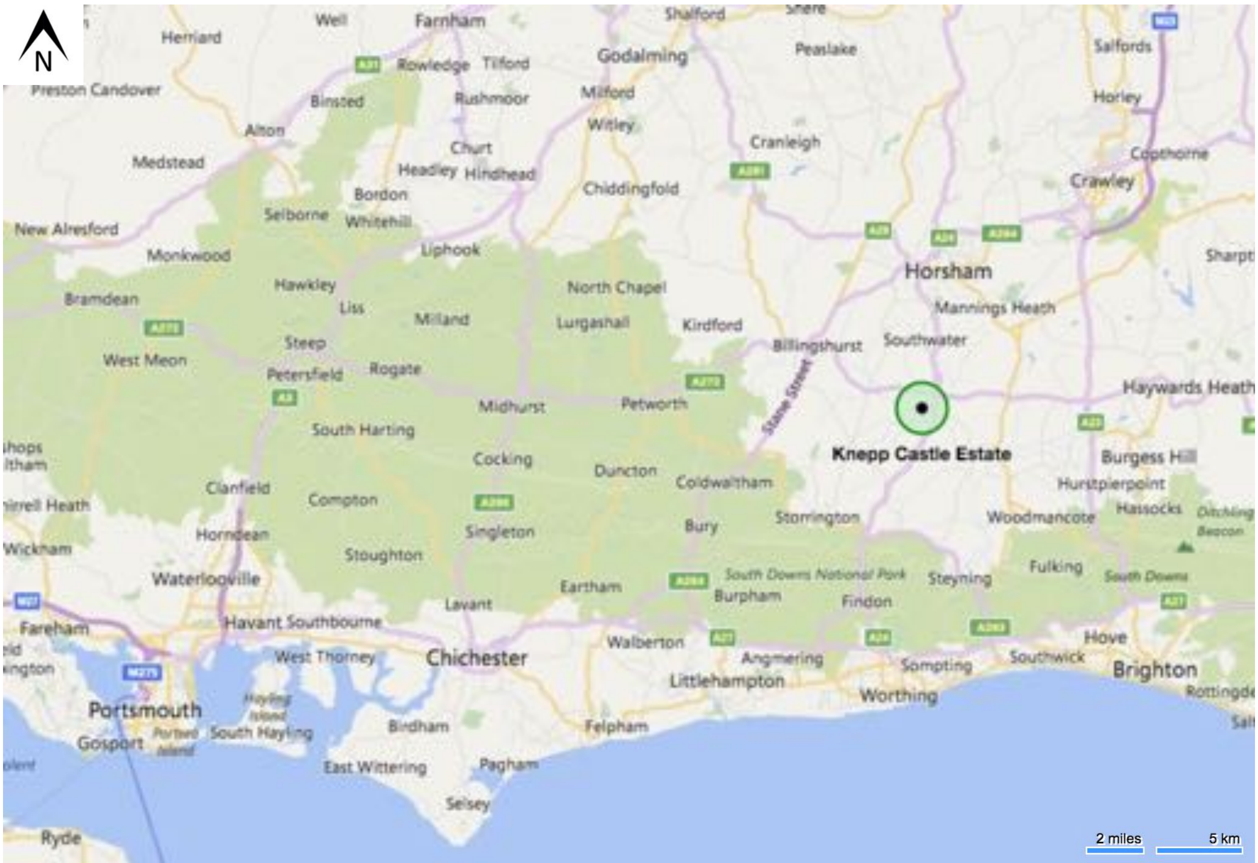


Figure 3. Location of Knepp Castle Estate in West Sussex, England. Source: www.bing.com/maps

open areas of woodland caused by fire, but not across the majority of the landscape (Hodder et al, 2009).

The debate between the two hypotheses is important because it influences present conservation methods in European re-wilding projects, and highlights the use of historical data as a potential to unlocking a baseline for recreating pre-human landscape. However, Hodder et al (2009) warns against relying on prehistoric baselines as a conservation tool because it is impossible to perfectly recreate a landscape. Therefore, conservationists must also consider the present context, such as, climate, soils and the introduction of new species, which have all played a part in the evolution of the landscape (Hodder et al, 2009). Whilst admittedly, some refute Vera's theory, there are conservation schemes, such as Knepp, applying his model as a tool to restore nature.

1.2.4 Knepp Castle Estate

The Knepp Castle Estates (Knepp) lies to the south of Horsham in West Sussex, England (Figure 3). Knepp lies alongside the River Adur and consists of a heavy clay soil. Charlie Burrell is the present owner of the estate and initiated the re-wilding project after coming into contact with Frans Vera's theory (Greenaway, 2006). Charlie Burrell aims to restore the landscape using a variety of large herbivores which will roam as freely as possible (Figure 2). This is known as semi-natural/near natural grazing, it cannot be defined as natural grazing because certain control measures are still in place which require some human intervention, such as culling (Greenaway, 2006).

Despite this, semi-natural grazing aims to de-domesticate the animals and replicate their natural use of the land with as little human intervention as possible (Vermeulen, 2015).

Figure 4. Photo comparing the same field at Knepp. The bottom in 2004 before the re-wilding project, sown with wheat. The top in 2009 after a five year fallow period and before grazers were introduced (Greenaway, 2011).

The project is supported by the Environment Agency and Natural England and intends to record and evaluate changes following the reversion of land from its previous use, intensive arable production (as seen in Figure 4) (Knepp, 2016). The hope is that the semi-natural grazing regime will improve floral and faunal diversity and the landscape will eventually manage itself (Knepp, 2016).

Little empirical work exists in the field of re-wilding as a whole (Svenning, 2015). Re-wilding with semi-natural grazing is a fairly unique conservation project, which means relatively few studies on the impacts exist (Greenaway, 2006). Additionally, no study has been carried out comparing the affect of multiple grazers on vegetation biodiversity across Knepp. This study helps to fill in the gaps of knowledge as well as add to the consistent monitoring of fauna taking place at Knepp. It will provide a better understanding of the ecological effects caused by the grazing system and help identify its successes in order to contribute to the project as a whole.

1.2.5 Defining and measuring biodiversity

Studying biological diversity (biodiversity) is important due to its significance to human life on earth and because floral and faunal species are being lost at an accelerating rate. Biodiversity has multiple benefits for all life on earth, including: healthy ecosystem functioning, the provision and financial value of biological resources, medicinal value and cultural importance (Magnurran, 2013). Broadly defined, biodiversity is ‘the variability among living organisms and the ecological processes of which they are part; this includes diversity within species, between species and of ecosystems and landscapes’ (CBD, 2012). Quantifying biodiversity is a complex task because there are so many aspects to cover, however it is possible to characterise biodiversity through the use of surrogate measures, from a landscape scale down to a genetic scale. A decision about which level of surrogacy to use depends on the scale of measurement and the resources available.

Many components make up biodiversity in order to express the variety of life. These can generally be divided into three categories: genetic diversity, ecological diversity, and the diversity of organisms (Heywood and Baste, 1995). Genetic diversity is the difference in genetic make-up between individual organism and populations; ecological diversity represents the ecological variation between biological units, such as niches and habitats; the diversity of organisms refers to the taxonomic variations from entire kingdoms to individuals (Heywood and Baste, 1995). Some elements of biodiversity are easier to define than others, but the most commonly used are species richness, species evenness and species heterogeneity (Magnurran, 2013). A high species richness usually infers greater genetic diversity, ecological diversity and organismal diversity (Gaston and Spicer, 2004). Species richness fails to account for species abundance, so evenness measures provide this wider aspect. Heterogeneity is a combination of species richness and evenness, therefore it can be used in comparison to the other two in order to determine which has a more influential effect on diversity.

2. Methodology

This study is located at Knepp Castle's 1400 hectare estate in West Sussex, southern England and took place between 14/7/2015 — 23/7/2015. Knepp Castle Estate's re-wilding project consists of the majority of this land which has been restored from agricultural use. The re-wilding project is divided into three areas, the southern block (470 ha), the middle block (283 ha) and the northern block (215 ha) (Fig. 6). Three study sites were chosen within each of these areas, totaling nine study sites, 27 quadrats and 9 line transect surveys. This was deemed an appropriate amount to gather the required data and ensure it is accurately representative of the vegetation (Gilbertson, 1985).

Knepp's re-wilding aims

- To improve the biodiversity of land by enabling natural processes to take place, encouraging the return of native wildflower, grasses, trees, shrubs, insects, butterflies, birds and small mammals, and all the other components of the ecosystem that once prevailed in this region of the British countryside
- To implement as near-natural grazing regime as possible, within the limits requires by animal welfare law and the production of top-quality meat for human consumption.
- To monitor the changes to landscape and biodiversity as the land slowly reverts to a more varied patchwork of habitats, away from the strict regime of arable fields and commercial plantation.
- To contribute to the scientific research on naturalistic and conservation grazing, and to inform habitat restoration projects.
- To advise our other Knepp Castles Estate projects such as field sports, recreation and forestry, so that any adverse impacts on wildlife are avoided or kept to a minimum .

Figure 5. Knepp's re-wilding aims as outlined on their website (Knepp, 2016)

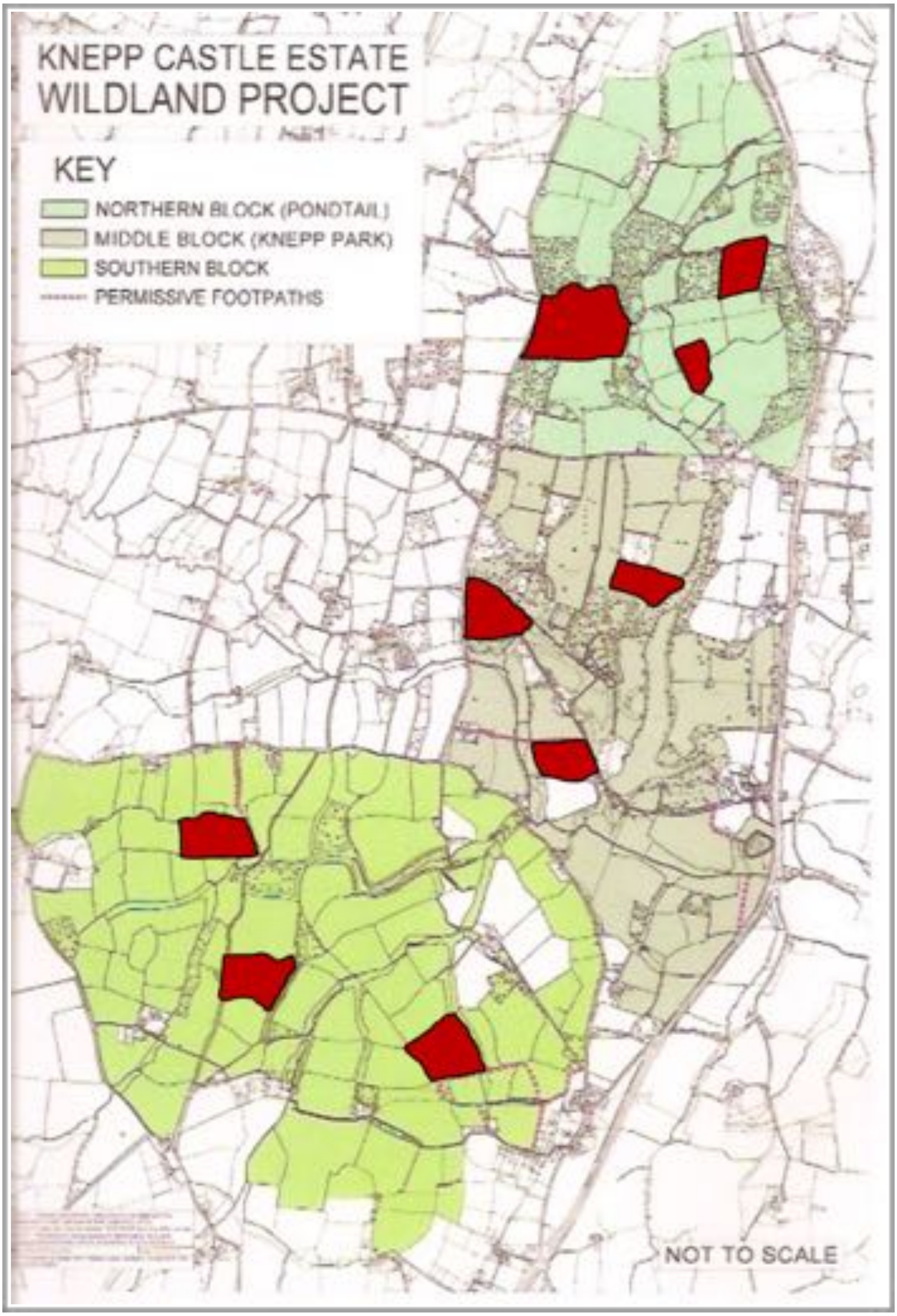


Figure 6. Map of the 1400 hectares of Knepp estate included in the re-wilding project. The key marks out each block and the red fields highlight the nine study sites, adapted from (Greenaway, 2011)

2.1 Site Selection

Figure 6 shows each block and highlights the nine study sites. The southern, middle and northern blocks have each been implemented with alternative grazing regimes (defined as a variety of grazing herbivores and stocking levels) and a different restoration method (defined as a technique to revert the landscape from its former agricultural production). Each block represents a form of re-wilding experiment. These experiments were decided in a combination of ways. Initially, in 2001 Knepp initiated a historic park restoration scheme using 500 acres of land. This grant aided scheme was created in agreement with DEFRA and

Area	Land restoration method	Grazing regime
Northern Block	Grass seed mix sown in 2005	Long-horn cows introduced in 2005 (roe deer and rabbits occur naturally)
Middle Block	High weald meadow mix sown in 2001	Red deer, fallow deer, Exmoor ponies, Long horn cattle and tamworth pigs introduced in 2001 (Tamworth pigs removed in 2014)
Southern Block	28 Fields – left fallow (2005) 7 Fields – left fallow (2006) 37 Fields – set-aside (2000-2004)	Red deer, fallow deer, Exmoor ponies, Long horn cattle and tamworth pigs introduced in 2009

Table 1. Description of each land restoration method and variety of grazers used in each block in Knepp's re-wilding project (Green, P, pers comm, October 20, 2015).

Countryside Stewardship. The scheme aimed to revert arable production to native grassland, mainly using deer (Knepp, 2016). A landscape architecture practice advised the restoration methods in a 'Restoration Management Plan' (Colson Stone, 2001). This scheme resulted in

Charlie Burrell taking the project further by introducing more land and a variety of grazing animals.

Southern Block		
Species	**Common name**	**Stocking level**
Bos primigenius taurus L	Long-horn cattle	130
Equus ferus caballus L	Exmoor ponies	11
Sus scrofa L	Tamworth pig	22
Cervus elaphus L	Red deer	30
Dama dama L	Fallow deer	235
	5 Species	**Total stocking level – 428**
Middle Block		
Species	**Common name**	**Stocking level**
Bos primigenius taurus L	Long-horn cattle	81
Equus ferus caballus L	Exmoor ponies	4
Cervus elaphus L	Red deer	31
Dama dama L	Fallow deer	410
	4 Species	**Total stocking level - 525**
Northern Block		
Species	**Common name**	**Stocking level**
Bos primigenius taurus L	Long-horn cattle	141
	1 Species	**Total stocking level - 141**

Table 2. Species and number of each grazing herbivore present in the three blocks during the time of study. This tables shows that the middle block has the highest total stocking level, the southern block the highest number of species and the northern block has the lowest stock level and one species of grazing herbivore.

This experiment aims to uncover which one of the three blocks will be more successful at achieving Knepp Castle Estate's aims (Figure 5). Table 1 describes the grazing regime and land restoration method used in each of the blocks, until summer 2015. In the northern and middle blocks the grazing herbivores were introduced straight away. Whereas in the southern block the grazing herbivores were introduced in 2009, after the fields had been set aside. The reason for this delay was because the available funds did not exist to erect a perimeter fence around the block, which is necessary to prevent the grazing herbivores from going beyond the estate boundaries (Green, P, pers comm, October 20, 2015). The northern block was reverted from arable production by spraying all the fields with herbicides, then planting a grass seed mix sown (Appendix II). The middle block had the same herbicide treatment and was sown with a high yield meadow mix in 2001 (Appendix II), as advised by land use consultants (Kernon, 2007). The southern block consists of 71 fields, all or which were set aside and left fallow, rather than sprayed with herbicides. In the southern block separate fields were taken out of production at different times, with the least productive fields reverted first (Green, P, pers comm, October 20, 2015).

Each block at Knepp has a variety of grazing herbivores introduced by the project, which roam freely within the block, and have been specifically chosen as analogues of ancient species. Table 2 shows the number of each species in each block in summer 2015, for example the Southern block has a total of 428 grazers consisting of 5 species of large herbivores which are divided into 130 Long-horn cattle (*Bos primigenius taurus L*), 11 Exmoor ponies (*Equus ferus caballus L)*, 22 Tamworth pigs (*Sus scrofa L*), 30 red deer (*Cervus elaphus L*) and 235 fallow deer (*Dama dama L*). Along with these introduced species the entire estate has grazing animals that exist naturally in the wider landscape. This includes thousands of rabbits, which were not part of the data analysis of this survey, since they have not been quantified by Knepp. Around 80 roe deer (*Capreolus capreolus L*) have also not been included because the species movements and time spent in each blocks is unknown (Green, P, pers comm, October 20, 2015)

Sites boundaries were determined by the pre existing field boundaries. These boundaries consist of mostly hedgerows, which provide an important habitat for small fauna. The nine study sites (three in each block) where chosen at random, by assigning each field

Fields set-aside in Southern Block	
Year field set-aside	Number of Fields set-aside
2000	4
2003	19
2004	13
2005	28
2006	7

Table 3. The date and amount of fields set-aside in the southern block after arable production was stopped and before grazers were introduced.

within the block a number, then using a random number generator to choose which field would be surveyed. In the northern and middle block all fields were introduced to the re-wilding experiment at the same time, therefore they were all included in the random selection. However, the southern block consists of 71 fields introduced to the re-wilding project at a variety of different times (Table 3), or kept as pasture or woodland. Only the 28 fields set aside in 2005 were included in the random selection because this year provided the widest set of possible sites within the same year, this maintains a more accurate comparison (Table 3).

2.2 Field methods

Once each site was selected vegetation surveys were carried out, were carried out as this is the appropriate process in order to provide data which indicates biodiversity (Magnurran, 2013). This included three quadrat surveys, one 15 m line transect survey, and one phase one habitat map per site. All notes were recorded in the field notebook, including additional details about the weather, time, GPS coordinates, animal species seen, and any further points of interest (e.g. evidence of soil rooting from pigs).

Figure 7. Example of a 1m x 1m quadrat used for surveying vegetation. southern block, site 2, quadrat 1, 18/7/2015.

Square quadrat plots (1 m x 1 m) were chosen in order to establish a standard sampling unit for describing the plant species composition of each site (Gilbert et al., 2009). This size provides a reliable representation because it is large enough to include significant numbers of individuals, yet small enough so that plants can be measured without duplication or omission (Cox, 1990; Barbour et al., 1987). Additionally, 1 m x 1 m quadrats are the recommended size for grassland and dwarf heath communities on the British Isles (Kent, 2011). This size was kept throughout all sites for continuity purposes. Random quadrat sampling was chosen because the common statistical techniques used to process ecological data are valid when data have been randomly collected (Elzinga et al., 2009). In order to determine quadrat placement, a grid overlay system was used (Gilbertson, 1985). Coordinates for the grid were randomly generated and provided a position to place each quadrat. The same technique was used to place the line transect. Three quadrats were surveyed on each of the nine sites, totalling 27 quadrats throughout the data collection. In each quadrat all species present were identified, this included only species rooted within the quadrat (Elzinga et al., 2009). Species were identified using the appropriate keys (e.g. 'Concise British flora in Colour' (Martin, 1982) and 'Wild Flowers of Britain and Ireland' (Blamey et al., 2003)). Additionally, a sample of each species recorded was taken from the site, pressed and kept as part of the herbarium (Appendix I). Where species could not be identified on site, further research took place in the lab with the collected specimens in order to identify species. Following that, the cover of each species was counted by calculating the percentage area of quadrat beneath any rooted species canopy (Fidelibus and MacAller, 1993). Moreover, height was measured in order to provide an understanding of the impacts grazing has on vegetation (Spehn et al., 2006). This was carried out by measuring a sample of each species with a ruler from root to tip.

Line transects were chosen as a supplementary technique to gain the broadest information of each site (Buckland et al, 2007). Each Line transect was 15 m and measured the presence and absence of species. Records were taken at 1 m intervals along the line transect, noting every species that intersected the line (Buckland et al, 2007).

Vegetation maps were drawn at each site, these maps were influenced by Phase 1 habitat surveys (JNCC, 2010). This is a widely used system that provides a rapid method of recording vegetation where each habitat feature is defined by way of a brief description, and is allocated a unique mapping colour or pattern (JNCC, 2010). Such surveys are often on a larger scale (Foulkes and Heard, 2003). The surveys for this report however, have been adapted to show the small scale (1:20) vegetation patterns present on each site.

2.3 Data analysis

The data analysis included both descriptive and inferential statistics in order to interpret the results. Firstly, descriptive statistics were carried out in order to summarise the raw data and enable patterns to emerge. Then inferential statistics were used to make deductions about the survey results and help provide general conclusions. This included calculating the mean abundance and height data for each site, block, species and function group. Where possible the line transect data and quadrat data was combined to give a total representation of each area, and therefore each block. However, some analysis requires only abundance (%) data and some requires counts of individuals, so in certain circumstances only one set of data was used. This approach was deemed acceptable because when line transects were carried out, no more than three new species were recorded in any given block. This number represents <10% of all species recorded. It was thus felt that either quadrat or line transect data provided an adequate reflection of species composition and still ensure the validity of the results. Following that, biodiversity indices were calculated which are single numbers or scores transformed from quantitive data, that summarise the characteristics of biological communities (Spellerberg, 2005).

Five biodiversity indices have been used to assess the species diversity of each block. Five indices were chosen after extensive research determined the appropriate index to analyse the data and complete the research objectives (Bibi and Ali 2013, Fowler et al., 2003, Magnurran 2013, Rowntree 2000). No single measure is adequate to describe the biodiversity of a community (Magnurran, 2013). Therefore, a selection of indices have been used in order to cover numerous components of biodiversity and gather a full picture of vegetation at Knepp. These are species richness, species evenness, species dominance and species heterogeneity (which combines species richness and evenness into one index).

a) Specie richness (S) – the most basic measure of biodiversity, a count of species present in each sampling unit (Magurran, 2013)

b) Shannon-Weiner diversity index (H') – a heterogeneity measure that was calculated using the formula below (Chipman and Johnson, 2002).

$$H' = -\sum(p_i * \ln(p_i))$$

Where $p_i = n_i/N$ and n_i =the abundance of the i^{th} species and N = the total abundance. Shannon-Weiner is a very popular ecological statistic for calculating biodiversity.

c) Shannon's evenness index (E)

A separate measure of evenness was calculated by dividing the reciprocal form of the Simpson's index by the number of species in each quadrat (Rowntree, 2000).

$$E = H'/\ln(S)$$

The evenness for each block was calculated by averaging the evenness results of all quadrats within that block.

d) Simpson's diversity index (D) – a heterogeneity measure that was calculated with the formula below (Rowntree, 2000).

$$D = \sum (n_i(n_i-1)/N(N-1))$$

Where n_i = the abundance of individuals in the i^{th} species and N = the total abundance. Simpson's index emphasises the dominant species and so was chosen along with Shannon-Weiner.

e) Berger-Parker dominance (d)

The Berger-Parker index expresses the proportional abundance of the most abundant species (Zuur et al, 2007)

$d = N^{max} / N$

Where N^{max} is the number of individuals in the most abundant species. The line transect data was used to calculate this because it involves numbers of individuals rather than cover (%) of individuals.

A rank abundance curve (also know as a Whittaker plot) was formed for each block using proportional abundance data (Magurran, 2013). Firstly, the relative abundance of each species in each block was calculated, and then ranked in order of most abundant to least abundant and displayed in a graph. The rank abundance curves visually display and compare species richness and evenness.

The other aspect of this study is grazing and so the grazing intensity of each block was calculated as an index. Using the stocking level data, the grazing intensity was calculated and statistically compared to biodiversity indices. Stocking level numbers alone do not have as great a significance so the calculation includes time and area grazed in order to provide context (Riddiford, 2002). This is because the same density of animals would have a very different affect in 100 hectares as they would in 1000 hectares. Grazing intensity was calculated using the following formula (Riddiford, 2002).

(Duration of grazing (months) x number of stock)/size of the area (hectares)

Before any inferential statistic could be carried out, normality tests were calculated for all the data sets. Most statistical analysis test are carried out under the assumption that the data is normally distributed, therefore a normality test is required to ensure data is normally distributed before any further statistical tests can take place (Ghasemi and Zahediasl, 2012). The Sharipo-Wilk test, rather than the Kolmogorov-Smirnov test, was used to decipher normality. Although the Kolmogorov-Smirnov test is popular in similar studies, more recently it has proved outdated and less suitable for small sample sizes (Razali and Wah, 2011).

Once it was established that all data sets had a normal distribution, further statistical tests were carried out. One-way ANOVA was used to assess the analysis of variance of the rank abundance curve. The one-way ANOVA compares the means between three of more groups and determines whether those means are significantly different from each other (Laerd statistics, 2013a). A one-way ANOVA is the appropriate comparative statistical test for data sets with two or more independent variable groups. It was chosen over a multiple T-test, which carries out a similar function but as the number of t-tests increase so does the probability of finding significant results by 'chance' (Park, 2003).

Pearson's correlation coefficient was calculated to compare the biodiversity indices with grazing intensity and mean height. Correlation analysis was chosen over regression analysis because it assesses the relationship among variables, rather than the relationship between an independent and dependent variable. While a correlation does not equate to causation, it indicates the strength and direction of an association between two variables. Pearson product-moment correlation attempts to draw a line of best fit through the data of two variables and is a measure of the strength of a linear association between two variables, denoted by *r (*Laerd statistics, 2013b)

3.Results

3.1 Data

3.1.1 Cover Data

The cover data for each block is shown in Table 4. All graminoid species cover was amalgamated and calculated under the one function group due to time constraints. Individual graminoid species were identified within quadrats, but only as a presence/absence count. The species identified in each quadrat (discounting the cover) can be seen in table 6.

Southern Block									
Species	Site 1	Site 1	Site 1	Site 2	Site 2	Site 2	Site 3	Site 3	Site 3
	Q1	Q2	Q3	Q1	Q2	Q3	Q1	Q2	Q3
Vicia tetrasperma L	11	22	3	10			1	14	8
Ranunculus repens L	4				0.25				
Rumex crispus L	10								
Pulicaria dysenterica L	65	30	70	30	30	0.25	75	2	32
Jacobaea vulgaris L	7						0.5	3	0.25
Graminoid (combined)	7	70	32	16	30	8	30	56	50
Quercus robur L		1			0.5				
Trifolium repens L		3						1	
bare ground		6		20	20	60			2
Cirsium arvense L			15						11
Lathyrus pratensis L			20						
Bryophyte				6	4	22	25	4	6
Dipsacus spp				1	0.5				
Geranium dissectum L							2		
Centaurium spp									0.25
Rubus fruitcosus L				68	28	0.25	14	42	
Cirsium palustre L				3					2
Anagallis arvensis L		0.25			0.5				
Cerastium fontanum (Baumg)		2							
epilobium parviflorum (Schreb)				3					
Salix cinerea L					55	12	0.25		20
Prunus spinosa L								8	
Cerastium fontanum B					2				4

Middle Block									
Species	Site 4 Q1	Site 4 Q2	Site 4 Q3	Site 5 Q1	Site 5 Q2	Site 5 Q3	Site 6 Q1	Site 6 Q2	Site 6 Q3
Vicia tetrasperma L									
Ranunculus repens L		3		2	1		2	28	
Graminoid (combined)	85	90	90	40	75	95	85	80	100
Trifolium repens L	33	18	18		12	6	22	28	2
Cirsium arvense L	2	2							
Lathyrus pratensis L				0.25		0.25		8	
Prunella vulgaris L				0.5					
Centaurium spp								0.25	
leaf litter				40					
Galium verum L				12	30	8			
Cerastium fontanum B				0.5					
Glechoma hederacea L	4								
Myosotis sylvatica L				0.5					

Northern Block									
Species	Site 7 Q1	Site 7 Q2	Site 7 Q3	Site 8 Q1	Site 8 Q2	Site 8 Q3	Site 9 Q1	Site 9 Q2	Site 9 Q3
Vicia tetrasperma L								0.5	
Ranunculus repens L		30	22	12	2	16	20	12	
Rumex crispus L	1					4			
Pulicaria dysenterica L									8
Jacobaea vulgaris L									
Graminoid (combined)	95	25	80	60	90	88	92	40	72
Quercus robur L									
Trifolium repens L		40	8	46	2		12	12	6
bare ground		4				4		3	6
Cirsium arvense L							14		20
Bryophyte								2	
Taraxacum spp		6							
Prunella vulgaris L								1	4
Cirsium vulgare (Savi) Ten						14			
Calystegia sepium L	15				4				
Ranunclus acris L		8	2		16				
Cyperaceae spp								30	

Table 4. Each block has been broken into sections. The species present in each block are recorded in rows. The site and quadrats surveyed are listed in the columns. The numbers recorded are the cover (%) of each plant species. For example, Site 1, quadrat 1 (Q1) in the southern block contained 11% *Vicia tetrasperma L*, 4% *Rannunculus repens L*, 10% *Rumex crispus L*, 65% *Pulicaria dysenterica L*, 7%

3.1.2 Height data

For each species that had cover (%) recorded, the height was also recorded. The data in Table 5 shows the average height data for each species in each of the nine quadrats per site.

Southern Block									
Species	Site 1 Q1	Site 1 Q2	Site 1 Q3	Site 2 Q1	Site 2 Q2	Site 2 Q3	Site 3 Q1	Site 3 Q2	Site 3 Q3
Vicia tetrasperma L	19	41	15	95			44	22	34
Ranunculus repens L	15				5				
Rumex crispus L	42								
Pulicaria dysenterica L	17	44	70	30	30	0.25	75	2	32
Jacobaea vulgaris L	50						60	58	15
Graminoid (combined)	34	50	7	83	43	73	23	38	47
Quercus robur L		19			20				
Trifolium repens L		3						8	
bare ground									
Cirsium arvense L			16						8
Lathyrus pratensis L			8						
Bryophyte									
Dipsacus sp				13	11				
Geranium dissectum L							46		
Centaurium sp									15
Rubus fruitcosus L				70	100	4	32	26	
Cirsium palustre L				114					20
Anagallis arvensis L		6			2				
Cerastium fontanum (Baumg)		5							
Epilobium parviflorum (Schreb)				69					
Salix cinerea L					145	560	30		32
Prunus spinosa L								135	
Cerastium fontanum B					5				4

Middle Block									
Species	Site 4	Site 4	Site 4	Site 5	Site 5	Site 5	Site 6	Site 6	Site 6
	Q1	Q2	Q3	Q1	Q2	Q3	Q1	Q2	Q3
Vicia tetrasperma L									
Ranunculus repens L		4		4	3		4	4	
Graminoid (combined)	10	11	19	9	9	18	13	9	12
Graminoid (flowers)	36	25	50	17	41	55	38	31	28
Trifolium repens L	9	5	12		8	5	3	5	2
Cirsium arvense L	30	12							
Lathyrus pratensis L				12		22		15	
Prunella vulgaris L				5					
Centaurium sp								10	
leaf litter									
Galium verum L				4	15	15			
Cerastium fontanum B				4					
Glechoma hederacea L	5								
Myosotis sylvatica L				3.5					
Euphorbia peplus L				8					

Northern Block									
Species	Site 7	Site 7	Site 7	Site 8	Site 8	Site 8	Site 9	Site 9	Site 9
	Q1	Q2	Q3	Q1	Q2	Q3	Q1	Q2	Q3
Vicia tetrasperma L								15	
Ranunculus repens L		5	6	6	4	5	9	7	
Rumex crispus L	56					81			
Pulicaria dysenterica L									25
Graminoid (combined)	15	10	16	11	13	9	17	17	21
Gramoinoid (flowers)	31	15	35	27	39	34	45	45	48
Trifolium repens L		8	5	8	6		10	3	8
bare ground									
Cirsium arvense L							72		42
Bryophyte								3	
Taraxacum sp		3							
Prunella vulgaris L								15	14
Cirsium vulgare (Savi) Ten						63			
Calystegia sepium L	13				16				
Ranunclus acris L		30	38		26				
Cyperaceae sp								50	

Table 5. The table shows the mean height (cm) recorded for each species in each block (broken into sections). Species are recorded in rows and location recorded in columns. For example, the average height of *Vicia tetrasperma* in site 1, quadrat 1, Southern block was 19cm.

3.1.1 Grass species presence

The grass species surveyed in each quadrat is shown in Table 6. Although the abundance of each species in the quadrat was not calculated, it was still deemed important to note what species were present. Therefore, these tables show the presence and absence data for each grass species

Southern Block									
Species	Site 1	Site 1	Site1	Site2	Site 2	Site 2	Site 3	Site 3	Site3
	Q1	Q2	Q3	Q1	Q2	Q3	Q1	Q2	Q3
Agrostis capillaris L	1							1	
Holcus lanatus L				1	1	1	1	1	
Anthoxanthum odoratum L		1							1
Bromus biebersteinii Roem				1					
Deschampsia cespitosa (L.) P. Beauv		1		1					
Agrostis stolonifera L		1			1	1	1	1	
Agrostis canina L								1	

Northern Block								
Species	Site 7	Site 7	Site 8	Site 8	Site 8	Site 9	Site 9	Site 9
	Q2	Q3	Q1	Q2	Q3	Q1	Q2	Q3
Agrostis capillaris L					1			
Holcus mollis L	1	1	1	1				
Cynosurus cristatus L						1	1	1
Holcus lanatus L	1	1	1	1	1	1	1	1
Arrhenatherum elatius L							1	1
Lolium perenne L					1			
Agrostis stolonifera L	1	1		1		1	1	1
Phalaris arundinacea L						1	1	1

Middle Block								
Species	Site 4	Site 4	Site 5	Site 5	Site 5	Site 6	Site 6	Site 6
	2	3	1	2	3	1	2	3
Agrostis capillaris L			1	1			1	
Holcus mollis L	1	1	1		1	1	1	
Cynosurus cristatus L			1	1	1			
Holcus lanatus L	1	1		1	1	1	1	1
Anthoxanthum odoratum L	1	1	1					
Bromus biebersteinii Roem					1			
Hordeum secalinum Schreb						1		
Lolium perenne L					1	1		
Deschampsia cespitosa (L.) P. Beauv	1							
Agrostis stolonifera L	1	1		1	1	1	1	1
Agrostis canina L	1	1	1		1	1	1	
Phalaris arundinacea L	1					1		

Table 6. The grass species found are recorded in rows and the location in columns. Where there is a 1 the species was present, where there is no number the species was not present. For example, Site 1, quadrat 1, in the southern block had only one grass species, *Agrostis capillaris L*.

3.1.4 Line transect data

The line transect data for each block is shown in Table 7 and includes all species surveyed along the line transect.

Southern Block			
Species	Site 1	Site 2	Site 3
Vicia tetrasperma L	19	3	4
Ranunculus repens L		5	
Rumex crispus L		3	
Pulicaria dysenterica L	23	11	4
Jacobaea vulgaris L			6
Quercus robur L		1	
Trifolium repens L	5	3	5
Cirsium arvense L	1	0	6
Taraxacum spp	2		
Dipsacus vulgaris L		1	
Prunella vulgaris L	1	3	
Cirsium vulgare L			1
Centaurium littorale D.Turner		1	1
Centaurium pulchellum Sw			1
Rubus fruitcosus L		2	5
Cirsium palustre L		4	1
Anagallis arvensis L			2
Epilobium parviflorum Schreb	1		1
Salix cinerea L		6	
Prunus spinosa L	4		4
Rosa canina L	2		1
Lathyrus pratensis L		1	2
Matricaria discoidea DC		1	
Graminoid (unidentified)	21	9	14
Agrostis spp	3		2
Holcus mollis L		2	
Holcus lanatus L	9	2	1
Bromus spp	2	3	
Juncus conglomeratus L	1		
Bryophyte	1	1	

Middle Block			
Species	Site 4	Site 5	Site 6
Ranunculus repens L		2	
Rumex crispus L	1		
Jacobaea vulgaris L	1		
Trifolium repens L	12	7	10
Cirsium arvense L	6		
Galium verum L		2	
Lathyrus pratensis L	2		1
Glechoma hederacea L	1		
Calystegia sepium (L) R.Br		2	
Graminoid (unidentified)	19	18	17
Agrostis spp	15	13	20
Holcus mollis L	7	2	8
Cynosurus cristatus L		10	1
Holcus lanatus L	10	12	12
Anthoxanthum odoratum L	2	7	
Bromus spp		2	
Arrhenatherum elatius L		1	
Deschampsia cespitosa (L) P.Beauv	1	2	
Juncus conglomeratus L			1
Hordeum secalinum L	1	7	2
Lolium perenne L	6	6	2
Phleum pratense L		9	1

Northern Block			
Species	Site 7	Site 8	Site 9
Vicia tetrasperma L			13
Ranunculus repens L	2	6	4
Rumex crispus L	3	1	
Pulicaria dysenterica L			
Jacobaea vulgaris L			2
Quercus robur L			
Trifolium repens L	11	10	7
bare ground		1	1
Cirsium arvense L	6		9
Taraxacum sp	3		
Argentina anserina L			1
Trifolium dubium L			2
Hypericum perforatum L			1
Calystegia sepium L	7		7
Cruciata laevipes Opiz	1		
Achillea millefolium L			2
Graminoid (unidentified)	17	18	19
Agrostis sp	9	10	0
Holcus mollis L	1	3	4
Cynosurus cristatus L			13
Holcus lanatus L	15	11	13
Bromus sp	2		2
Arrhenatherum elatius L			1
Lolium perenne L		7	
Phleum pratense L			10
Veronica chamaedrys L			1

Table 7. Line transect data with all species found is recorded in rows. The numbers represent the species count, for example, in site 1 in the southern block, *Vicia tetrasperma L* was counted 19 times along the line transect, *Pulicaria dysenterica L* was counted 23 times and so on.

Figure 8. Three photos comparing each block . (From top to bottom) the southern block (site 3), the middle block (site 4) and the northern block (site 9).

4. Data Analysis

4.1 Descriptive statistics

Descriptive analysis summarises raw data in a meaningful way and reveals patterns in the data. The descriptive analysis results visually show the vegetation composition of each of the blocks surveyed at Knepp. This provides a quantitative image of the biodiversity in its simplest form.

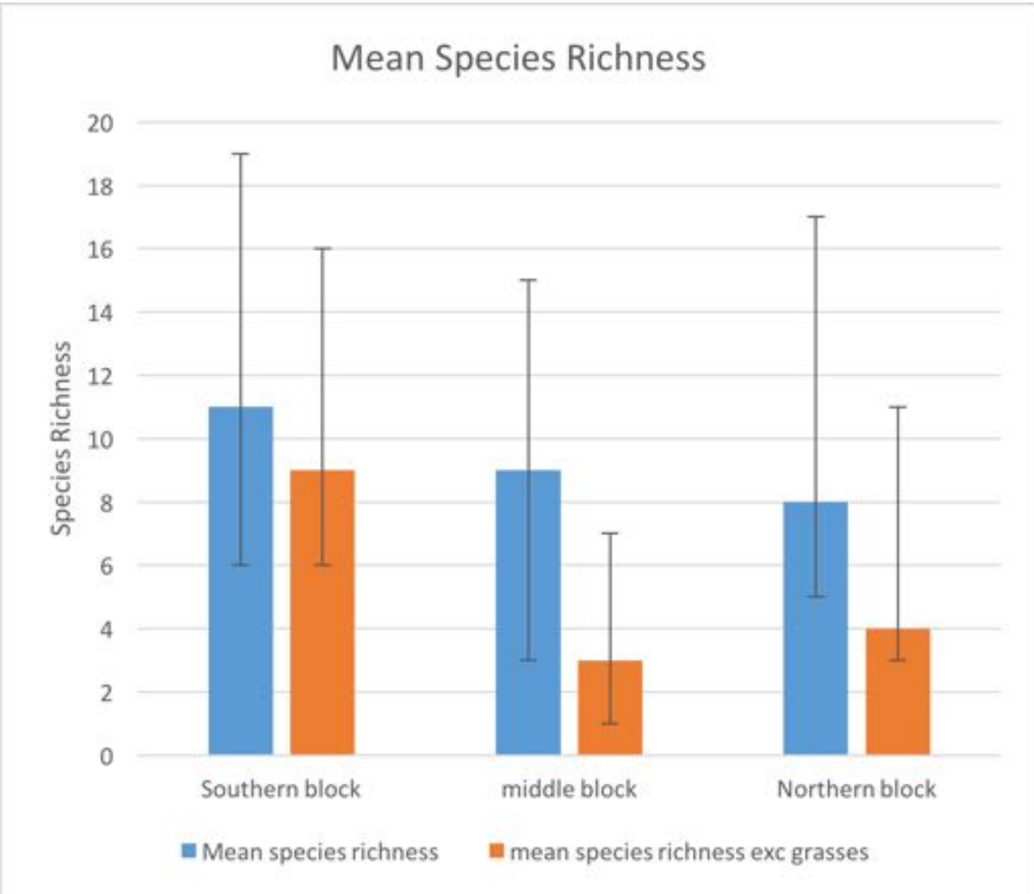


Figure 9. The bar graph compares mean species richness (including graminoid) and mean species richness (excluding graminoid) of each block. The error bars show the maximum and minimum number of species counted in a given site for each block.

In total, 49 species were recorded throughout the data collection. Species richness is the number of species found present in each site and Figure 9 shows the mean species richness for each block (based on the combined quadrat and line transect data). In terms of species richness, the graph highlights that the graminoid function group dominates in the middle and northern block. Once graminoids were discounted from the mean species richness count, in the middle block the number of species dropped more dramatically than either of the other blocks. The error bars indicate that the southern block has the highest maximum species richness. The northern block displays a high maximum species richness, but also the largest disparity between mean species richness and maximum species richness. Furthermore, an ANOVA was carried out to compare the species richness of each area. The p value was 0.23 (where 0 is completely different and 1 is entirely similar), therefore there is a statistically significant difference between the species richness of each site.

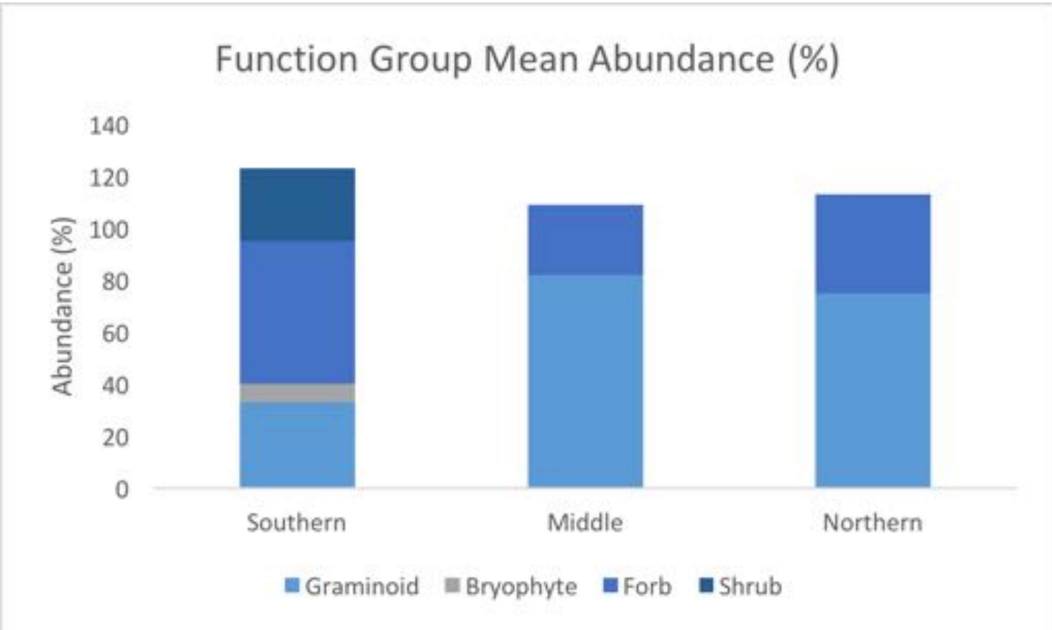


Figure 10. The bar graphs compares the mean abundance (%) of each function group surveyed in the three blocks. The flora surveyed at Knepp is divided into four key function groups. Graminoid (grasses, sedges and rushes), bryophyte (non-vascular plants such as moss and liverwort), forb (herbaceous flowering plant) and shrub (woody plants).

Additionally, it is important to include abundance data as well as a species count because communities could share similar species counts, yet differ greatly in the amount of each species. Figure 10 separates species cover (%) of each block into function groups which are categorised by species that share similar morphological and physiological properties (Pokorny, 2005).

Figure 10 shows that no shrub or bryophyte species were counted in the middle and northern blocks. Shrubs are not present in any quadrats in the middle and northern blocks, but occur (on average) in 40% of the quadrats in the southern block. The middle and northern

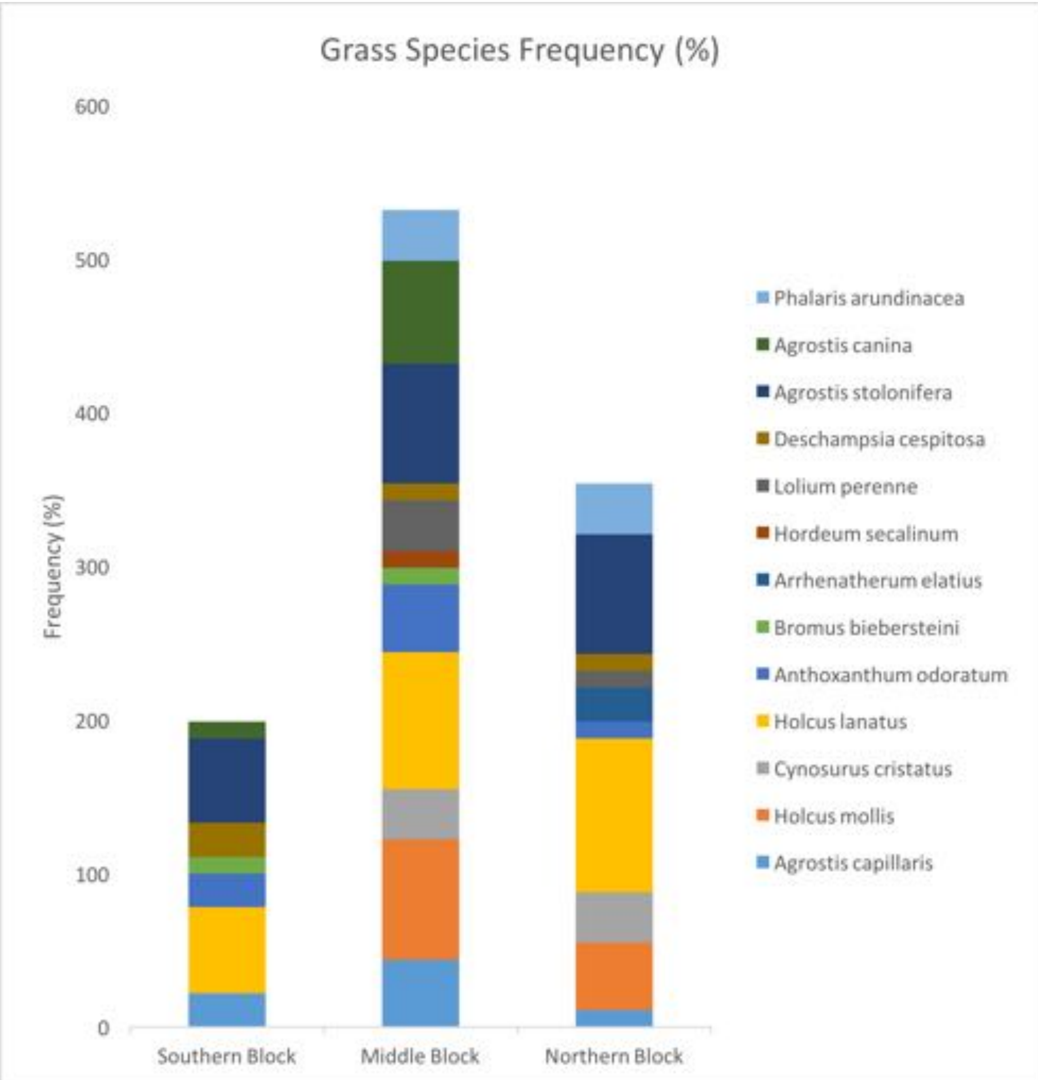


Figure 11. The species frequency (%) of each grass species recorded in each block, with all species names listed in the key.

blocks appear to have similar function group patterns, with graminoid representing the dominant function group, whereas forb is dominant in the southern block. Most notably, the southern block has species from each function group present, whereas the other two blocks do not. Additionally, an ANOVA was carried out comparing the function groups in each area. The calculation revealed a p value of 0.008, suggesting a very high statistically significant difference between function groups.

Figure 11 shows that in the southern block, graminoids have a mean frequency of 15.3% whilst the northern block graminoids have a mean frequency of 27.2%. Graminoids are most frequent in the middle block, occurring in 40.9% of quadrats. Unlike the other function groups, each block share the most common graminoid species, with *Holcus lanatus* and *Agrostis stolonifera* occurring the most often across all blocks.

4.2 Inferential statistics

Inferential statistics make deductions about the survey results and help provide general conclusions and predictions. The initial calculation was a rank abundance plot (Figure 12). Tables 8 and 9 show which species are most common, and which are the rarest. Note the tables omit graminoid species because it uses cover (%) data.

The relative abundance data of each species provides the information to produce a rank abundance curve (also known as a Whittaker plot) for each block to compare within a graph. One advantage of a rank abundance curve is that it clearly displays the pattern of all the species data in one graph. The graph provides a visual depiction of both species richness and species evenness. The x axis is the species abundance, where the most abundant species would be ranked 1. A steep gradient indicates low evenness and high dominance of a species in a community, and a shallow gradient represents high evenness as the abundances of different species are similar. Figure 11 indicates that all blocks have steep gradients and many species with small proportions, therefore each block is dominated by a small number of species. A normality test was carried out for the rank abundance data and ANOVA calculated to compare the variance of species distribution in each site. The p value = 0.89 (on a 0-1 scale), which would indicate that there is a statistically significant similarity between abundance proportions in each block.

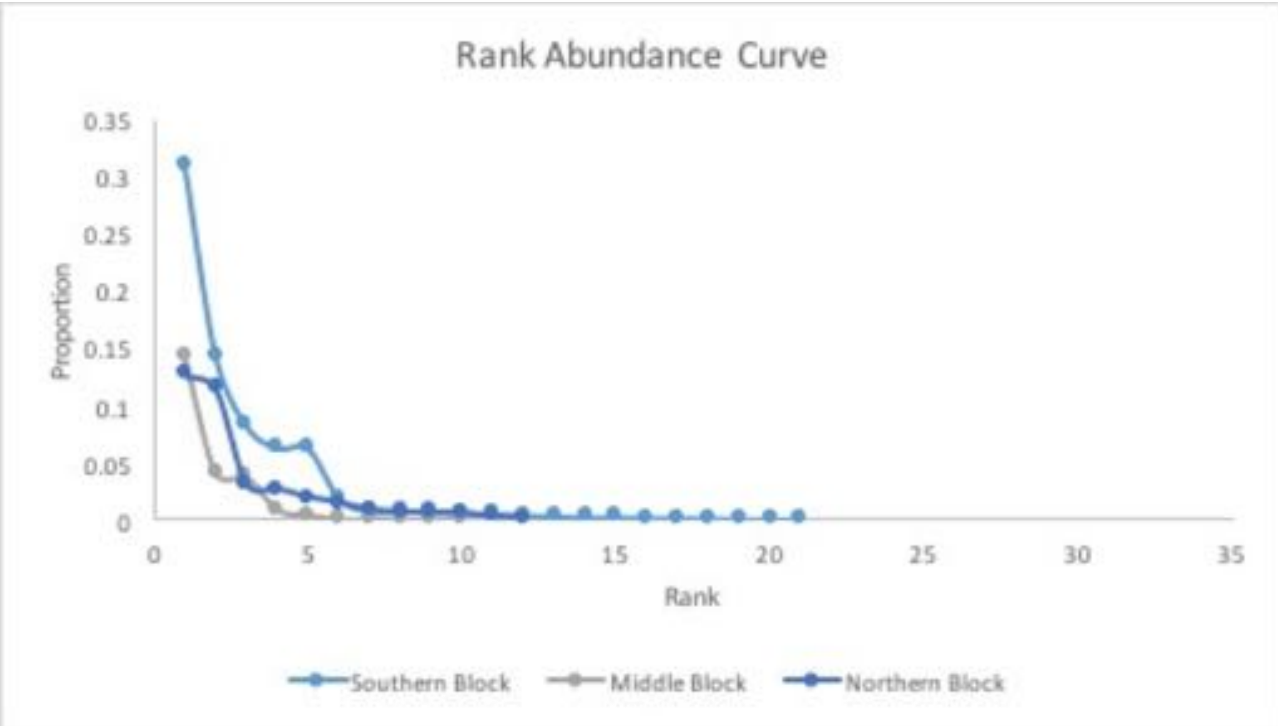


Figure 12. The rank abundance plot compares the proportion of each species in each block.

Tables 8 and 9 show that the middle and northern block share dominant species in common, which are all forbs. Whereas the southern block is unique, as the only block with dominant shrubs. Table x displays the least common plants in each block based on relative abundance. One of the southern blocks least common plants, *Rannunculus repens*, is one of the most common plants in the middle block. The middle and northern block share an uncommon plant, *Centaurium sp*. Although these plants are uncommon in the survey they are not necessarily uncommon wild flowers, with perhaps the exception of *Angallis arvense*.

Block	Species	Function group	Relative abundance
Southern	Pulicaria dysenterica	Forb	0.30
	Rubus fruitcosus	Shrub	0.14
	Salix cinerea	Shrub	0.08
Middle	Trifoloium repens	Forb	0.14
	Gallium verum	Forb	0.05
	Rannunculus repens	Forb	0.04
Northern	Trifolium repens	Forb	0.12
	Rannunculus repens	Forb	0.11
	Cirsium Arvense	Forb	0.031

Table 8. The three most common species and their function group for each block based on the species relative abundance.

Block	Species	Function group	Relative abundance	UK abundance
Southern	Centaurium spp	Forb	0.0002	Common
	Angallis arvense	Forb	0.0002	Rare
	Rannunculus repens	Forb	0.001	Very common
Middle	Centaurium spp	Forb	0.0003	Common
	Myosotis arvensis	Forb	0.0005	common
	Cerastium fontanum	Forb	0.0005	Very common
Northern	Vicia Tetrasperma	Forb	0.0005	Common
	Lathyrus pratensis	Forb	0.002	Very common
	Prunella vulgaris	Forb	0.005	Very common

Table 9. The three rarest species and their function group found in each block based on the species abundance. The final column in the table describes how rare each species is in Britain, ranking from very common to very rare.

4.3 Biodiversity Indices

Four indices are used to calculate biodiversity, the components that have been chosen to indicate biodiversity levels are heterogeneity, evenness and dominance. Note that cover (%) data was used to calculate all diversity indices and all graminoid species were included under one label. Table 10 shows the results for all the following calculations together. The results for the measures of heterogeneity include Shannon-Weiner (H): southern block – 1.40, middle block – 0.75, northern block – 1.04. Shannon-Weiner results are not on a 0-1 range, but the larger the number, the higher the diversity (results generally never exceed 4). Shannon-Weiner (H) results are then used to calculate evenness (Shannon (E)). A scale of 0-1 is used and 0 equates to no evenness and 1 to complete evenness. Simpson index (D): southern block – 0.68, middle block – 0.39 and sorthern block – 0.52. Simpson's diversity measure ranges from 0 – 1, where 0 is no diversity. Therefore, the southern block is the most heterogeneous with a moderate heterogeneity score, the middle block has a low heterogeneity score and so is a more homogeneous community. The results of the evenness analysis (southern block – 0.16, middle block – 0.32, northern – 0.2) show that all blocks have a reasonably low evenness. Most notably, the southern block has the lowest evenness even though it has the highest diversity. Furthermore, the middle block has the smallest disparity between indices.

	Species richness	Mean species richness	Simpson's diversity	Shannon-Weiner diversity	Evenness	Berger-Parker dominance
Southern Block	19	17	0.68	1.4	0.67	0.30
Middle Block	15	13	0.39	0.75	0.50	0.72
Northern Block	17	12	0.52	1.04	0.69	0.62

Table 10 . The table shows all the biodiversity indices results for each block. It provides a side by side comparison of the key statistical results for each block.

The final index is the Berger-Parker dominance index. In the Berger-Parker index, on a 0 -1 scale, 0 is a high dominance and 1 is a low dominance. Ordinarily, if a community has a high evenness it should have a low dominance, and vice versa. The results show: southern block = 0.30, middle block = 0.72 and northern block = 0.62. These results reflect the evenness calculations because the middle block was the most even, and therefore has few dominant species and the southern block was the least even with the highest dominance. Although a function group dominates in the middle and northern blocks (graminoid), there are numerous grass species within this function group. However, the southern block is dominated by one species, *Pulicaria dysenterica*, covering an average of 37% of quadrats surveyed.

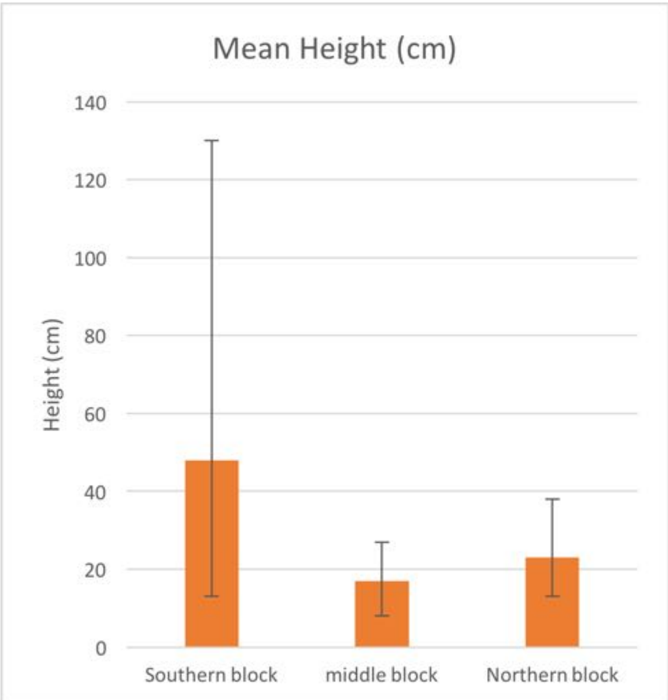


Figure 13. This bar graph compares the mean height of all vegetation in each block. The error bars show the mean height range using quadrat data. The southern block has the largest mean height (cm) as well as the largest height range.

4.4 Grazing

Grazing intensity was calculated. The results are southern block – 10.90, middle block – 22.26, northern block – 8.42. The higher the result number, the higher the grazing intensity. Therefore, the middle block has the highest grazing intensity by a considerable margin.

Height is also a consideration when assessing the impacts of grazing. Figure 13 shows the mean height data. The tallest species found in the southern block was *Salix cinerea*, standing at 5.6m. This is expected as the southern block is the only block with shrub species.

4.5 Correlations

	(S)	Mean (S)	(D)	(H')	(E)	(d)	Grazing intensity	Mean height (cm)
Grazing intensity	-0.77	-0.19	-0.73	-0.73	-0.99	0.55	1	-
Mean height	0.94	0.88	0.96	0.96	0.56	-0.99	-0.5	1

Table 11. The Table shows the Pearson's correlation values when comparing the grazing intensity and mean height to each biodiversity index. There are strong negative correlations between grazing intensity and the heterogeneity and evenness measures (Shannon's diversity index and Simpson's diversity index), as well as the species richness.

Correlation was chosen to deal with the relationship among variables, and interpret the relationship between grazing and the biodiversity indicators. Firstly, the normality of all data sets was calculated using the Shapiro-Wilk test. All data sets tested as parametric, and are therefore normally distributed. Following this, Pearson's correlation was calculated in order to compare all five biodiversity indices to grazing intensity and mean height (cm), the results are shown in Table 11.

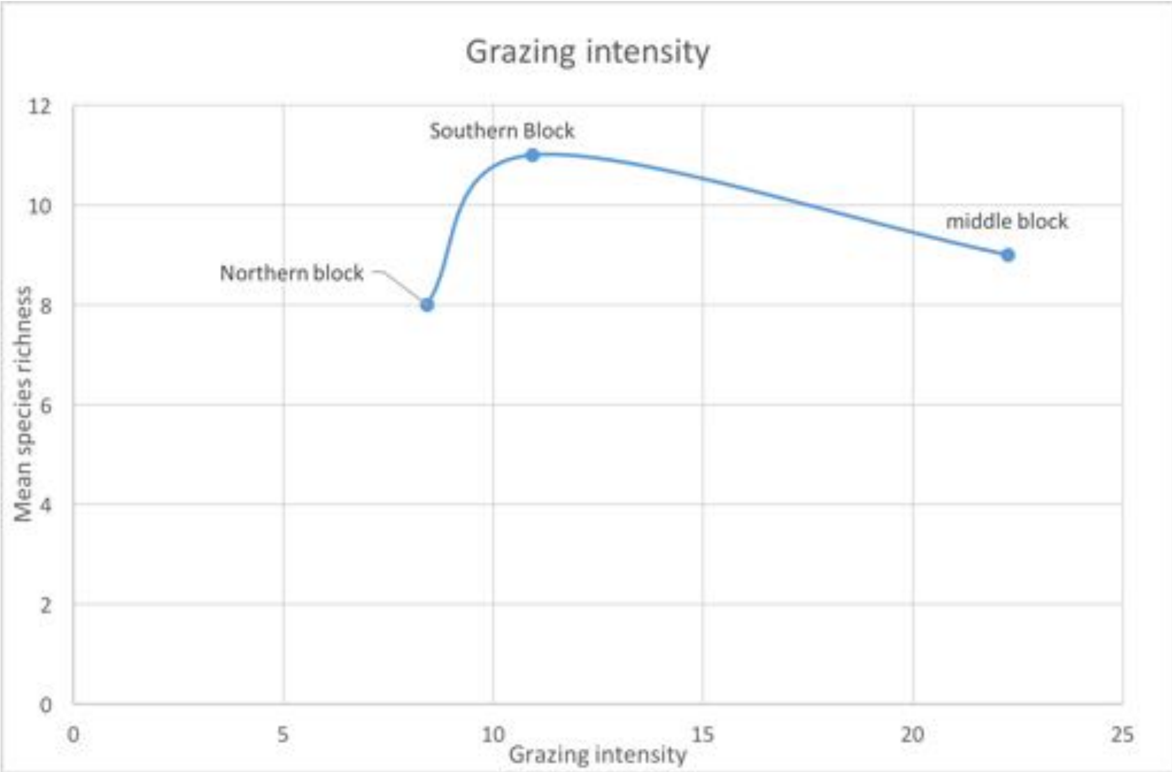


Figure 14. The scatterplot shows the relationship between grazing intensity and mean species richness for each block. As grazing intensity increases, so does the mean species richness. However once the grazing intensity reaches a certain level (marked by the southern block) it begins to decrease again.

Most notably, Table 11 shows that as grazing intensity increases, biodiversity decreases. These results are reflected in the middle block, which has the highest grazing intensity, as well as the lowest species richness. Figure 14 shows the relationship between grazing intensity and biodiversity, using species richness.

In addition, there is a moderate negative correlation between grazing intensity and mean vegetation height. Thus, as grazing intensity increases the mean vegetation height decreases. Furthermore, there is a very strong relationship between mean vegetation height and all biodiversity indicators. Higher vegetation corresponds to higher biodiversity and the southern block has the highest vegetation. Note the strength of the Pearson's correlation relationship is based on Table 12.

Value	Level of correlation
0.0 - 0.2	Very weak/negligible
0.2 - 0.4	Weak/low
0.4 - 0.7	Moderate
0.7 - 0.9	Strong/high/marked
0.9 - 1.0	Very strong/very high

Table 12. The table describes the strength of the Pearson's correlation results, displayed in Table 11 (Rowntree, 2000).

5. Discussion

This discussion highlights the complex and dynamic nature of the vegetation ecosystem at Knepp. The results reveal mosaic habitat patterns, varying stages of succession and the co-existence of species, which are all used as indicators of biodiversity. These factors are affected mainly by the impacts of grazing herbivores, but other components also affect the biodiversity in each block. It is thereby difficult to evaluate precisely which factor has the biggest impact on each block. A better understanding would require further experimentation and modelling. This report concludes that although the northern and middle block are lower in biodiversity than the southern, this is because they are at an earlier stage of Vera's vegetation cycle. The lessons that should be taken from these experiments are that: vegetation biodiversity benefits from a fallow period before the introduction of semi-natural grazing; and that the intermediate disturbance hypothesis supports the theory that an optimal grazing intensity (between 10-11) will encourage vegetation biodiversity.

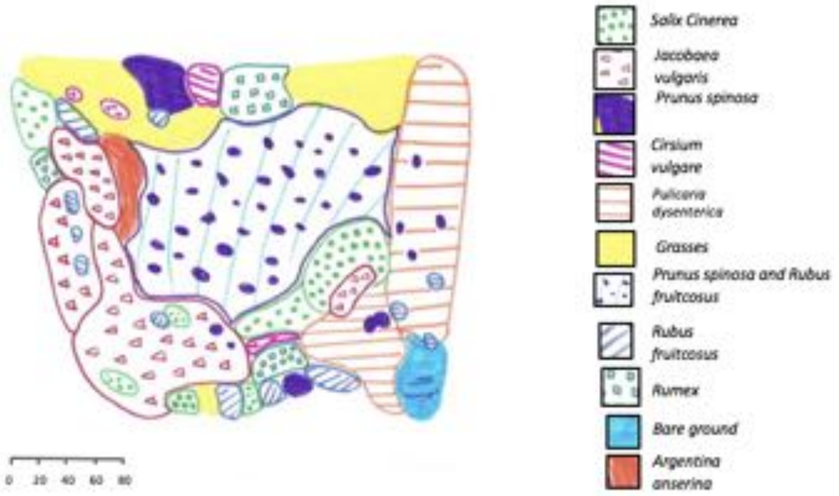
5.1 Biodiversity

Overall the results suggest that each block is different and biodiversity is highest in the southern block. The southern block has the highest mean and maximum species richness, as well as the highest heterogeneity scores and evenness score. The only biodiversity index that suggests the southern block has a low biodiversity is the Berger-Parker dominance index. However, this can be explained depending on the scale at which the results are interpreted. For example, although one species may dominate a site (e.g. *P. dysenterica* in site 1 and *S.*

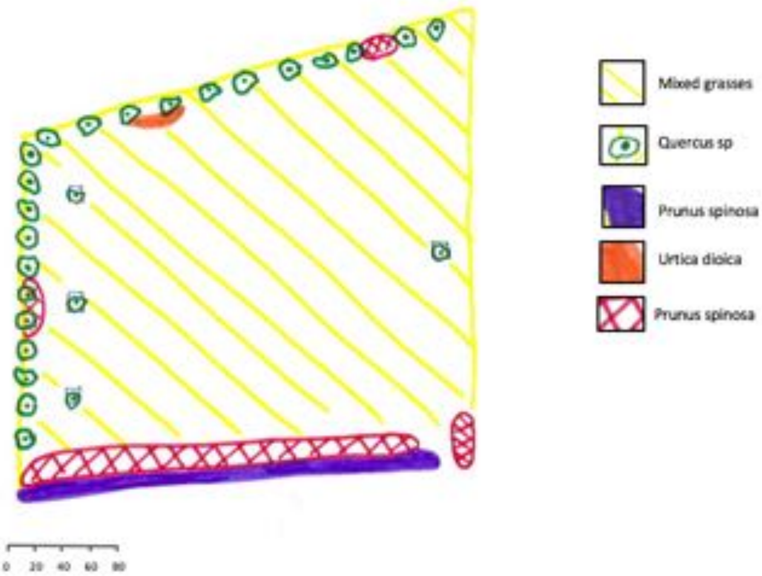
cinerea and *R. Fruitcosus* in site 2 in the southern block), the dominant species varies from site to site. This results in a mosaic habitat across the block, rather than within a quadrat or site. This can be seen when comparing the vegetation maps for the blocks (Figure 15). In comparison to the southern block, the northern and middle blocks have a low dominance of plant species even though the environment is more homogenous. This is because they consist of a high dominance of one function group in particular, graminoid. The vegetation maps show that grass swards make up the majority of each site in the middle and northern block, and that there is little mosaic vegetation from site to site. Instead, there are mosaic patterns of grass species on a smaller scale within each field. Mosaic habitat structure represents a more biodiverse community because it provides the conditions for numerous different species of flora (and fauna) to co-exist (Magnurran, 2013).

The results also reveal that the middle block has the lowest species diversity, which is surprising considering the block appears to be more similar to a natural grazing regime than the northern block. Arguably the southern block has the least amount of human intervention on account of it being the closest to a natural landscape (mixed grazers and no herbicide restoration). According to this logic the middle block is the next nearest a natural landscape because it has a variety of grazers even though it was treated with herbicides. The northern block is the farthest from a natural grazing system as it holds only one species of grazer and was treated with herbicide. However, the middle block still displays the lowest vegetation biodiversity. The two main differences between the northern and middle blocks are the variety of grazers and the grazing intensity. Considering this, the two similarities between the southern and middle block are the variety of grazers, this suggests that the grazing intensity in the middle block has a greater impact on vegetation biodiversity than the variety of species grazing.

Site 3



Site 5



Site 7

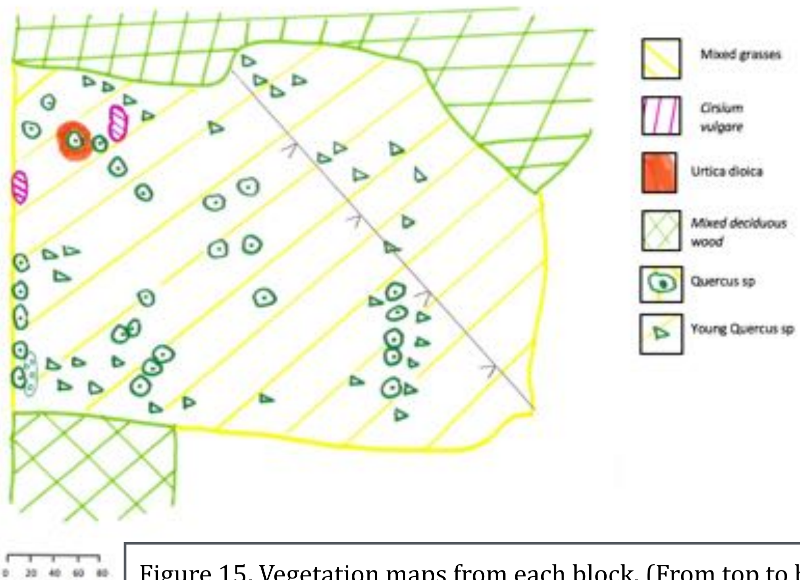


Figure 15. Vegetation maps from each block. (From top to bottom) site 3, southern block, site 5, middle block, site 7 northern block. The keys show the prominent species. (Appendix V)

5.2 Vegetation succession

The plant species present in a habitat help identify which stage of succession a community is in, and vegetation succession plays a role in vegetation biodiversity. Research suggests that biodiversity is higher in areas at a later stage of succession and that this is improved with intermediate intensity of disturbances (Binelli et al., 2000). Traditional models of vegetation succession are linear, assuming the vegetation grows from a primary stage, with a few grass and forb species. These species are then gradually replaced by shrub and tree species and ending with climax vegetation (Binelli et al., 2000). Vera (2000) proposes that natural vegetation succession is not linear but instead cyclical. He proposes that disturbances (such as grazing herbivores) in the landscape break up succession patterns, causing a mosaic of age classes and communities. This increases biodiversity since different species of flora and fauna thrive in alternative succession habitats (Ohlson et al., 1997). Therefore, the lower biodiversity of the northern and middle block does not necessarily suggest they are stagnantly experiencing low biodiversity, but that they are simply at a different stage of cyclical succession.

Pioneering species help identify the early stages of succession and are present in high abundance throughout the northern and middle block. Pioneer species are the first species to establish in a community. They often grow well and quickly in poor soil; have nitrogen fixing properties which enhances soil fertility; and are intolerant of shade. Grasses are an example of a pioneer species. Ecologically, grass species provide good structural support for forb establishment as they bind soil together with a fibrous root system. This in turn increases species biodiversity due to a new function group beginning to establish. Grasses are necessary players in a dynamic ecosystem because they provide food and shelter for wildlife. However, too few grasses in an ecosystem means it is unlikely to progress beyond a pioneering stage because it does not support other biota. Conversely too many grasses and they will dominate over time, which also does not support biota development. The southern block has the lowest mean abundance of grasses (33%), implying it has moved on from the early stages of succession. On the other hand the middle block is still dominated by grasses (82%) which has subsequently retarded succession. This result is unexpected considering that grazing intensity is highest in the middle block. It is therefore evidence of the Intermediate Disturbance hypothesis (IDH). IDH suggests that species diversity is at its highest when ecological disturbance (such as grazing, fire, human impact) is neither too frequent nor too rare (Roxburgh et al., 2004). According to the IDH intermediate levels of disturbance maximise species diversity because species that thrive at numerous stages of succession can coexist (Roxburgh et al., 2004).

Figure 16. Site 1 in the southern block, with a large patch of *R. fruitcosus* in the foreground. Knepp 16/7/2015

The southern block is indicative of the next stage of succession in Vera's cycle, a stage which leads to the establishment of forests, and shows examples of species co-existing. Herbivores pose a threat to the regeneration of forests because tree saplings are vulnerable to grazing in open grassland. However there are certain species that play a crucial role in protecting tree saplings which in turn allows for woodland to establish. For example *R. fruitcosus* and *P. spinosa* are useful shrub species because they protect themselves with features like thorns. In all three sites of the southern block, both these species (which have a close relationship with *Quercus sp)* were well established (Figure 16). *Quercus sp* was also present on the site and is one of the main tree species in European forest communities (Vera, 2000). It does not establish well in shaded forests, instead open grasslands provide suitable conditions for the shade intolerant species. The acorns of *Quercus sp* are often hidden in shrubs by birds, like Jays, and germinate well under the thorny shrubs protective conditions until they are established enough to be protected from larger herbivores to then start shaping the new forest (Gomez, 2003). Examples of this can be seen in the southern block, where *Quercus sp* was found encircled by *R. Fruitcosus*. *R. Fruitcosus* is a native species that grows well in cultivated lands and is extremely useful for establishing native woodland. Additionally, this plant is a beneficial insect attractor as well as a source of food and shelter for wildlife. The role these species play show that they are a support network for numerous biota which in turn highlights the complex interplay of dynamic ecosystems.

S. cinerea is another species which was abundant in the southern block, particularly site 2 (Appendix V). This tree species is demonstrative of the early stages of forest succession. Unlike *Quercus sp* this species establishes well with little help from other plants because it is fast growing, tolerant to exposure and the seeds are light and easily dispersed (Newsholme, 1992). As well as the plant species, the plant height also indicates vegetation succession, *S.Cinerea* was recorded growing up to 5.6 m in the southern block (Table 5) and Figure 17 shows the dense growth in site 2, this well established species has only grown this tall and thick in one of the sites surveyed, this shows that a mosaic habitat is forming between sites.

Figure 17. Dense *Salix cinerea* in site 2. Knepp, 17/7/2015

5.3 Co-existence

A combination of mutualism and competition fosters species co-existence in a diverse community. *T. repens* is the most abundant forb in both the middle and northern block and plays a specific role as a pioneer species. It is a common weedy herbaceous plant with sprawling stems that is native or naturalised across most of the temperate regions of the world (Ohlsen et al 2008). *T. repens* harbors microbes in its root system that are capable of fixing nitrogen and producing a more fertile soil. One of the reasons this species is so abundant in the middle and northern blocks is the symbiotic nature it has with particular grass species. Grass species play a key role in *T. repens'* struggle for niche establishment. *T. repens* lives in close harmony with grasses such as *Lolium perenne* because of the marked asynchrony of growth cycle, therefore reducing mutual interference (Levine and HilleRisLambers, 2009). The results reflect this since *T. repens* was flowering at the time of survey collection and L. perenne was not often counted as it was not flowering. This example shows how *T. repens* responds to grass species pressure using a form of micro evolution and how plant species play a dependent role with each other when establishing their niche space. Ecological niches are central in understanding the spatial patterning of a community. An ecological niche relates to the specific environmental conditions a species will live under (Schoener, 2009). The niche represents how an organism responds to the distribution of competitors and resources and how it can, in turn, alter these factors (Peterson et al., 2011).

Evidence of co-existing species in a niche space connects to a long running debate within ecological biodiversity. The traditional argument is that niche differences allow species to share out the environment. That two species competing for the same resources cannot co-exist and eventually the inferior species will be eliminated (Hutchinson, 1961). However this theory lies in contrast with the results in this report, species co-existing. Other studies resolve this enigma by reasoning that symbiotic relationships between differing species properties prevent the elimination of competitors. This is a result of niche differences, where species that are different co-exist because they each take what they need from the niche space, which maintains diversity and prevents competitive exclusion (Chesson 2000, Adler et al. 2007). This theory does not discount species competition, but rather supplements it, in order to highlight that species mutualism also occurs and is a sign of a diverse community (Bachelot et al., 2015). Additionally, this theory has been found to closely like with IDH, which acts as a mechanism for species co-existence (Roxburgh et al., 2004). The variety of plants co-existing, whether in competition or mutualism, show how species utilise niche spaces and create species-rich habitats which improve biodiversity through their interaction with one another.

5.4 The effects of grazing on vegetation

Competition and facilitation between grazing species strongly contributes to a richness of biodiversity (Vermuelen, 2015). The effect grazing has on vegetation can depend on the individual species of grazer, with each species playing a unique role in shaping the habitat. Cattle, ponies, red deer and pigs are grazers whereas roe deer and fallow deer are predominantly browsers. Grazers are species which feed mostly at ground level, clipping grasses and some forb species. On the other hand browsers feed on woody and shrub species, eating leaves and stripping bark. Each herbivore has eating patterns which affect the architecture of a plant. For example, cattle use their tongue to pull material, whereas ponies, in a similar way to rabbits, tend to nibble and eat shorter grass that cattle cannot graze on. Both ponies and cattle prefer grassland grazing and often avoid certain species such as hairy grasses and hardy shrubs unless there is poor food availability. It has been observed by WallisDeVries et al (1998) that such grazing animals transform grasslands dominated by a few common grass species (such as *holcus lanatus* and *agrostis capillaris*) into a mosaic sward of various grass species. This was the case in the middle block since it has a high number of grazers and the most diverse variety of grass species. However, this is the only indicator of high biodiversity in the middle and therefore this block is not the most suitable re-wilding experiment.

Pigs have slightly different grazing habits, unlike cattle and ponies they have been found to prefer woodland areas at Knepp (Johnson, 2011) and therefore play a key role in the later stages of succession. Pigs clear the woodland ground layers, allowing light to reach the forest floor, which improves regeneration. They are useful for preventing certain plants dominating because they target species such as bramble, bracken, and willow, which are known to dominate landscapes (Brickell, 2010). Johnson (2011) found that they are preferential grazers for white clover and grasses, both of which have been surveyed growing together in a symbiotic relationship (as discussed previously). This is further proof of the dynamic ecosystem in which animals and plants cooperate with each other and evidence that such a system is at work at Knepp. However, pigs can also cause considerable damage due to rooting (Figure 18). This can destroy young establishing plants and create large areas of bare soil (Welander, 1995). They eat the roots of plants using their noses to find the root by churning up the soil. Rooting was observed in site 3. It can be beneficial for managing invasive and dominant species, such rhododendron, which is conducive to creating conditions for wider biodiversity (Welander 1995). The effect of pigs is similar to the intermediate

Figure 18. Evidence of pig rooting in site 3, Knepp 18/7/2015

disturbance hypothesis: too much is detrimental whilst too little reaps few benefits. Considering these circumstances, it is important to remember that Knepp is aiming to let natural processes predominate as much as possible. Therefore, grazing cannot contribute to the biodiversity of every plant species and there will always be winners and losers within a habitat.

The advantage of having a variety of grazers and browsers leads to partitioning of food resources, since species share some food habitats and not others. This ultimately allows herbivores to strike a balance and contribute to creating a mosaic landscape and encouraging vegetation biodiversity. This occurs because the interaction of different herbivores facilitates a small amount of competition, while also exploiting different diets so the animals can co-exist. Thereby explaining why the northern block, which only has one species of grazer (discounting all deer and rabbits) does not have the highest biodiversity index results and shows less evidence of succession and mosaic habitat (Appendix V).

As well as the herbivores preferences to certain species, the behaviour of each animal will affect the vegetation biodiversity. Animal movement is a natural behaviour and certain animals will favour, and be more active, in particular blocks of Knepp. There are certain reasons which motivate this behaviour, such as: pack dynamics; soil moisture; security in protected areas (narrow, ditches, trees); routine pathways (Figure 19); areas where preferred plant species grow; and areas with higher nutritional value in the plants due to previous use (Palmer et al., 2005). As expected, low use of certain fields by the herbivores will encourage sapling recruitment and scrub expansion through associational resistance (Loucougaray et., 2004) and contribute to mosaic habitat structure leading to increased biodiversity. Examples of social structure that influence spatial structure include being less mobile during birthing season, which means that birthing grounds are grazed more intensively and other areas are given a recovery period (Vermeulen, 2015). Eating in one place and defecting somewhere else enables the transport of nutrients, stimulating differences in local soil fertility because horse dunghills and cattle manure are high in nutrients (Vermeulen, 2015). Furthermore, dominant stallions tend to mark their territory competitively with other stallions using dunghills, this can create nutrient hotspots for vegetation; young male bulls or stallions may migrate to new habitats in search of maters and dominant bulls dig bull pits

Figure 19. A regular path trodden by grazing herbivores in site 1, southern block. Knepp, 16/7/2016

which disturb the soil (Vermeulen, 2015). The behaviours unique to each species have a significant impact on biodiversity because each plant species has its own niche. Heard behaviours can facilitate these niches so that eventually, dozens of plant species grow adjacent to one another.

Grazing herbivores play an integral role in creating mosaic habitats and influencing vegetation succession, thereby improving vegetation biodiversity. The data from this survey suggests that both mosaic habitats and various stages of succession have emerged as a result of the re-wilding experiments. Both the southern and middle blocks consist of a variety of grazing herbivores but the southern block is more successful in terms of biodiversity. Although admittedly the middle block has improved this is most likely due to the fallow period the southern block was given, or alternatively because the stocking levels of the middle block are too high. Thus, it would appear that the middle block is at an earlier stage in succession.

5.5 Stocking levels

The number of grazing animals will affect their natural behaviour as well as the vegetation because in their natural density, each species should be able to exploit their surroundings. Despite this however, Knepp has experimented with different stocking levels since most references to animal densities are based on human controlled systems. Some general information about conservation grazing most likely contributed to the decision, for example low stocking densities (lower than agricultural stocking densities) are important in order to allow vegetation to grow and have vegetation succession take place. Whereas, a high stocking density can lead to a homogenous environment, scrub dieback, and the reversion of grassland due to browsing increases and the prevention of new growth (Newton et al., 2009). This suggests a balance must be struck so that stocking levels can create a mosaic habitat with some scrub expansion that does not expand uncontrollably (Vera, 2000).

Figure 14 is evidence of the intermediate disturbance hypothesis. As grazing intensity increases so does species richness, yet only to a point. Connell (1978) proposed that too little disturbance results in a low diversity because of competitive exclusion, and too much disturbance removes species incapable of rapid recolonisation. Figure 14 is indicative of the IDH and is an example of the 'hump-backed model', which displays the relationship between species diversity and the intensity of environmental stress. The graph first appeared in Grimes (1973) paper 'Competitive exclusion in herbaceous vegetation' and has been a widely supported by ecological studies ever since. Figure 14 from this study closely resembles the hump-back species richness curve calculated in other similar studies (Graham and Duda, 2011). Both these theories combined with the data are evidence that the grazing intensity in the southern block is the most favourable to improved vegetation biodiversity and that it is too high in the middle block and too low in the northern block.

5.6 Effects other than grazing herbivores

Large herbivores are most likely not the only determinant impacting vegetation biodiversity. The mosaic habitats and variation of plants from field to field and between blocks could also be a result of numerous other factors. For example, a wet May is likely to promote the growth of wind dispersed seeds and create a particularly good mast year (Howe and Smallwood, 1982). Additionally, the time of year may affect the vegetation biomass, since ecosystem productivity and the duration of a season determine plant densities, and therefore the number of grazers (Gorme et al., 2012). The fields with *Rosa canina* (wild rose) are most likely due to seed dispersal through bird droppings and have subsequently self-perpetuate (Green, P, pers comm, October 20, 2015). The pre-restoration use of each field may also influence vegetation. The land had been used in arable farming and certain arable crops and pesticides used in farming affect the soil composition and fertility differently and therefore the vegetation (De Vries et al., 2012). In addition, earthworms are extremely important in soil health. They are instrumental to several ecosystem services the soil provides, such as nutrient cycling, drainage, and regulating greenhouse gas emissions (Willem van Groenigen et al., 2014). Certain factors influence the recovery of earthworms in the soil, for example Crowther et al (2012) found that a low intensity of insect grazing directly relates to high earthworm abundance. Therefore earthworms may have recovered in some fields at Knepp and not in others, which would in turn affect the soil fertility and vegetation biodiversity. Although this report focuses on vegetation biodiversity Knepp is also aiming to improve animal biodiversity in small mammals, birds, insects and butterflies. The effects on animal biodiversity have increased rapidly since the project began (Greenaway, 2011) and animals such as birds are a good 'yard-stick' for general levels of biodiversity since they rely on insects which benefit from a biodiverse environment (Brouwer and Crabtree, 1999). Therefore, because faunal biodiversity has improved, vegetation biodiversity is also likely to improve. Many factors will affect vegetation biodiversity at Knepp and no component works independently, all impacts are connected because of the dynamic nature of ecosystems. Henceforth, influences other than grazers will impact biodiversity. This being said however, these factors are unlikely to have as significant an impact as the large herbivores.

5.7 Wider implications

A study of an experiment such as the one at Knepp cannot be assessed without also considering the role of humans play, as well as addressing the controversy it has faced. The re-wilding experiments were initiated with human selection. It was people who decided on the stocking levels, species of grazers and restoration methods. These measures were taken in the hope that eventually humans will be able to let go, allowing for natural selection to take over. For example, in the 'wild' large grazers would be susceptible to diseases, which might cause a temporary population crash. This lightens grazing pressure and leaves room for rapid growth. However, it is difficult to recreate a pre-human landscape in such a way, because we now live in a noticeably anthropogenic landscape (Bastian and Burnhardt, 1993). This highlights one of the many difficulties a re-wilding project such as Knepp faces. For instance, diseases in wild animals are now less common because they are protected by human livestock interests or animal rights (Greenaway, 2011). Such hurdles can be overcome, this type of 'waiting' period was simulated by the fallow stage in the southern block. This proved helpful in recreating a natural process in addition to aiding the advancement of succession.

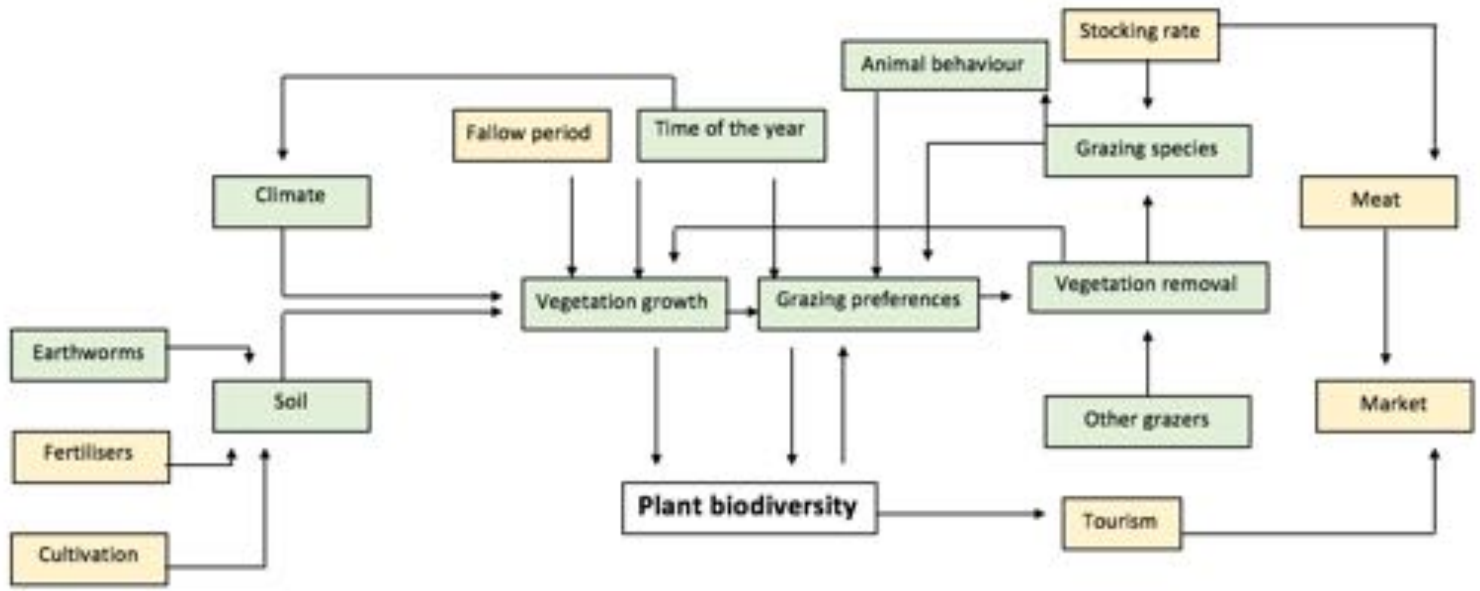


Figure 20. A flowchart that represents the processes discussed in this study which affect plant biodiversity. Boxes marked yellow are influenced by human management and the market. Green boxes are ecological features.

Re-wilding experiments have been criticised for attempting to do what is thought impossible, turn back time and consider a landscape without humans (Rubenstein et al., 2006). But a study like Knepp is not under any delusions that the landscape will ever be exactly how it once was, the project understands that future nature is unlikely to be the same as past nature due to of social evolution's permanent effects on the landscape, such as the industrial revolution (Warren, 2014). Instead, Knepp has taken the pre-human as a guide, a guide unlike many other conservation or wilderness managements that seeks to contain or suppress natural processes, or control the environment for the benefit of one species (Vermeulen, 2015). Knepp aims to withdraw human intervention in order to let natural processes prevail. Knepp is not unrealistic, just extremely ambitious, in aiming to create a future in which humans and nature are equal players in a global ecosystem.

One element in the success of Knepp is not just seeing changes in the landscape but also changes in societies preconceptions of nature. Humans current views about forests and succession are often strongly influence by what they are used to and comfortable with (Swift, 2009). A study by Swift (2009) found that some local people were averted to the aesthetics of Knepp because it did not match the surrounding manicured cultural landscape of the Sussex countryside. Swift's findings position this vegetation study in the wider implications of Knepp's success. That, data aside, it also requires a sense of open-mindedness in the local people who must be willing to appreciate that semi-natural grazing is a natural process. Along with continual monitoring and further experimentation, in order for the Knepp project to progress, it is crucial that the benefits of re-wilding are disseminated. Fundamentally, this project is not a surrender to nature, but a chance to experiment and learn from nature with the aim of working increasingly with nature's rhythms, rather than dictating them.

Re-wilding, as a concept and restoration tool, is still in its infancy and undergoing preliminary experimentation. With this in mind the results from this study support Frans Vera's woodland hypothesis. This is because mosaic habitat formation is present, which is an indication of the beginning stages of open canopy woodland. Support for Vera's hypothesis is important for the future of re-wilding in Europe since it provides a standard for pre-human landscapes which other European re-wilding projects can emulate.

5.8 Suggestions for the future

This study has exposed some suggestions for the future of Knepp's re-wilding project. Firstly, it is important to consider the youth of the project. It highlights that extreme changes in vegetation are more likely to happen on a century scale, rather the decadal. Another important lesson is to repeat surveying the sites used in this study in order to monitor any changes in vegetation. What may be useful is a prediction model that weighs up and includes the multiple factors affecting the vegetation. Figure 20 shows just how complex the components influencing plant biodiversity previously discussed truly are. Another suggestion that would make coming to a more definitive conclusion possible would be to conduct further, and more controlled, experiments at Knepp. After all Knepp's entire re-wilding project is fundamentally a set of experiments.. As a result, although three variation of re-wilding experiments add to our understanding of semi-natural grazing, they by no means provide sufficient information to advise similar projects. Whilst this study is realistic about the uncontrollable behaviour of in-situ experiments, a shortcoming of the experiments is the number of independent variables in each block. This is because using a different combination of grazing species, grazing stocking levels and restoration management in each block makes it difficult to pinpoint which aspect of the experiment causes the change in biodiversity. Therefore further testing should include fewer independent variables per experiment.

6. Conclusion

This study sought to determine the impacts of semi-natural grazing on vegetation biodiversity at Knepp Castle Estate's re-wilding project. Re-wilding as a form of conservation is a relatively new enterprise. The aim of re-wilding is to establish an understanding of pre-human landscapes. It seeks to restore nature to its previously wild state in a progressive manner, by which nature and humans can live harmoniously. There is currently little consensus over the best way to 're-wild' and as such, it is important to experiment in order to fully understand the effects of re-wilding on vegetation. Knepp is 're-wilding' by using a variety of large grazing herbivores in three different experiments. This study compares all three experiments to one another in the hope that any differences can be explained using the vegetation data. No such study has been carried out at Knepp before, so the following objectives strives to fill in the gaps of knowledge and inform future decision making:

- Examine current vegetation biodiversity across the three blocks of Knepp Castle Estate.

- Assess the impact of grazing herbivores on vegetation biodiversity by comparing the three blocks of Knepp Castle Estate.

- Evaluate which block is most successful at achieving Knepp's re-wilding aims.

Firstly, I carried out vegetation surveys over a two week period in July 2015 in order to collect raw data, this included quadrat and line transect surveys as well as vegetation mapping of each site. Once data was collected, it was processed and statistically analysed and biodiversity indices were calculated along with the grazing intesnity of each block.

A summary of the key results are as follows:

- All three sites differ from one another in terms of vegetation biodiversity and structure.
- All blocks consists of a few species of dominant plants (high relative abundance).
- The southern block has the highest vegetation biodiversity.
- The middle block has the lowest vegetation biodiversity.
- The middle and northern blocks are dominants by grass (both species count and species abundance).
- Grazing intensity is highest in the middle block.
- Mean plant height is tallest in the southern block.
- There is a statistically strong relationship between grazing intensity and vegetation biodiversity in each block.

The findings from this study suggests that each experimental block at Knepp has a different level of vegetation biodiversity and that the Southern block, in particular, stands out. It has the highest vegetation biodiversity, the most extensive mosaic habitat development and has progressed furthest in the vegetation cycle. Along with this conclusion the additional literature highlights that a complex interaction of factors influence the vegetation biodiversity and habitat structure. These include grazing behaviour, species of grazer, stocking levels, pre restoration use, climate, other fauna and flora's ecological interactions. Nonetheless the variety of grazing herbivores and the grazing intensity is most likely to have had the biggest impact. This is a result of intermediate disturbances in particular areas of the block breaking up vegetation succession, which encourages the formation of mosaic habitats.

Furthermore, the results also reflect those of the intermediate disturbance hypothesis, which suggests that an optimal level of grazing intensity is preferential for vegetation biodiversity, as found in the southern block. The low levels of grazing intensity in the northern block and high levels of grazing intensity in the middle block both retarded vegetation succession and growth, therefore the biodiversity.

The species recorded in each block also reveal the stage of vegetation succession each block is currently under, this infers that the middle and northern blocks are not necessarily unsuccessful, but just at an earlier stage of vegetation succession. Most likely the variety of nine to four year fallow periods that fields received in the southern block provided a jump start into succession recovery and mosaic habitats, hence the southern block is currently the most successfully 're-wilded' area. Such evidence suggests the reason for the success in the southern block is due to the restoration processes most closely mirroring that of a natural grazing regime. The fallow period, variety of herbivore species and grazing intensity are the most suitable analogue for a landscape with little to no human intervention. The middle and northern block experiments are not as successful equivalents to pre-human landscapes. The high grazing intensity of the middle block, use of only longhorn cattle in the northern block, and application of herbicides in both blocks mimic a less natural landscape with more human intervention. Considering this, the low biodiversity scores for the middle block suggest that high grazing intensity has the most detrimental effect on vegetation biodiversity and landscape restoration.

Vegetation change occurs over a long period of time and Vera's wood pasture hypothesis imagines a vegetation succession over hundreds of years. With this in mind, propositions for the future of the project include further monitoring over time to see if the middle and northern blocks will follow similar patterns to the southern block. This will determine whether the middle and northern blocks are simply just a few steps behind the southern block in terms of vegetation succession. On the other hand it may reveal completely different habitat structures. In addition, although this study recommends that grazing intensity has the biggest impact on vegetation biodiversity, further experiments with fewer independent variables and more controlled sites would be recommended. Such a course of action would help to pinpoint whether grazing species, grazing stocking levels or land restoration management have the most significant impact on the vegetation biodiversity and the success of Knepp's re-wilding project.

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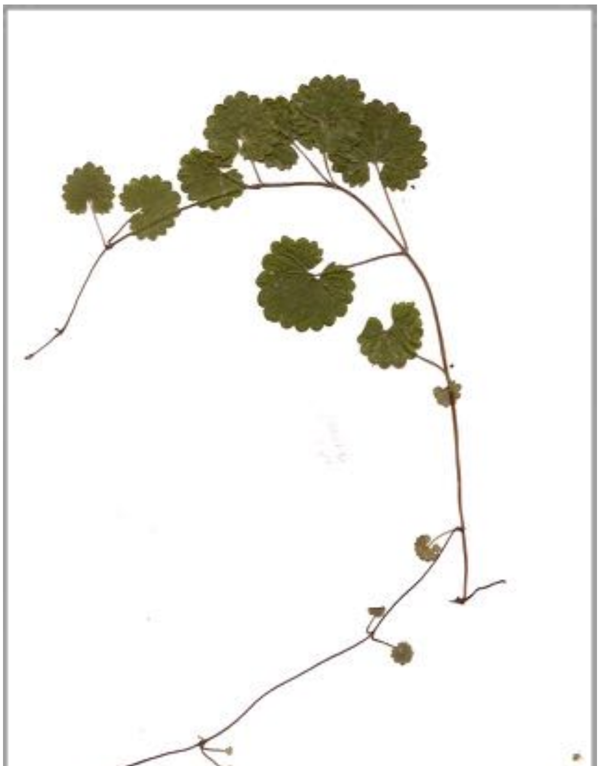
Centaurium erythraea


Glechoma hederacea


Dryopteris filix-mas

Appendix I - Herbarium

Quercus sp

Prunella vulgaris

Vicia tetrasperma

Galium verum

Myosotis arvensis

Lathyrus pratensis

Anagallis arvensis

Euphorbia peplus

Crespis

Hypericum perforatum

Cerastium fontanum

Centaurea nigra

Epilobium parviflorum

Jacobaea vulgaris

Cruciata laevipes

Rosa canina

Rannunculus

Calystegia sepium

Tripleurospermum inodorum

Geranium sp

Agrostis sp

Juncus sp

Phleum pratense & Bromus biebersteinii

Holcus lanatus

Argentina sp

Achillea millefolium

Sagina spp

Salix cinerea

Lolium perenne

Hordeum secalinum

Appendix II

The grass seed mix sown in the northern block consisted of:

Dactylis glomerata (Cocksfoot)

Festuca ovina (Sheep's fescue)

Poa pratensis (Smooth meadow grass)

Anthoxanthum odoratum (Sweet vernal grass)

Agrostis canina (Velvet bent)

Holcus lanatus (Yorkshire fog)

Agrostis capillaris (Common bent)

Agrostis stolonifera (Creeping bent)

Cynosurus cristatus (Crested dog's tail)

Phleum pratense (Large-leaved Timothy)

Festuca pratensis (Meadow fescue)

Festuca rubra (Red fescue)

The meadow mix sown in the middle block included the above, as well as:

Stachys officinalis (Betony)

*Centaurea nigra (*Black knapweed)

*Ranunculus bulbosus (*Bulbous buttercup)

*Lotus corniculatus (*Common bird's-foot trefoil)

*Lotus pedunculatus (*Greater bird's-foot trefoil)

*Hypochaeris radicata (*Cat's ear)

*Cerastium fontanum (*Common mouse ear)

*Rumex acetosa (*Common sorrel)

*Succisa pratensis (*Devil's-bit)

*Stellaria graminea (*Lesser stitchwort)

*Ranunculus acris (*Meadow buttercup)

*Lathyrus pratensis (*Meadow vetchling)

*Pilosella officinarum (*Mouse-ear hawkweed)

*Leucanthemum vulgare (*Ox-eye daisy)

*Plantago lanceolata (*Ribwort plantain)

*Trifolium pratense (*Red clover)

*Vicia cracca (*Tufted vetch)

*Achillea millefolium (*Yarrow)

*Rhinanthus minor (*Yellow rattle)

Appendix III

Function Group	Latin Name	Common Name
Forb	Vicia tetrasperma	Smooth vetch
	Ranunculus repens	Creeping buttercup
	Rumex crispus	Curly dock
	Pulicaria dysenterica	Fleabane
	Jacobaea vulgaris	Common Ragwort
	Trifolium repens	White clover
	Cirsium arvense	Creeping thistle
	Lathyrus pratensis	Meadow vetchling
	Taraxacum	Dandelion sp.
	Dipsacus fullonum	Wild teasel
	Prunella vulgaris	Selfheal
	Geranium dissectum	Cut-leaved Cranesbill
	Cirsium vulgare	Common thistle
	Centaurium erythraea	Common Centaury
	Cirsium Palustre	Marsh thistle
	Epilobium parviflorum	Hoary willowherb
	Galium verum	Lady's bedstraw
	Calystegia sepium	Hedge bindweed
	Ranunuculus acris	Tall buttercup
	Argentina anserina	Silverweed
	Sagina spp	Pearlwort
	Tripleurospermum inodorum	Scentless Mayweed
	Anagallis arvensis	Scarlet Pimpernel
	Glechoma hederacea	Ground Ivy
	Cerastium fontanum	Common Mouse-ear
	Veronica chamaedrys	Germander Speedwell
	Trifolium dubium	Lesser Trefoil
	Achillea millefolium	Yarrow
	Myosotis arvensis	Field Forget-me-not
	Euphorbia peplus	Petty spurge
	Cruciata laevipes	Crosswort
	Hypericum perforatum	St Johns wort
	Crepis spp	Hawksbeard
Shrub (<5m)	Rubus fruitcosus	Blackberry
	Prunus spinosa	Blackthorn
	Rosa canina	Dog rose
Tree (>5m)	Salix cinerea	Grey willow
	Quercus robur	English oak
Graminoid	Agrostis capillaris	Common bent
	Holcus mollis	Creeping soft grass
	Cynosurus cristatus	Crested dog's-tail
	Holcus lanatus	Yorkshire fog
	Anthoxanthum odoratum	Sweet vernal grass
	Bromus biebersteinii	Meadow brome
	Arrhenatherum elatius	Oat grass
	Hordeum secalinum	Meadow barley
	Lolium perenne	Perennial rye-grass
	Deschampsia cespitosa	Tufted hairgrass
	Agrostis stolonifera	Creeping bent
	Agrostis canina	Velvet bent
	Phalaris arundinacea	Reed canary grass
	Cyperaceae	Sedge
	Juncus	Rush

Appendix IV

ANOVA comparing species richness of each block

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	42.72222222	2	21.36111111	1.532982965	0.230871399	3.284917651
Within Groups	459.8333333	33	13.93434343			
Total	502.5555556	35				

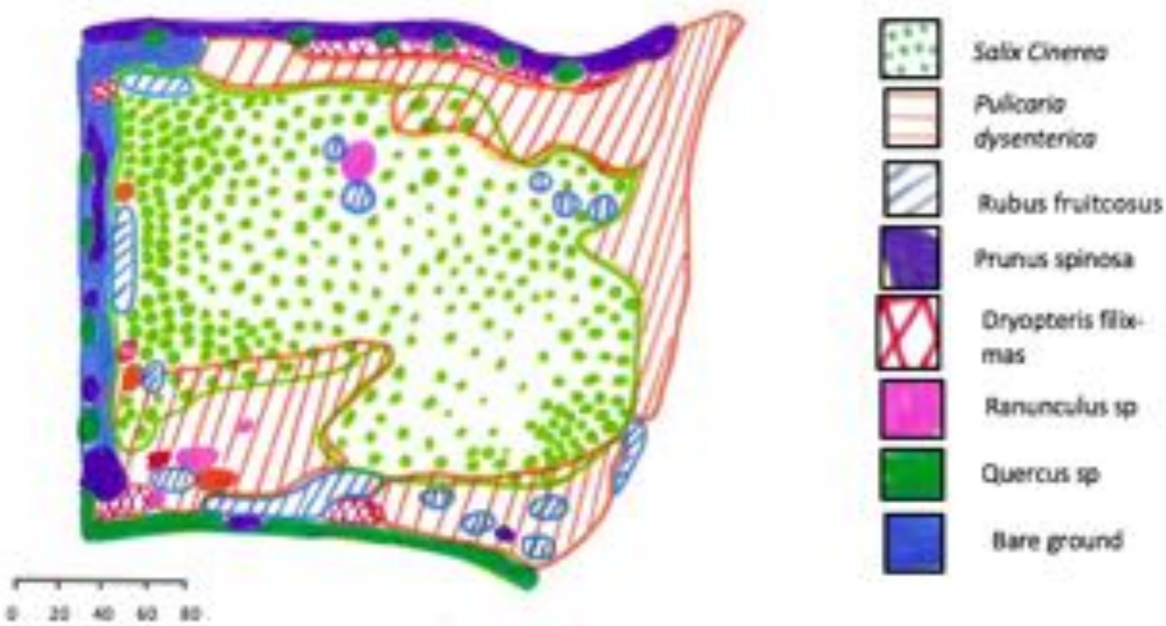
ANOVA comparing relative abundance of all species in each block

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.000834289	2	0.000417144	0.115885635	0.890874968	3.231726993
Within Groups	0.143984807	40	0.00359962			
Total	0.144819095	42				

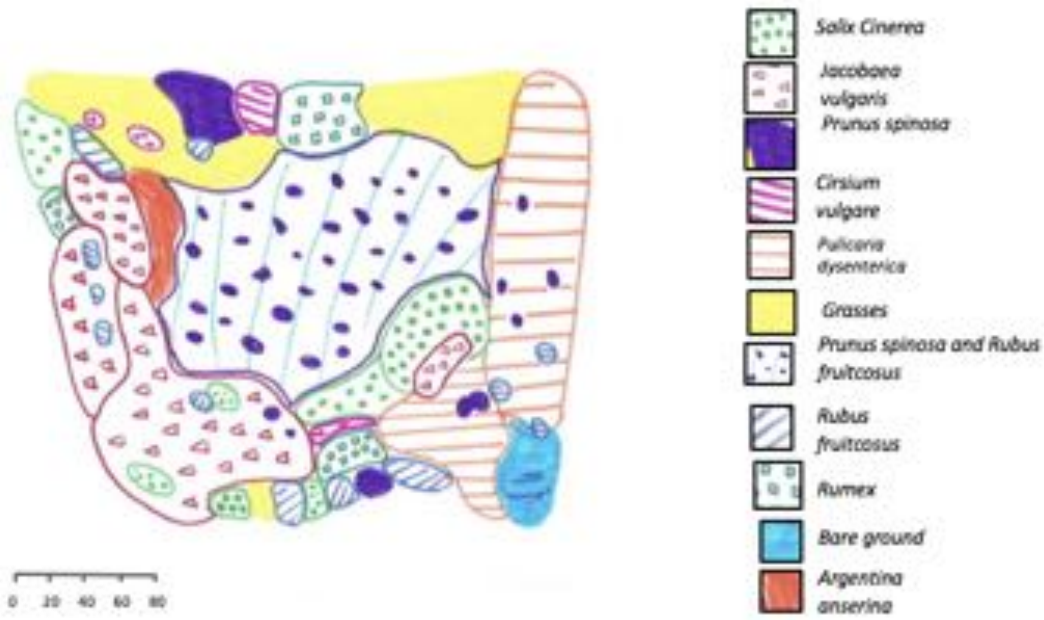
Appendix V
Vegetation maps

Site 1

Site 2



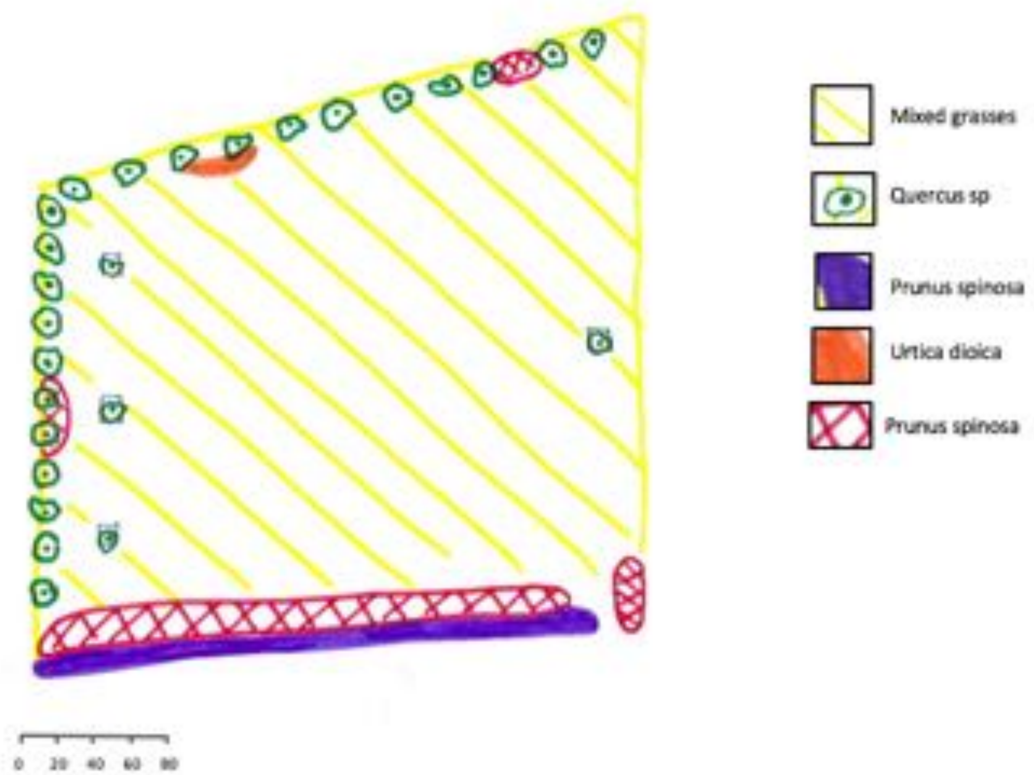
Site 3


Salix Cinerea
Jacobaea vulgaris
Prunus spinosa
Cirsium vulgare
Pulicaria dysenterica
Grasses
Prunus spinosa and Rubus fruitcosus
Rubus fruitcosus
Rumex
Bare ground
Argentina anserina
0 20 40 60 80

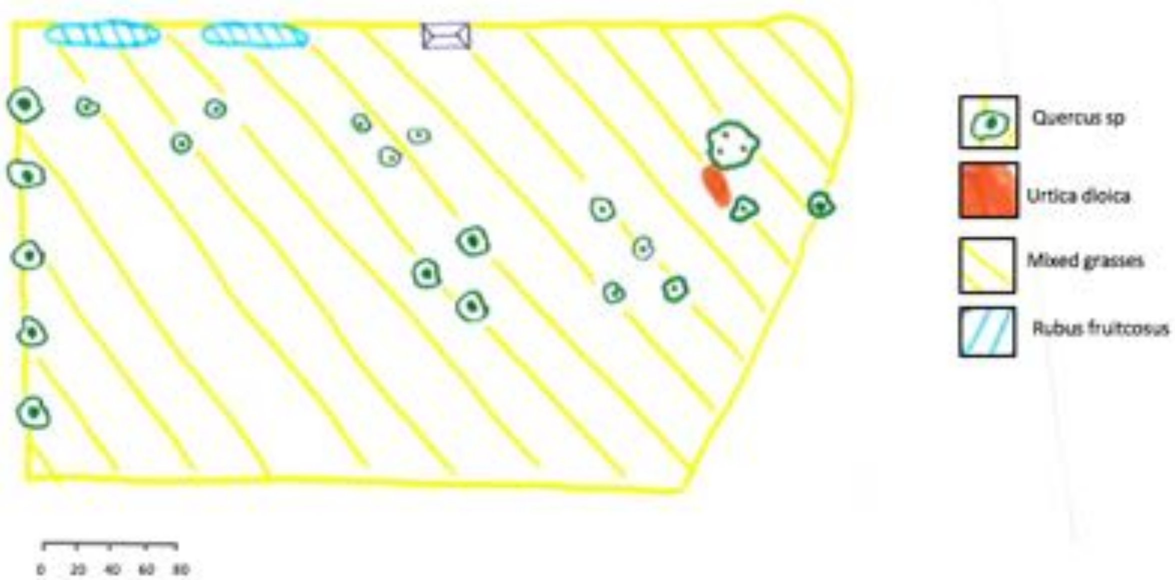
Site 4


Mixed grasses
Urtica dioica
Dryopteris filix-mas
Quercus sp
Rubus fruitcosus
Cirsium vulgare
footpath
0 20 40 60 80

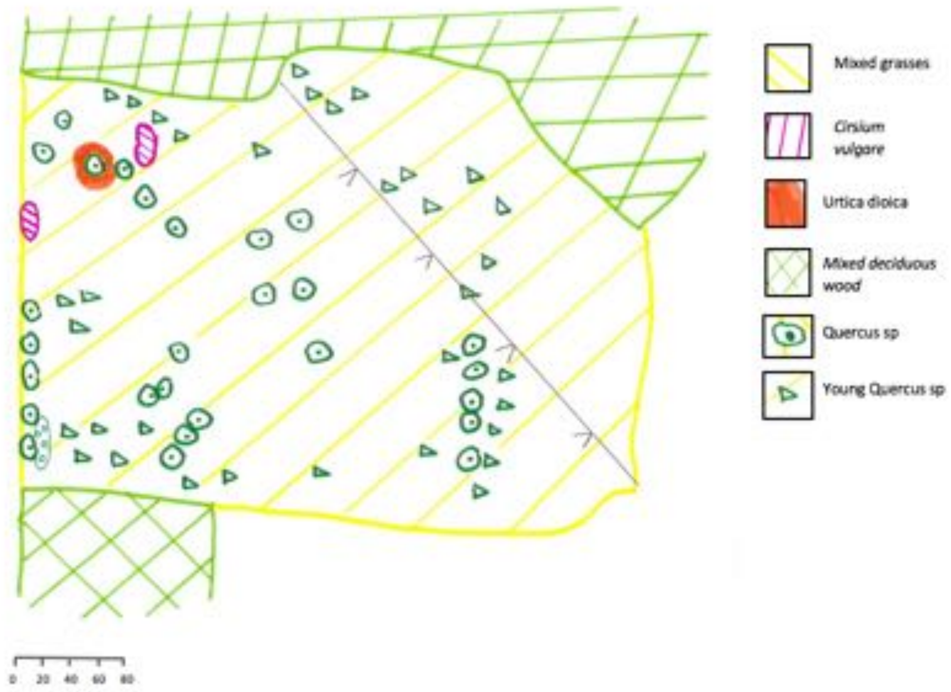
Site 5


Mixed grasses
Quercus sp
Prunus spinosa
Urtica dioica
Prunus spinosa
0 20 40 60 80

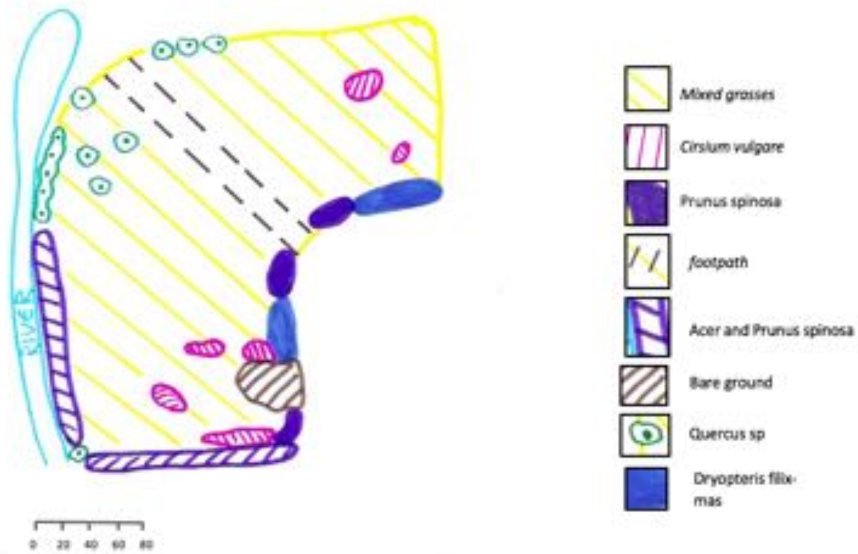
Site 6


Quercus sp
Urtica dioica
Mixed grasses
Rubus fruitcosus
0 20 40 60 80

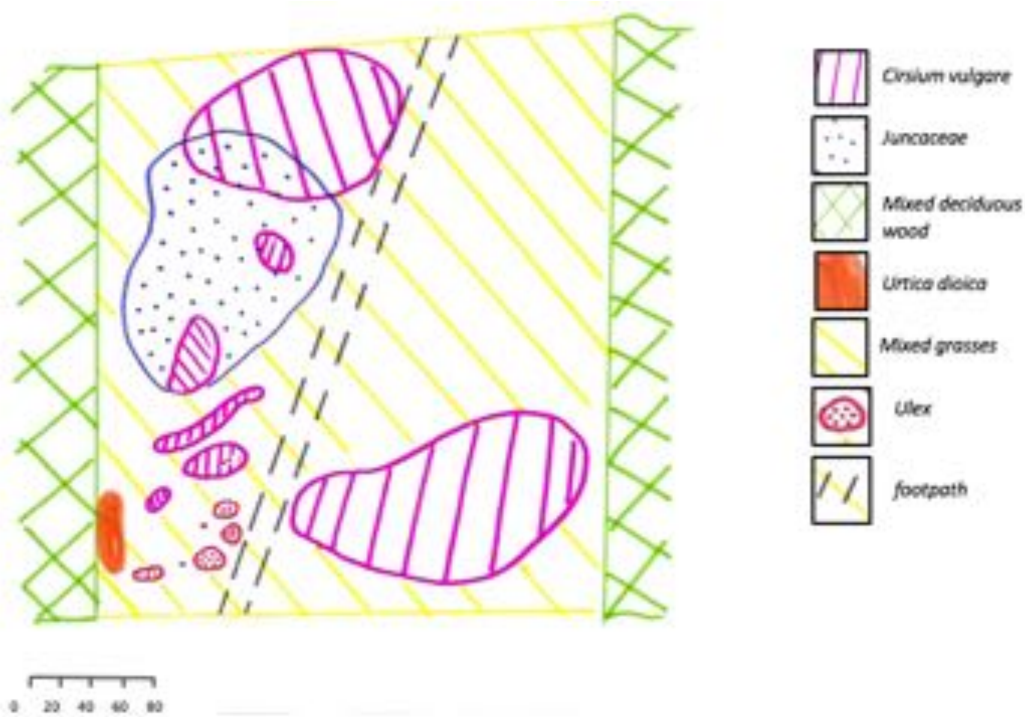
Site 7


Mixed grasses
Cirsium vulgare
Urtica dioica
Mixed deciduous wood
Quercus sp
Young Quercus sp
0 20 40 60 80

Site 8


RIVER
Mixed grasses
Cirsium vulgare
Prunus spinosa
footpath
Acer and Prunus spinosa
Bare ground
Quercus sp
Dryopteris filix-mas
0 20 40 60 80

Site 9


Cirsium vulgare
Juncaceae
Mixed deciduous wood
Urtica dioica
Mixed grasses
Ulex
footpath
0 20 40 60 80

Appendix VI

Quadrat and line transect coordinates

southern block

S1 Q1: 50.96672, -0.36621
S1 Q2: 50.96638, -0.36783
S1 Q3: 50.96563, -0.36340
S1 LT: 50.96746, -0.36674

S2 Q1: 50.97060, -0.38100
S2 Q2: 50.97057, -0.38135
S2 Q3: 50.97097, -0.37831
S2 LT: 50.97044, -0.37820

S3 Q1: 50.97707, -0.38432
S3 Q2: 50.97664, -0.38335
S3 Q3: 50.97715, -0.382080
S3 LT: 50.97600, -0.382100

middle block

S4 Q1: 50.98545, -0.36162
S4 Q2: 50.98579, -0.36314
S4 Q3: 50.98598, -0.36366
S4 LT 50.98622, -0.36310

S5 Q1: 50.98064, -0.35654
S5 Q2: 50.97995, -0.35764
S5 Q3: 50.97947, -0.35751
S5 LT: 50.97993, -0.35832

S6 Q1: 50.98749, -0.35333
S6 Q2: 50.98804, -0.35240
S6 Q3: 50.98747, -0.35128
S6 LT: 50.98747, -0.35043

northern block

S7 Q1: 50.99862, -0.35810
S7 Q2: 50.99978, -0.35602
S7 Q3: 50.99862, -0.35617
S7 LT: 50.99937, -0.35410

S8 Q1: 50.99742, -0.34870
S8 Q2: 50.99628, -0.34952
S8 Q3: 50.99679, -0.34744
S8 LT: 50.99683, -0.34870

S9 Q1: 51.00222, -0.34345
S9 Q2: 51.00130, -0.34538
S9 Q3: 51.00172, -0.34443
S9 LT: 51.00080, -0.34420

All quadrats and line transects mapped with stars (source: www.map.google.couk) (not to scale).