

# An investigation into grazing impacts by New Forest herbivores on inclosure and open woodland patch areas

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## Abstract

Several studies have researched the effects that ungulates can have on woodland ecosystems, however, limited applied research has been undertaken on the unique nature of the New Forest National Park. This research compares two small woodland parcels in the New Forest – one that herbivores are excluded from, and one that herbivores have access to. In both woodlands, data was collected on browse heights, plant species diversity, tree diameters, and light levels. Data was analysed using a multivariate Generalised Linear Models (GLM) to assess the significance of abiotic, ecological and impact factors. The results show that herbivores in the New Forest significantly impact woodland through browsing effects. The woodland that grazing herbivores had access to was shown to have significantly lower plant species diversity and abundance, demonstrating the influence of browsing effects over time. Such findings have implications for how nature reserves are managed using open and inclosure techniques across large national parks, where land is divided into parcels. Open areas may need to control for browsing depending on conservation aims. Further research to extend on this study should take into consideration seasonality.

# 1. Introduction

## *Herbivore Browsing*

Herbivores, and specifically large ungulates (Euungulata), can modify entire ecosystems (Sabo et al., 2017). This often occurs through direct effects such as browsing and trampling, but also indirectly through defecation and physical disturbances to habitats that influence plant dispersal (Boulanger et al., 2018). These combined direct and indirect effects from herbivory can subsequently affect plant community composition and even soil properties (Kolstad et al., 2018). Small-scale, localised disturbances such as soil trampling damage plant tissues, break shoots, uproot whole plants (Boulanger et al., 2018), and contribute to compaction of soils (Sabo et al., 2017). Browsing and trampling disturbances can also lead to thinner leaf-litter layers that in turn affects nutrient cycling (Ramirez et al., 2019).

Within forest areas, wild ungulates become major determinants of plant community composition, structure, dynamics, and succession, and therefore can be regarded in some terms as ecosystem engineers (Chollet et al., 2016; Ramirez et al., 2019). Selective herbivory from ungulates can influence both spatial and temporal vegetation heterogeneity (Lilleeng et al., 2016) and modify plant-plant interactions (Boulanger et al., 2018). The manner that ungulates selectively browse palatable plant species also has the potential to determine succession of forests when populations of ungulates reach significantly higher level, for example as they do in northern Europe (Kolstad et al., 2018). Larger population level effects can consequently influence plant-species turnover and long-term maintenance of biodiversity to create broader impacts that become important determinants of habitat, biodiversity, and the functioning of ecosystem services (Faison, 2015).

Such effects by ungulates have been reported mostly from a negative perception, especially regarding biodiversity (Vild et al., 2016). The pressure ungulates place on an ecosystem can significantly affect its stability, as well as reduce phylogenetic diversity by as much as 63%, causing significant levels of phylogenetic clustering (Begley-Miller et al., 2014; Lilleeng et al., 2016; Boulanger et al., 2018). Heavy browsing pressure slows down, or even ceases forest regeneration and thereby reduces plant diversity (De Vriendt et al., 2020b). Previous research has shown that ungulates can have strong deleterious effects on plant growth, reproduction, and survival, directly limiting plant abundance (Maron and Crone, 2006). It has also been suggested that grazing should be controlled to retain plant diversity in some areas (McIntyre et al., 2003). Overall, large ungulates are seen as biological disturbance agents that modify and shape the structure of ecosystems (Kolstad et al., 2019), often by creating simplified understory vegetation with decreased diversity (Simoncic et al., 2018).

From a positive perspective, some effects of ungulate pressure on forest diversity can be beneficial for several reasons. Firstly, suppression of tree regeneration and competitive plants provide opportunities for low-lying plants to generate improved survival (Vild et al., 2016). Some authors even argue that such herbivory can result in richer, more diverse assemblages of plants due to the decreased dominance of larger, more woody plants. Therefore, it can be perceived that ungulates are key regulators of processes in these forest ecosystems (Milne-Rostkowska et al., 2020). Notwithstanding, ungulates can reduce or

promote plant diversity depending on the intensity and selectivity of their herbivory (Nopp-Mayr et al., 2020).

From a species group perspective, Cervid ungulates, such as deer (Cervidae) and moose (Capreolinae), alongside applied forestry management, are principal drivers of vegetation structure and diversity where they occur in woodlands (Speed et al., 2014). Specifically, deer can drive biotic homogenization as well as forest composition by selectively browsing palatable plants (Rooney, 2009; Bernard et al., 2017). Because of this selective browsing, deer drive the assemblage of plant communities and can be considered a biotic filter (Begley-Miller et al., 2014). A further impact from deer is that they can significantly reduce the mass of leaf-litter (Chollet et al., 2020). Domestic ungulates (such as cattle, ponies, and donkeys) are frequently used to manage habitats, however there is limited research on the differences in their effects compared to that, or in conjunction with, that of wild ungulates.

### *Wild Ungulate Browsing*

Deer can also produce indirect effects such as influencing environmental conditions, and subsequently understorey communities (Sabo et al., 2017). These cervids can have natural and/or negative effects on woodland ecosystems with effects leading to biotic disturbances on the understories of forests (Faison et al., 2016a/b/c). High moose population densities have raised concerns about potential negative effects on ecosystem functioning and properties such as biological diversity and timber production (Kolstad et al., 2018). Research has shown that white-tailed deer (*Odocoileus virginianus*) can reduce plant diversity by favouring browse-tolerant plants (Begley-Miller et al., 2014). Furthermore, the presence of deer can also reduce the ability of decomposers to break down carbon by changing the bacterial to fungi ratio of the soil as a result of browsing induced changes in the plant community (Chollet et al., 2020). In the absence of predators, deer have the potential to greatly simplify ecosystems over time (Stockton et al., 2005) and their subsequent overabundance can have perceived negative effects on plant communities (Webster, 2016). Therefore, excluding herbivores, especially deer, from an area of land is often regarded as an effective way to encourage woodland regeneration (Moore and Crawley, 2014).

However, in some cases cervid herbivory can develop benefits to some ecosystems. For example, in boreal regions, forest floor richness may benefit from high intensity of red deer herbivory (Hegland et al., 2013). Also, where populations of deer are not overabundant, the forest ecosystem can benefit structurally. The favourability of deer towards brightly coloured flowers pollinated by animals can ensure their large seeds are dispersed by deer, increasing plant distribution and abundance (Bachand et al., 2015). Deer browsing can also moderate the development of shrub layers which can subsequently impact herbaceous layers because cervids will eat younger trees and dwarf shrubs in preference to ferns, forbs, bryophytes and older trees (Boulanger et al., 2018). Subsequently, bryophytes such as mosses and liverwort can benefit from the lack of competition. Such community composition knowledge can help inform forest management of planting procedures and ensure vegetation in deer territories remains unpalatable. Pioneer plant species have also been shown to benefit from frequent disturbances formed from deer-mediated seed dispersal (Vild et al., 2016). This corroborates with other findings that suggest decreased abundance of species such as *Acer* spp. and other palatable tree species where deer were

abundant, whereas unpalatable species or species resilient to browsing such as grasses (Poaceae) increased in abundance (Pellerin et al., 2010).

Other species that appear to be resilient to browsing impacts include Norway spruce (*Picea abies*), which can survive and has shown signs of re-growth, even when accessible to deer (Bernard et al., 2017). Browsing by deer has also shown to cause a 12% reduction in species diversity and a 17% reduction in species richness, with graminoids being the least vulnerable to their browsing impact (Begley-Miller et al., 2014).

### *Forestry Management*

Due to the impact that ungulates are known to have on ecosystems, whether positive, or negative, it is apparent that there are huge implications for their conservation management as well as for applied forestry globally. For forest managers, increasing ungulate populations is proving to be a large issue since damages ungulates cause to trees and seedlings can jeopardize intended forest regeneration processes (Boulanger et al., 2018). Therefore, controlling large herbivore populations can offer potentially significant conservation benefits to forest understorey plant communities (Chollet et al., 2016). Management by simple culling of deer has been found to increase richness and abundance of native plant communities, while introduced plants subsequently decrease (Chollet et al., 2016). This emphasises the importance of regulation of ungulate numbers. However, pairing wildlife management alongside silviculture decisions can make for viable decisions for forestry in regions where plantations grow browse-resistant trees (De Vriendt et al., 2020a).

In the UK, national parks often have mixed forestry management applied. The New Forest in the south is unique in that it is subject to free roaming domestic grazing stock including New Forest ponies, cattle, and donkeys, as well as pigs during the pannage season (New Forest Living, 2017). This is in addition to browsing from wild ungulates including three cervid species. The owned livestock grazers are restricted to open 'forest', which consists mostly of heathland, marshland, and woodland (New Forest National Park, 2015). Gates and cattle grids prevent these animals from accessing busy roads and entering specific land parcels (New Forest – Explorers Guide, 2013) that are home to both deciduous and coniferous trees (Forestry England, 2019). The New Forest has 29% deciduous broadleaved trees and 17% coniferous woodland, with the remainder being a mix of bogs, marshes, scrub, and heath (Iqbal et al., 2013).

### *New Forest Inclosures*

Areas of the New Forest have been repeatedly inclosed and reopened since the 1300's. This practice originated as a method to allow forest regeneration for subsequent harvesting for building materials, through the exclusion of grazing animals from the inclosure (Tubbs, 2017). This practice continues to this day in order to preserve and protect the wildlife, habitats, and natural resources within them (Tubbs, 2017). Inclosed areas are managed by relevant statutory authorities and local landowners to ensure their long-term sustainability and can restrict access by livestock, vehicles or humans (Newton, 2010). They may also include measures to control the spread of invasive species and to encourage growth of native vegetation.

## *Research Direction*

This research aimed to investigate the impact of herbivory in the New Forest by domestic livestock, compared to that of herbivory by cervids. This was achieved by surveying a variety of floral characteristics in inclosures with only cervid herbivory permitted, and open woodland with herbivory by both cervids and domestic livestock. To the best of our knowledge no extensive quantitative research has been conducted in the New Forest to assess impacts of herbivory, and because it is a unique study site in terms of its resident ungulates, it is a novel arena to study such relationships. The relevance of this research aims to help inform forest management practices as well as conservation efforts about the true impact of herbivory – whether positive or negative.

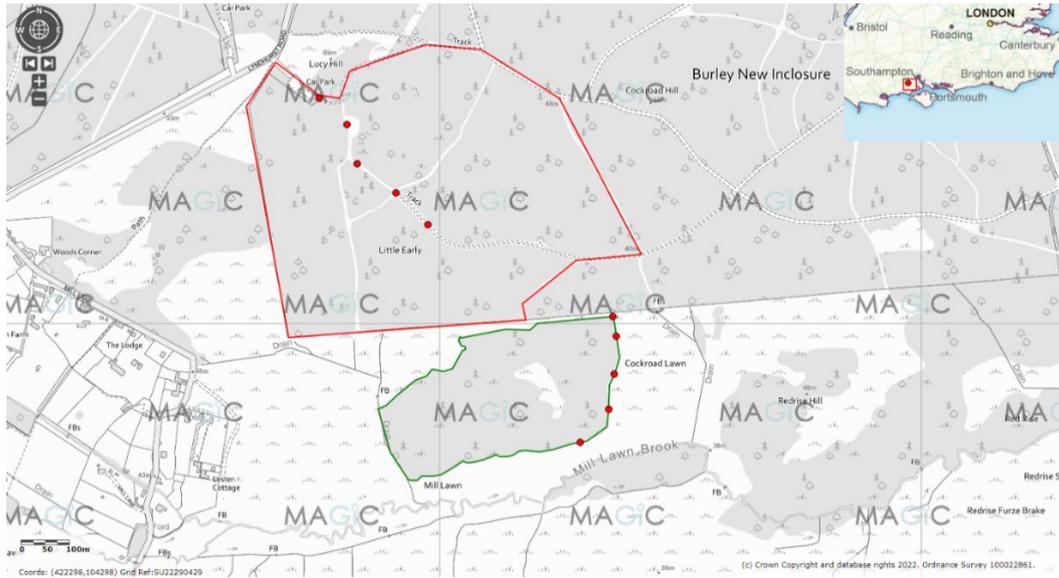


**Figure 1:** New Forest Pony shown on the edge of Area B. Browse line is evident.

## 2. Method

### *Study Site*

The chosen study site consisted of two areas of woodland: Area A (red outline in Figure 2) and Area B (green outline Figure 2). The two areas were chosen due to their proximity to each other and similarities in underlying geology, therefore, the natural floral communities of the two areas were considered to be similar. Area A, known as Lucy Hill Inclosure, is an area where cattle and New Forest ponies are excluded, whereas Area B, known as Mill Lawn is an area where cattle and New Forest ponies roam freely. Three cervid species are known to graze in both areas. The red points marked on Figure 2 show locations where data was collected. These areas of woodland sit in the village of Burley, New Forest, and are part of the wider Burley New Inclosure. There are records of the Burley New enclosure existing as



**Figure 2:** Location of the study within the New Forest National Park - red dots indicate locations of sample sites, red outline delineates Area A, green outline delineates Area B.

far back as 1310 in Anglosaxon times. This enclosure seems to have been reopened to grazing from 1570 until 1810 when it was inclosed under the 1808 enclosure act. It was subsequently open again from 1851 until the end of WW1 when it was enclosed once more. The Lucy Hill Inclosure (Area A) is part of that enclosure which still exists in the present day.

The following research questions were applied to understand differences between wild grazed and wild/domestic grazed areas;

1. How does vegetation compare between Area A and Area B?
2. How does diameter breast height (tree girth) compare between Area A and Area B?
3. How does browse line height compare between Area A and Area B?
4. How do light levels compare between Area A and Area B?

Data collection was conducted during a single site visit to both areas in November 2021. The data collection method is described in Table 1 and was applied to both areas (Figure 3).

Data Collection	Research Question	Description
<p><b>Sampling Strategy</b> In both Area A and Area B, a systematic sampling method was used for standardisation. A 500 m stretch of path in both areas was chosen before data collection. Along both 500 m stretches, a 30 m transect was placed every 100 m in Area A and every 50 m in Area B. Data was collected every 5 m along each transect.</p>	<p>1</p> <p>2</p> <p>3</p> <p>4</p>	<p>At each 5 m interval along the transect, plant species within 1 m radius were recorded.</p> <p>At each 5 m interval along the transect, the nearest tree(s) within a 2 m radius was measured; diameter breast height (dbh).</p> <p>At each 5 m interval along the transect, the nearest] tree (s) within a 2 m radius was measured for its browse line height.</p> <p>At each 5 m along the transect, one sample of light level was recorded.</p>
<p><b>Measurement Technique</b> A clinometer was used to map out the 500 m stretch in both areas and a 30 m tape measure to map out the 30 m transects.</p>	<p>1</p> <p>2</p> <p>3</p> <p>4</p>	<p>Plants were identified using standard identification keys and cross referenced with iNaturalist app.</p> <p>Using breast-height as a constant, diameter was measured using a tape measure.</p> <p>Using the lowest hanging branch from the tree, browse line height was measured using a tape measure.</p> <p>A lux meter was used to measure light levels.</p>

**Table 1:** Data collection methodology for Areas A and B in the New Forest.



**Figure 3:** Measuring browse heights in Area A.

## Data Analysis

Summary data was visualised graphically for research question 1. For questions 2, 3 and 4 multivariate Generalised Linear Modelling (GLM) was deployed to contrast mean responses of Species or abiotic variables dbh, Browsing, and Lux, with Area, Transect and Metres as predictor factors. Data were placed into a multivariate matrix with responses as columns combined with separate predictors. Three multivariate GLM models were constructed with separate family distributions for contrast. For all models, the function ‘manyglm’ in package ‘mvabund’ was used to assess relationships (Wang et al., 2012). The independent models were written in the style of:

$$\text{Resp.MV} \sim \text{Pred\$Area} + \text{Pred\$Transect} + \text{Pred\$Metres}...$$

...reflecting the species or abiotic values as a response matrix (Resp.MV) and summoning predictor variables. The ‘manyglm’ models were fitted using Poisson and Binomial family regressions and used block resampling of rows for multivariate inference. The models were fitted using a log-linear model, with the mean model as:

$$\log(\mu_{ij}) = \dots Y_{ij} \sim NB(\mu_{ij}, \Phi_j)$$

The model for the abiotic variables for each survey session  $i$  ( $Y_{ij}$ ) was negative binomial (NB) as written as  $\text{Resp.MV} \sim \text{Pred\$Area} + \text{Pred\$Transect}...$  The overdispersion parameter  $\Phi_j$  was constant across sample sessions but can vary across species/abiotic factors, and the mean of  $Y_{ij}$  is  $\mu_{ij}$ , a log-linear function of block and treatment.

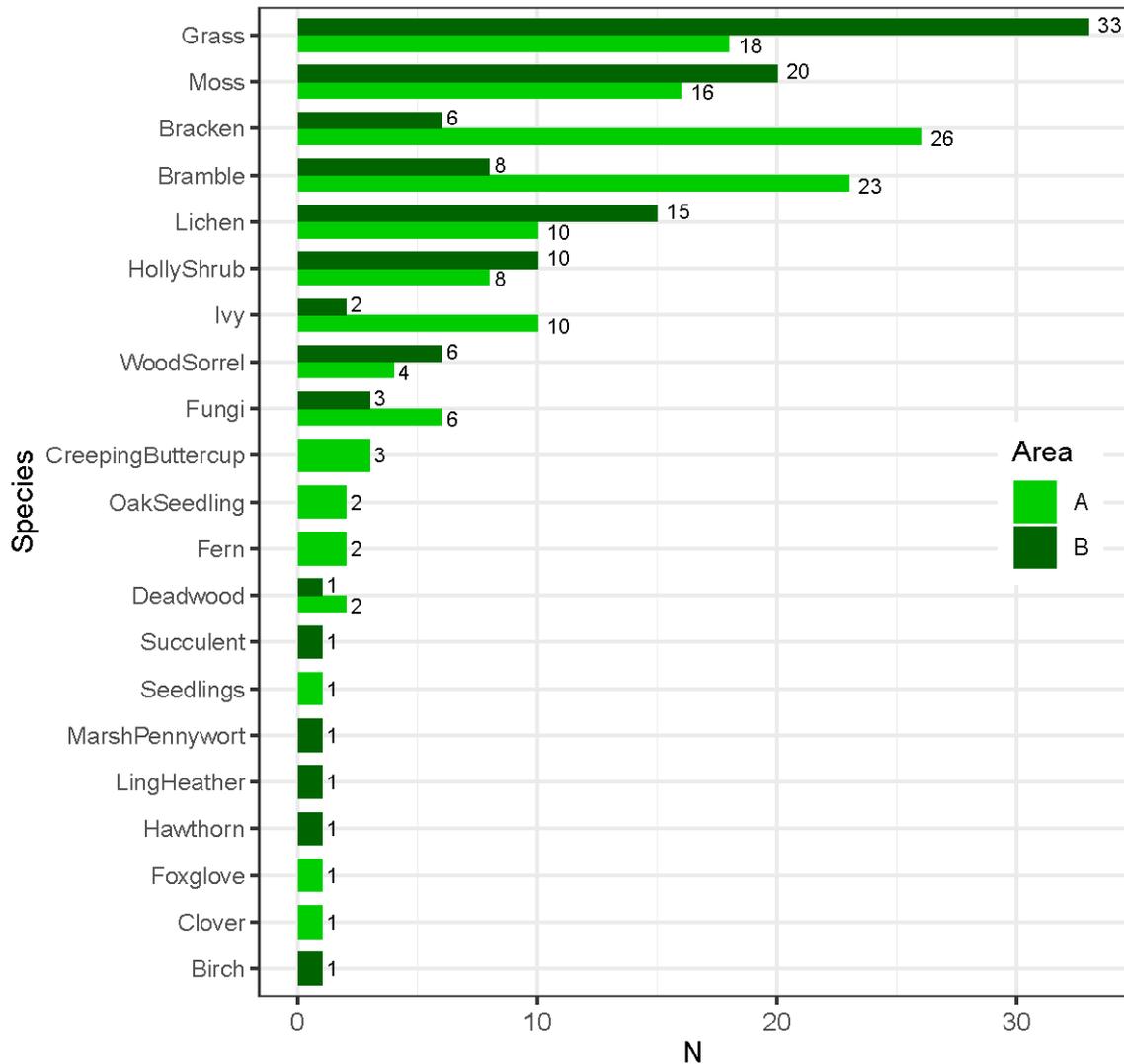
Model fit was confirmed by inspection of Dunn Smyth residual-fits and Q-Q-plots (Dunn and Smyth, 1986). AIC values were assessed to inspect differences between Poisson and negative binomial families. Significance summary tables were derived using multiple univariate ANOVA tests from model subsets to contrast significance. Resampling of ANOVA testing was performed using Monte-Carlo permutation bootstrapped to 999 iterations and adjusted for model correlation. Coefficients were inspected where significant for multivariate model results, and a colour plot of significant coefficients used to help contrast more intricate response/predictor relations. Analyses were conducted in R and R-Studio (R Core Team, 2022) with matrix handling utilising package reshape2 (Wickham, 2007), and plots produced in ggplot2 (Wickham, 2016).

## Ethics and Risk

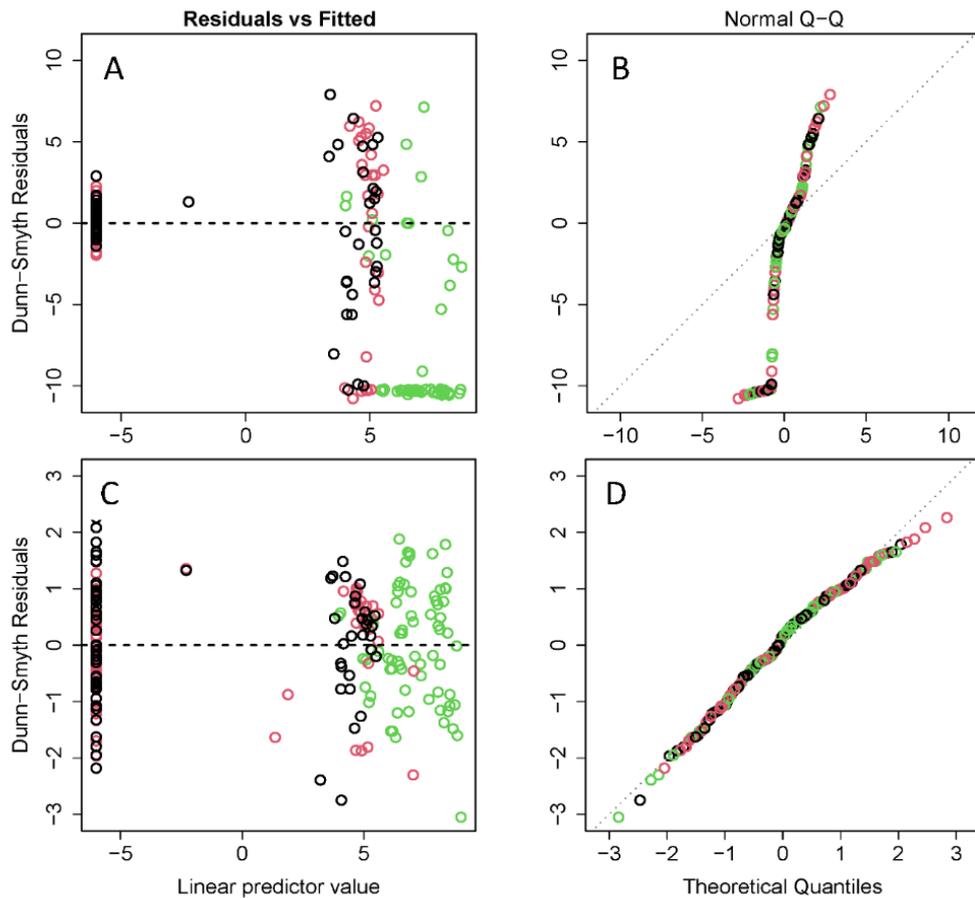
Ethics approval for the project was secured via the UWE student project consent process. In addition, we sought permissions from Natural England and performed site risk assessment. No ethical issues were determined as the study was non-invasive, purely observational, and measurement driven.

### 3. Results

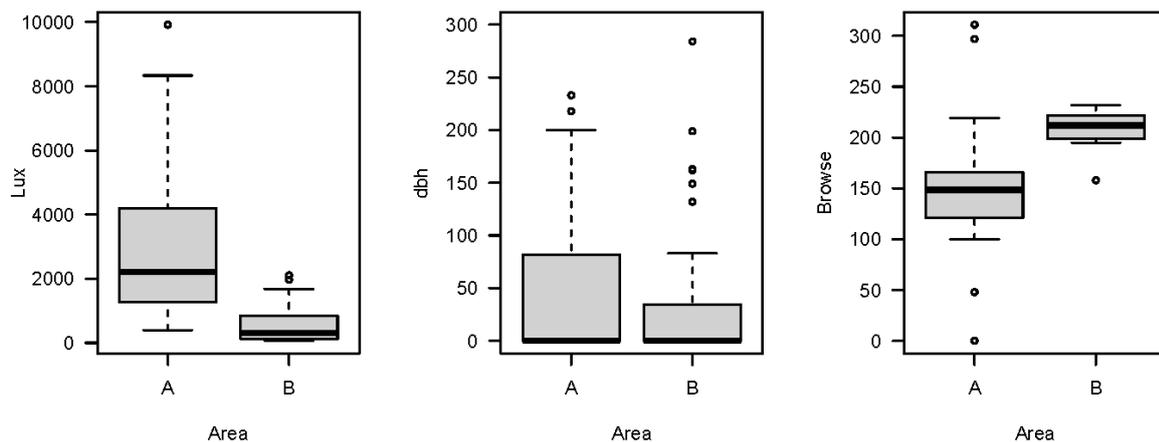
A total of 133 sample trees were recorded for Area A and 109 for Area B respectfully. Dominant species in both areas were grasses, bramble (*Rubus fruticosus*), lichen, bracken (*Pteridium spp.*), and moss (*Sphagnum spp.*). Area A had the highest abundance of bracken (26), bramble (23) and grass (18), whereas Area B had the highest counts of grass (33), moss (20), and lichen (15). There are clear differences in plant species composition between Areas A and B (Figure 4).



**Figure 4:** Plant species group abundance + Deadwood, for Area A (light green) and Area B (dark green).



**Figure 5:** Dunn-Smyth residual fits plots for a model fit. A/B = Poisson model demonstrating poorer fit. C/D = negative binomial model demonstrating superior model fit.



**Figure 6:** Boxplots of abiotic and physiological measured variables for Area A and Area B.

The best model fits were achieved with negative binomial family distributions (glm.POIS – df -1018, AIC -985.9; glm.NEGB df -167, AIC -12.9), so these models were selected for analysis (Figure 5). Mean differences between abiotic (Lux) and physiological (dbh + Browse height) variables are presented in Figure 6. Lux levels were significantly higher in Area A compared to Area B. Area A had generally larger dbh than Area B. Browse height was higher in Area B.

	Area	Species	Transect	Metres
dbh (mean)	0.687	0.001	0.727	0.842
Browse ht (mean)	0.57	0.001	0.774	0.984
Lux (mean)	0.001	0.006	0.001	0.074

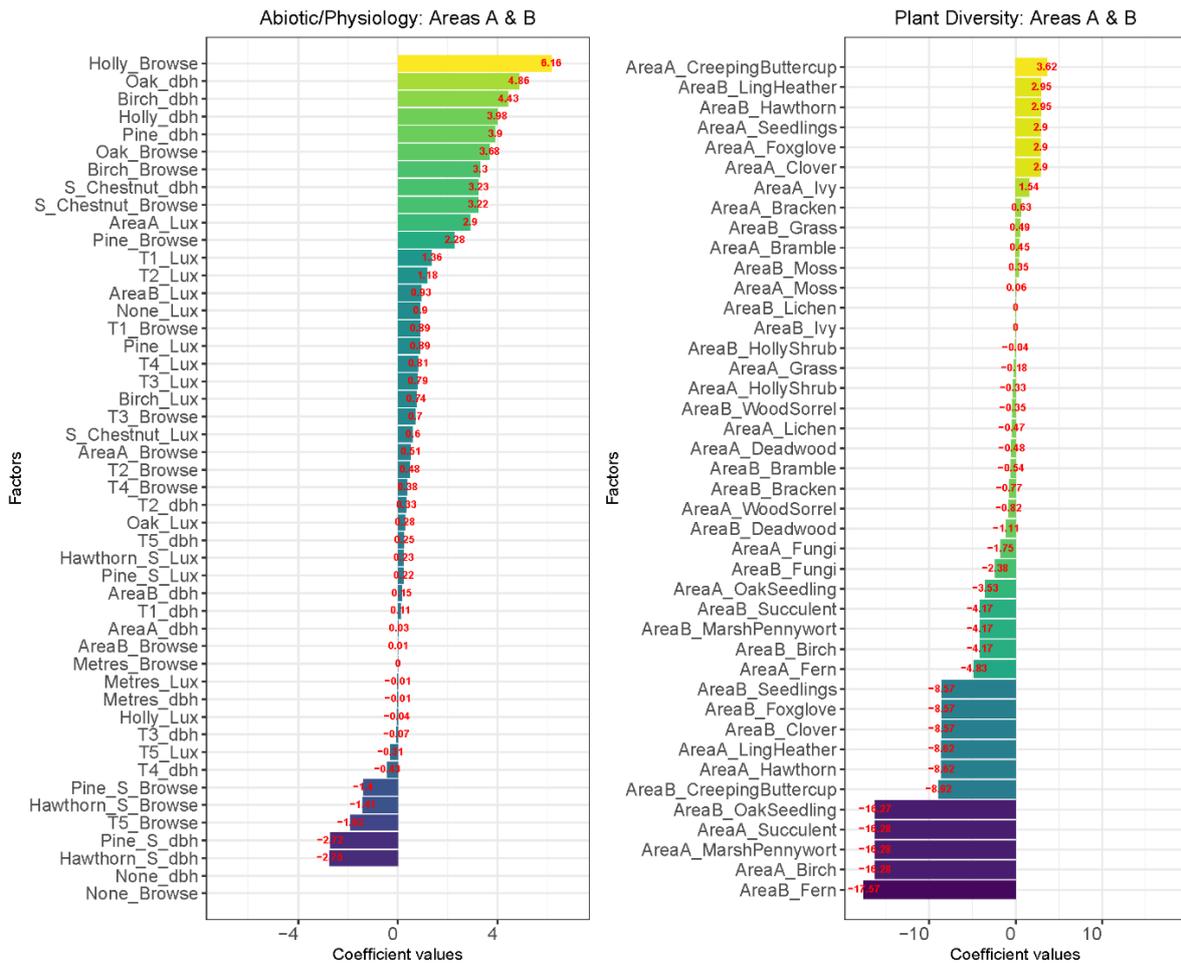
**Table 2:** ANOVA p-value plot matrices for negative binomial GLM. Significant responses are highlighted in yellow ( $p < 0.05$ ) and red ( $p < 0.01$ ).

ANOVA p-value plot matrices for the negative binomial GLM were successfully completed (Table 2). Each response value with a p-value colour is a significant result in the model with a p-value  $< 0.05$ . The range 0.001-0.08 is given as acceptable as these values often break below the 0.05 threshold with higher resampling. Resampling at 999 iterations was sufficient for the purposes of eliciting ANOVA hypotheses and took 15 seconds to cycle.

Coefficient caterpillar plots for Model 2 (glm.NEGB) negative binomial are presented regardless of their significance (Figure 7). Therefore, the relevant significant ANOVA values from Table 2 should identify significant factors first, and then values corresponding to a significant variable searched for on the coefficient plot to reveal a positive (yellow/green) or negative (blue) coefficient value with each predictor. Numerical predictors induce an inverted coefficient and therefore present as increased negative coefficient with increasing predictor value.

For the Abiotic/Physiological model, numerical predictors induced inverted coefficients and therefore present as increased negative coefficient with increasing predictor value. Area A had clearly modelled larger Lux and Browse height coefficient, suggesting that Area A is more shaded and has a lower browsing line. The coefficient response for dbh expressed a smaller value than Area B, suggesting that Area A has a larger mean dbh. However, the dbh coefficient signal was somewhat minor so results should be considered alongside raw data boxplots. Holly (*Ilex aquifolium*) was a species that responded strongly in coefficient to Browse height, suggesting that it is a species with a very low browse line height, showing with clarity that it is avoided by grazers. Hawthorn (*Crataegus monogyna*) and pine (*Pinus* spp.) responded negatively with browse line and other Abiotic/Physiological responses suggesting these are species are grazed more often. Contrastingly, oak (*Quercus robur*), birch (*Betula pendula*), and chestnut (*Castanea sativa*) expressed positive browsing coefficients determining that they had a lower browsing line.

For the Plant Diversity model, numerical predictors induced standard coefficients and therefore present as increased positive coefficient with increasing presence of a species. Coefficients expressed which species were positively dominant in each area (yellow/green +0 coefficients) and which were negatively recessive or absent in a particular area (dark blue -0 coefficients). Bracken, bramble, grasses, and moss were prevalent in both areas with Area B having higher abundance of holly and grass, and lower levels of bracken, ivy (*Hedera helix*), and foxglove (*Digitalis purpurea*) (Figure 4). The diversity coefficient plot (Figure 7) suggests Area A has higher diversity than B, and that model-based coefficients align with differences observed in abundance between the areas as suggested by Figure 4.



**Figure 7:** Coefficient plots of Abiotic/Physiological (A) variables and Plant Abundance/Diversity (B) models. Extrapolated coefficient values for plot A (left) are inverted, with negatively expressed signals corresponding to increased numerical value of a predictor factor. For plot B (right) response factors express positively with increasing abundance in the model (yellow/green +0 coefficients) and negatively recessive (dark blue -0 coefficients) with decreasing or contrastingly low abundance. Those near the zero-crossing point are either equal in both areas, absent, weakly expressed, or may have had insufficient data to express coefficients strength in the model.

## 4. Discussion

### Comparing 'Inclosures' (A) and 'Open' (B) Woodlands

When comparing plant species abundance, it is apparent there was a visible difference between Area A and B. Area A exhibits sizeably higher abundance of plants such as bracken and bramble. Perhaps these ground dwelling species are more apparent in Area A than B since the effects of trampling are lessened in the inclosure, whereas Area B has experienced higher levels of trampling and defecation. This basic and modelled scenario is supported by

literature, where evidence suggests large herbivores can modify entire ecosystems by direct effects such as browsing, trampling, and defecation (Kolstad et al., 2018). Furthermore, Figure 4 has illustrated higher plant abundance in Area A than B. Reasons contributing to this are that Area A is subject to less browsing pressure, and therefore vegetation is allowed to prosper considerably more. Literature supports this basic notion where ungulates' feeding behaviour has been shown to result in lower abundance of some species groups (Simoncic et al., 2018). Differences in bracken and bramble were also apparent across areas and these aligned with differences in deadwood whereby 50% more deadwood was recorded in Area A than B. This was mirrored by the finding that there were also 50% more fungi recorded in Area A than B. This loosely suggests that ungulate presence and herbivory may play a part in altering decomposition rates. Research supports this notion evidencing that litter decomposition is significantly faster in an exclusion than in a control site with excessive permitted deer herbivory (Kasahara et al., 2016).

Contrasting tree diameter at breast height (dbh), browse line heights (Browse) and light levels (Lux) between 'Inclosed' Woodland (Area A) and the 'Open' Woodland (Area B) revealed expected differences that also made realistic sense, and confirmed the model's accuracy. Tree diameter (dbh) was higher in Area A compared with B as shown in Figure 5 suggesting that trees have a greater chance to grow and regenerate in a woodland that is excluded from ungulates. Research on moose removal by fencing exclusion also resulted in greater heights and basal diameters than in control subplots (McLaren et al., 2009). We might expect Area A to have increased dbh levels from inclosure, however, results may simply relate to how saplings are browsed by ungulates, perhaps more so than adult trees, and the raw data herein may have gathered more recordings of saplings in Area A that have smaller dbh. This notion is supported by research from Sabo et al. (2017) which also found that excluding heavy ungulate grazing from plots increased sapling abundance.

Browse heights also differed between the two woodlands. This is recognisable on the basis that ungulates can drive forest composition by browsing selectively palatable plants (Bernard et al., 2017). However, mean browse values showed that browse heights were higher in Area A, but this may be because the trees were generally older and taller, with higher bow branches. This is supported by research from Speed et al. (2015) which revealed that height growths of rowan trees were constrained when browsed by ungulates. This supports findings herein in showing that any higher browse heights in Area A are more likely a result of simple lack of browsing due to those trees being taller with higher branches, and the lower browsing height in Area B being more likely due to intensified browsing that keeps tree height lower.

It could be expected given the above dbh and browse line height that light levels would be significantly brighter in Area B than A. This is based on the rationale that more light would be allowed to penetrate to the forest floor in Area B due to less vegetation from browsing. Research evidence supports this notion, with findings that ungulates control the shrub layer and indirectly increase herbaceous plant species richness by increasing light to reach the ground layers, and that species richness and cover decreases under browsing pressure, thereby increasing light penetration (Boulanger et al., 2018). Results herein confirm this effect (Figure 7).

### *Using model-based GLM's*

The use of GLM frameworks for ecological questioning is established and continues to be used successfully for multivariate datasets (Zuur et al., 2013; Freire et al. 2020; Lewis et al., 2021; Rice et al., 2022; Jenner and Lewis, 2023). Dunn-Smyth residual fit plots showed clearly the limitations of a Poisson family applied to a GLM demonstrating poorer fit of both quantile plot and residual-fits. Negative binomial family distribution has had a proven better fit for count data across many datasets and is a more reliable source of interpretation when ecological data sets are frequented by zeros across data matrices (Warton et al., 2016). Akaike Information Criterion (AIC) helped discern which model was the best fit in addition to plot fit inspection prior to ANOVA permutation and coefficient interpretation.

Transects were included in the model term to help express differences and add realistic parameter differences between areas in a finer context into the model. Transect 5 had low Lux and Browse height, but higher dbh than other transects. There is no obvious suggestion for this, and we conclude this part of the model is displaying useful levels of heterogeneity in sampling. In terms of the key findings from the GLM model, it is apparent that Area A is more ecologically productive. There is a higher abundance and diversity of plant species in Area A compared with Area B, which signals the importance of excluding ungulates from some woodlands to promote ecological heterogeneity, should that be the goal of an Area's conservation programme. Coefficient values in such GLMs can follow an association with higher abundance or detection for a specific species, yet they can often reveal more subtle predictor/response relations that are ecologically correct when raw data is too sparse for univariate analysis. The most dominant species observed herein often had the greatest coefficient size, or strongest negative signal, for their respective areas.

The model-based results herein suggest that conservation efforts aiming to preserve botanical diversity may be compromised by open grazing. As a result, some methods of intensive grazing from large ungulates local to the New Forest, may need to be restricted to retain a higher botanical diversity and a more natural structure to the forest as part of a conservation action plan. Research in this genre offers some pointers. McIntyre et al. (2003) found that for land managers to retain plant diversity on grazed landscapes, it is desirable to provide grazing pressure that spans across a variety of intensities. This seems an unlikely outcome for management plans in the New Forest, as grazing is either permitted, or not, across land parcels. However, there is somewhat a lack of clarity in such a regime as deer are free to graze on any land parcel with access, including the inclosures, in areas where they are not halted by cattle grids. This perhaps makes modelling of a larger area of the New Forest, and advice to more complex management prescriptions, inevitably more complex to offer or achieve.

### *Grazing Impacts*

Putman's (1986) study is widely considered as one of the best analyses of overgrazing in the New Forest. It was based on 10 years of research on impacts of the behaviour of large herbivores and estimated the carrying capacity for ponies by relating forage offtake to forage productivity. However, the values provided should be viewed cautiously due to uncertainties in estimating forage availability, animal densities, and vegetation productivity.

The study focuses on the capacity of the New Forest vegetation to support stock rather than the desirable stocking density for biodiversity. To the best of our knowledge, there has been limited study since, to evolve Putman's findings, and our model-based approaches may provide a useful framework method for doing so, which invites further study.

When the new forest enclosures were first set up, they had no ground compaction, which provided an advantage for some plant groups to succeed, and ultimately affected botanical diversity (Newton, 2010; Tubbs, 2017). Moth hunters, among other ecologists, were often attracted to inclosures to search for specimens on host plants such as bramble that was scarce elsewhere due to grazing. Inclosures and their management therefore make for a useful proxy when studying disturbance across the New Forest. In more recent years, overgrazing has been linked to declines in butterflies (Lepidoptera), Orthopterans (Orthoptera), and other insects (Newton, 2010) and work continues to try to understand such long-term population dynamics. However, standardisation of such monitoring is always a challenge due to the variability of site conditions inside, and out of, inclosures and subsequent variable grazing pressure (Newton, 2010). The removal of logs (deadwood) for firewood is a further impact unaccounted for in this study, is difficult to quantify, and also invites further study.

## 5. Conclusions and Limitations

This study aimed to investigate basic impacts of herbivory on two small land parcels in the New Forest National Park's woodlands. It revealed that browsing has impacts on plant species diversity and abundance, light levels, browse height, and tree diameter, in areas where ungulates have access to grazing. Significant differences were found in vegetation structure and diversity between Area A and Area B. Tree diameter at breast height (dbh) was higher for Area A, yet only marginally. Browse height was higher for Area B suggesting an impact by grazing ungulates. Light levels were lower in Area B. Modelled results developed a clearer picture through coefficients than raw data observation and it could be suggested that model-based analysis may serve such analysis better in future ecological studies. It could also be suggested, as within literature, that browsing from the New Forest's ungulates simplifies the overall ecosystem, creating woodlands with lower diversity and abundance of plant species. We do not believe that browsing and other herbivory effects are necessarily a problem for forest management, given they are now natural processes established for many years. There are ecological benefits and drawbacks to inclosures and open area grazing which invite further simple monitoring over time to reveal outcomes.

A limitation regarding data collection in this study was time restriction for data collection over a single day. This may have resulted in the lower light levels for Area B. A more robust study should consider full seasonality by collecting data over a whole phenological season to detect fuller plant species diversity and incorporate more time and opportunity for browsing risk to be expressed in the data modelled. Observer bias was controlled by having a single surveyor. However, group experience of the area may improve visual detection of species on site. A further limitation of this study is its' single timeframe picture. Over a longer term, land-use change may occur and therefore such study may be better placed in a time series analysis.

Overall, it is evident that there are notable differences in woodland parcels when they differ in exposure to herbivory. This research finding has uses for forest management strategies, experimentation for model-based analysis, and conservation planning.

## 6. Supplemental Material

Data and R-code can be accessed open source at; <https://doi.org/10.5281/zenodo.8436729>

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