The effect of management practices on bumblebee densities in hedgerow and grassland habitats

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Running head: Management type and bumblebee density

Abstract

Large-scale declines in pollinator species are a concern at present. Such declines have been attributed to a range of factors that act in tandem, rather than in isolation. Some of the most pervasive factors affecting pollinator populations are habitat loss and degradation, which results in the loss of floral resources, nesting sites and landscape connectivity. Intensification of agriculture and urbanisation are two major causes of such habitat alterations. Hedgerows and grasslands are two vital habitats for pollinators in European landscapes. When managed appropriately, these habitats may provide abundant floral resources and nesting opportunities, as well as connectivity between habitats in a fragmented landscape. This study examined the effects that management practices of hedgerows and grasslands may have on bumblebee species, an important group of wild pollinators. Bumblebee abundance was recorded using transect walks in managed and unmanaged sites, including both hedgerows and grasslands. Greater densities of bumblebees were found in unmanaged grasslands in comparison to managed grasslands. Unmanaged hedgerows were also found to have a greater density of bumblebees than managed hedgerows. These results indicate that sites which are less intensively managed provide a more suitable habitat for bumblebees. Therefore, our study underlines the importance of a) enforcing restrictions on hedge-cutting, and b) reducing the management intensity of grasslands to provide adequate habitat for pollinators.

Keywords: bumblebees; pollinators; management; anthropogenic disturbance; hedgerow; grassland; floral diversity.

Introduction

Animal (especially insect) pollination is necessary for the pollination of important crop species and wild plants, and thus is an important ecosystem service (Klein et al., 2007; Gallai et al., 2009; Ollerton et al., 2011). Indeed, the majority of the world's flowering plants are dependent on insects for their pollination (Ollerton et al., 2011) and further large-scale declines in pollinators could lead to a nutritional impoverishment of the human diet (Eilers et al., 2011).

Large-scale declines in pollinating insects have occurred across the globe (Williams & Osborne, 2009; Winfree et al., 2009; however, see Ghazoul, 2005). Regionally, such declines are particularly apparent in North America (Colla & Packer, 2008; Grixti et al., 2009; Cameron et al., 2011; Burkle et al., 2013) and Europe (Steffan-Dewenter et al., 2005; Biesmeijer et al., 2006; Kosior et al., 2007; Goulson et al., 2008; Potts et al., 2010a). Several causes have been proposed for such pollinator declines (Potts et al., 2010b), including land use change (Brown & Paxton, 2009; Winfree et al., 2009), agrochemicals (Alston et al., 2007; Holzschuh et al., 2008; Brittain et al., 2010), pathogens (Colla et al., 2006; Rosenkranz et al., 2010), alien species (Stout & Morales, 2009), and climate change (Williams et al., 2007).

Land use is probably the most important driver of bee declines (Brown & Paxton, 2009; Winfree et al., 2009). Although some alterations to habitat (e.g. urbanisation) may have positive effects on bee diversity and local abundance (Cane et al., 2006; Carré et al., 2009), changes in land use normally lead to the loss or degradation of suitable pollinator habitat, thus limiting their floral resources and potential nesting sites. Similar to global declines, natural and semi-natural habitat loss plays a huge role in the decline of pollinators in Ireland (Fitzpatrick et al., 2007; Fitzpatrick & Stout, 2015). The loss and changing nature of grasslands in Ireland is likely a major driver in pollinator losses, as well as shifts or contractions in their range (Fitzpatrick et al., 2007).

Twenty-one species of bumblebees have been recorded in Ireland. Six of these species are considered ‘threatened’, with a further 3 species being ‘near threatened’ (Fitzpatrick & Stout, 2015). The four bumblebee species that have declined the most in Ireland (*Bombus distinguendus, B. ruderarius, B. sylvarum* and *B. muscorum*), are all later emerging species associated with open grassland habitats (Fitzpatrick et al., 2007). Declines in these ecologically similar species are likely a result of the shift in the past 30 years from hay production to silage production. This shift in production decreases the amount of wild flowers occurring in meadows throughout the summer, when bees are more active (Vickery et al., 2001).

Many bumblebee species in Ireland have become rare in the eastern regions of their original ranges; this is likely a result of urbanisation and increasingly intensive agricultural practices in the east of the country (Santorum & Breen, 2005). Conversely, the midlands and western regions of Ireland have retained much of their grassland habitats and thus bumblebee diversity (Fitzpatrick et al., 2007).

Hedgerows are an important habitat for pollinators (Klein et al., 2007; Garibaldi et al., 2014; Goulson et al., 2015). Hedgerows provide food resources and safe nesting sites for pollinators, as well as acting as wildlife corridors, linking other suitable habitats (Stanley & Stout, 2013; Fitzpatrick & Stout, 2015). Hedgerows and uncropped field margins in close association are of particular benefit to pollinator populations in Irish agricultural landscapes (Ghazoul, 2005; Stanley & Stout, 2013).

As hedgerows and grasslands are important habitats for pollinator species, finding an appropriate management regime for these two habitat types is imperative to the persistence of sustainable populations of bumblebee species in Ireland. Both grasslands and hedgerows can be found in the Irish landscape in two distinct forms: managed and unmanaged. Our study seeks to establish which of these two management types in hedgerow and grassland habitats hosts a greater abundance of bumblebees. We predicted that unmanaged habitats, possibly having a greater floral diversity and abundance than managed habitats, will sustain higher bumblebee densities.

Materials and methods

Study System

Sixty study sites were selected in a rural area of County Carlow, in South East Ireland. All sites were contained within a triangular area of 19.76 km^2 of agricultural landscape. The greatest distance between two sites was 6.8 km. These 60 sites were selected based on their habitat type (hedgerow or grassland), and their management type (managed or unmanaged), resulting in 15 sites for each of the following four categories: managed hedgerow, unmanaged hedgerow, managed grassland, and unmanaged grassland. Managed hedgerows were characterised by a very uniform shape along their 100 m stretch. Very little, if any, vegetation in these managed hedgerows protruded out from the hedgerow. The grassy verge beneath managed hedgerow was generally mowed. Unmanaged hedgerows were less uniform in their shape in comparison to managed hedgerows; vegetation height and width often varied, protruding out from the hedge in areas, and no visible signs of recent physical management could be seen. Managed grasslands were within parks, gardens, and road verges, and were frequently mowed, in many cases once or twice every week. Unmanaged grasslands included undisturbed patches in farmlands, and abandoned areas in sports complexes and former construction sites. Areas that are purposely left idle and cultivated on occasion throughout the summer for hay or silage production were not included in our study. Use of herbicides was not observed in any of the sites. Each of the 60 sites was sampled four times (for a total of 240 transect walks, as explained below), on random occasions throughout the sampling period between 11th June and 2nd September 2017. Each of the four visits to a particular site took place at least one week apart. Three of these transects were discounted due to an alteration of management type taking place during the study. All transects were performed between the hours of 10 a.m. and 5 p.m., when bumblebees are most active. No transects were performed during or directly following periods of rain.

Prior to the study, we identified all sites, recorded their GPS locations, and measured the length (of hedgerows) or area (of grasslands). Floral diversity (as the number of different flowering plant species) at each site was recorded at the beginning of the study, prior to the first sampling taking place. If a new plant variety came into flower on subsequent visits to a site, this was added to the list of flowering plants for that site. The temperature (°C) and wind speed ($km*h^{-1}$) was logged prior to performing a transect walk at each site using a smartphone weather application (AccuWeather GPS Weather Widget).

The method of sampling differed between hedgerows and grasslands. In the case of hedgerows, a fixed 100-metre stretch of hedgerow was walked at a constant speed while scanning for bumblebees. In the case of grasslands, three repeated transect walks or 'laps' were carried out one after the other, due to the small size of some of the selected grasslands (range: 28 to 555 m^2; mean ± s.d. for unmanaged grasslands: 150.5 ± 96.74 m^2; mean ± s.d. for managed grasslands: 207.7 ± 135.16 m^2; t-test between areas of managed and unmanaged grasslands: $t_{28} = -1.15$, $p = 0.26$). These three repeated laps were performed within the grassland at a constant distance approximately 2 m from the site's outer perimeter. For each visit, the number of bumblebees present in a site was the maximum recorded over the three laps.

For both habitat types, we recorded bumblebees that were both landed on flowers and flying. We recorded many more landed than flying bumblebees (total number of bumblebees recorded in hedgerows: landed = 479 bumblebees, flying = 36 bumblebees; total number of bumblebees recorded in grasslands: landed = 273 bumblebees, flying = 44 bumblebees). Consequently, analyses considering only landed bumblebees or all bumblebees (i.e. landed plus flying bumblebees) offered qualitatively similar results. We only present results for the analyses considering all bumblebees recorded. Bumblebee species were not identified in this study.

Statistical Analyses

We conducted our statistical analyses using R, version 3.4.2 (R Core Team, 2017). We set significance at $p < 0.05$. As the sampling method of hedgerows and grasslands was dissimilar, data for both types of habitat were analysed separately.

We determined if the number of bumblebees differed between managed and unmanaged hedgerows implementing a generalised linear mixed model (GLMM), with the total number of bumblebees as the response; type of management, wind, and temperature as fixed factors; and site as a random factor (to account for the fact that we sampled each site on four different occasions). We fitted a GLMM with a negative binomial distribution and log link function using the glmer.nb function in the package lme4.

To determine if the number of bumblebees differed between managed and unmanaged grasslands, we fitted a GLMM with a Poisson distribution and log link using the function glmer in the package lme4. We used the maximum number of bumblebees observed over three consecutive laps as the response; type of management, wind, and temperature as fixed factors; site as a random factor; and area as an offset (to account for the fact that different grasslands had different areas).

To determine the effect of floral diversity on bumblebee abundance in managed and unmanaged habitats we implemented general linear models (GLM) using the function lm in the package stats, with the average of bumblebees recorded during the four visits to each site as the response (log transformed); and type of management and floral diversity as the fixed factors, including their interaction.

Results

We found more bumblebees in unmanaged hedgerows (mean ± standard deviation: 5.42 ± 5.97 bumblebees per 100 m hedgerow) than in managed hedgerows (3.25 ± 3.64 bumblebees per 100 m hedgerow; GLMM: $z = 2.06$, $p = 0.039$; Fig. 1A). Wind and temperature did not have an effect on the number of bumblebees recorded in hedgerows ($p > 0.2$ for both factors). Floral diversity did not have a significant effect on the number of bumblebees found in hedgerows overall ($t = -0.63$, $p = 0.53$) or depending on the type of management (interaction: $t = 1.26$, $p = 0.22$). This is not surprising, as floral diversity did not differ between managed hedgerows (mean ± standard deviation: 5.67 ± 1.8 species) and unmanaged hedgerows (5.87 ± 1.69 species; *t*-test: $t_{28} = 0.31$, $p = 0.76$). Unfortunately, we did not measure or estimate floral abundance.

We also found more bumblebees in unmanaged grasslands (mean ± standard deviation: 129.28 ± 129.54 bumblebees per hectare) than in managed grasslands (30.62 ± 32.08 bumblebees per hectare; GLMM: $z = 3.81$, $p = 0.0001$; Fig. 1B). Wind and temperature did not have an effect on the number of bumblebees recorded at grasslands ($p > 0.05$ for both factors). Floral diversity did not have a significant effect on the number of bumblebees found in grasslands overall ($t = 0.63$, $p = 0.54$) or depending on the type of management (interaction: $t = 0.66$, $p = 0.51$). Floral diversity did not differ between managed grasslands (mean ± standard deviation: 5.13 ± 2.13 species) and unmanaged grasslands (5.93 ± 1.91 species; *t*-test: $t_{28} = 1.08$, $p = 0.29$).

Discussion

Our results indicate that management of both hedgerows and grasslands can have a detrimental effect on bumblebee abundances. The influence of management on bumblebee densities was less apparent in hedgerows than in grasslands. One factor that can explain this finding is that hedgerows had not been managed as recently, or as frequently as grasslands over the sampling period (June 11^{th} to September 2^{nd}). This is due to the fact that the cutting of hedgerows is nationally prohibited between March 1^{st} and August 31^{st}; although some hedgerows may still be cut back under certain circumstances. Consequently, the first hedgerow transect of the study was walked, potentially, three months or more after the previous cut had taken place in February. This would have allowed plants in managed hedgerows an extended period to set seed and flower undisturbed, from early spring. This contrasts greatly to managed grassland habitats, which were often cut weekly and throughout the period of sampling. Although our aim was not to compare directly bumblebee densities between hedgerows and grasslands, we found that bumblebee densities were clearly higher in the linear habitat (hedgerow) than in the non-linear habitat (grassland), as previously shown (Osborne et al., 2008).

We found no difference in floral diversity between managed and unmanaged habitats. However, a greater abundance of floral resources is likely to make a site more valuable to bumblebees, as opposed to a site with fewer plants but with a higher variety of plant species. In fact, bumblebees have been documented to forage on a narrow range of flowering plant species (Santorum & Breen, 2005), so floral diversity may not be a particularly limiting factor to bumblebees. Although we did not measure floral abundance, this is likely to be greater in unmanaged habitats than in those that are routinely or intensively managed.

Managed grassland sites in this study were largely confined to roadsides, gardens or public spaces. In such locations, the nature of management is both frequent and intense, especially in the summer months when mowing is more common. Regular mowing will result in flowering plants being cut back; often at times when they are most useful to wild pollinators. Conversely, not managing grasslands will allow the available flora in the seed bank to set seed and flower undisturbed. As a result, flowering plants in unmanaged grasslands will likely be more abundant, and of greater value to bumblebees for a longer period of time than those in managed grasslands. Thus, unmanaged grasslands may be more reliable forage sites than managed grasslands.

A greater proportion of edge habitat will provide more benefits to wild pollinators in the form of floral resources and potential nesting sites. Edge habitat of grasslands and croplands is often much richer in floral resources than central areas (Gabriel & Tscharntke, 2007; Stanley & Stout, 2013). However, a few plant species abundant in central areas of grasslands (e.g. dandelions and clover) may also be highly important to pollinators during spring.

Neither of the assessed environmental conditions (temperature and wind speed) seemed to influence bumblebee densities to a great extent in this study. The summer months in Ireland are defined by a narrow range of temperatures. Transect walks from this study were carried out at temperatures between 14 °C – 25 °C. The low variability in temperature in Ireland likely means bumblebees can be active throughout the summer, independent of temperature on a given day. Hence, it is not surprising that we found no effect of temperature on bumblebee densities.

Wind speed was more variable than temperature (9 $km*h^{-1}$ – 30 $km*h^{-1}$). Although bumblebees may be less willing to fly greater distances, or with greater frequency, during periods of high winds, the range of wind speeds that we observed may not have been sufficient to significantly affect bumblebee behaviour. In any case, wind speed can be highly changeable during a day, and the number of bumblebees observed at any time may be determined by the current wind conditions.

In conclusion, here we show that greater densities of bumblebees are found in unmanaged grasslands and hedgerows than in managed habitats. Floral abundance may be a better measure of a habitat's suitability to bumblebees than floral diversity, and this should be a priority measurement in future studies of a similar nature. A recommendation derived from our findings is for management intensity and frequency to be reduced in order to make grasslands more hospitable for wild bumblebees. During times of peak bumblebee activity, management practices of grasslands should be limited as far as possible, to increase the availability of forage sources. Similarly, well-timed management of hedgerows, when necessary, should be implemented. Although management of hedgerows in February may not have had a great effect on bumblebees in mid-late summer (this study's sampling period), it may affect the floral resources and nesting sites available to them in spring, earlier in their life cycle (Fitzpatrick & Stout, 2015; Goulson et al., 2015). It is proposed that the period in which hedge-cutting is permitted (September – March) should not be extended; and further study should take place in spring to examine the effect of hedgerow management on these important pollinators earlier in the year.

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References

Alston, D.G., Tepedino, V.J., Bradley, B.A., Toler, T.R., Griswold, T.L., Messinger, S.M. (2007). Effects of the insecticide phosmet on solitary bee foraging and nesting in orchards of Capitol Reef National Park, Utah. *Environmental Entomology*, **36**, (4), pp.811-816.

Biesmeijer, J.C., Roberts, S.P., Reemer, M., Ohlemüller, R., Edwards, M., Peeters, T., Schaffers, A.P., Potts, S.G., Kleukers, R., Thomas, C.D., Settele, J. (2006). Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. *Science*, **313**, (5785), pp.351-354.

Brittain, C.A., Vighi, M., Bommarco, R., Settele, J., Potts, S.G. (2010). Impacts of a pesticide on pollinator species richness at different spatial scales. *Basic and Applied Ecology*, **11**, (2), pp.106-115.

Brown, M.J., Paxton, R.J. (2009). The conservation of bees: a global perspective. *Apidologie*, **40**, (3), pp.410-416.

Burkle, L.A., Marlin, J.C., Knight, T.M. (2013). Plant-pollinator interactions over 120 years: loss of species, co-occurrence, and function. *Science*, **339**, (6127), pp.1611-1615.

Cameron, S.A., Lozier, J.D., Strange, J.P., Koch, J.B., Cordes, N., Solter, L.F., Griswold, T.L. (2011). Patterns of widespread decline in North-American bumblebees. *Proceedings of the National Academy of Sciences of the United States of America*, **108**, pp.662-667.

Cane, J.H., Minckley, R.L., Kervin, L.J., Williams, N.M. (2006). Complex responses within a desert bee guild (Hymenoptera: Apiformes) to urban habitat fragmentation. *Ecological Applications*, **16***,* (2), pp.632-644.

Carré, G., Roche, P., Chifflet, R., Morison, N., Bommarco, R., Harrison-Cripps, J., Krewenka, K., Potts, S.G., Roberts, S.P., Rodet, G., Settele, J. (2009). Landscape context and habitat type as drivers of bee diversity in European annual crops. *Agriculture, Ecosystems & Environment*, **133***,* (1-2), pp.40-47.

Colla, S.R., Otterstatter, M.C., Gegear, R.J., Thomson, J.D. (2006). Plight of the bumble bee: pathogen spillover from commercial to wild populations. *Biological Conservation*, **129**, (4), pp.461-467.

Colla, S.R., Packer, L. (2008). Evidence for decline in eastern North American bumblebees (Hymenoptera: Apidae), with special focus on *Bombus affinis* Cresson. *Biodiversity and Conservation*, **17***,* (6), pp.1379.

Eilers, E.J., Kremen, C., Greenleaf, S.S., Garber, A.K., Klein, A.M. (2011). Contribution of pollinator-mediated crops to nutrients in the human food supply. *PLoS One*, **6***,* (6), pp.21363.

Fitzpatrick, Ú., Murray, T.E., Paxton, R.J., Breen, J., Cotton, D., Santorum, V., Brown, M.J. (2007). Rarity and decline in bumblebees–a test of causes and correlates in the Irish fauna. *Biological Conservation*, **136***,* (2), pp.185-194.

Fitzpatrick, Ú., Stout, J. (2015). All-Ireland pollinator plan 2015–2020. *National Biodiversity Data Centre Series* **No. 3**, Waterford, Ireland.

Gabriel, D., Tscharntke, T. (2007). Insect pollinated plants benefit from organic farming. *Agriculture, Ecosystems & Environment*, **118***,* (1-4), pp.43-48.

Gallai, N., Salles, J.-M., Settele, J., Vaissière, B.E. (2009) Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. *Ecological Economics*, **68**, 810- 821.

Garibaldi, L.A., Carvalheiro, L.G., Leonhardt, S.D., Aizen, M.A., Blaauw, B.R., Isaacs, R., Kuhlmann, M., Kleijn, D., Klein, A.M., Kremen, C., Morandin, L. (2014). From research to action: enhancing crop yield through wild pollinators. *Frontiers in Ecology and the Environment*, **12***,* (8), pp.439-447.

Ghazoul, J. (2005). Buzziness as usual? Questioning the global pollination crisis. *Trends in Ecology & Evolution*, **20***,* (7), pp.367-373.

Goulson, D., Lye, G.C., Darvill, B. (2008). Decline and conservation of bumblebees. *Annual Review of Entomology*, **53**, pp.191-208.

Goulson, D., Nicholls, E., Botías, C., Rotheray, E.L. (2015). Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. *Science*, **347**, (6229), pp.1255957.

Grixti, J.C., Wong, L.T., Cameron, S.A., Favret, C. (2009). Decline of bumble bees (*Bombus*) in the North American Midwest. *Biological Conservation*, **142**, (1), pp.75-84.

Holzschuh, A., Steffan-Dewenter, I., Tscharntke, T. (2008). Agricultural landscapes with organic crops support higher pollinator diversity. *Oikos*, **117**, (3), pp.354-361.

Klein, A.M., Vaissiere, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C., Tscharntke, T. (2007). Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society of London B: Biological Sciences*, **274***,* (1608), 303-313.

Kosior, A., Celary, W., Olejniczak, P., Fijał, J., Król, W., Solarz, W., Płonka, P. (2007). The decline of the bumble bees and cuckoo bees (Hymenoptera: Apidae: Bombini) of Western and Central Europe. *Oryx*, **41***,* (1), pp.79-88.

Ollerton, J., Winfree, R., Tarrant, S. (2011). How many flowering plants are pollinated by animals? *Oikos*, **120***,* (3), pp.321-326.

Osborne, J.L., Martin, A.P., Shortall, C.R., Todd, A.D., Goulson, D., Knight, M.E., Hale, R.J., Sanderson, R.A. (2008). Quantifying and comparing bumblebee nest densities in gardens and countryside habitats. *Journal of Applied Ecology*, **45**, (3), pp. 784-792.

Potts, S.G., Roberts, S.P., Dean, R., Marris, G., Brown, M.A., Jones, R., Neumann, P., Settele, J. (2010 a). Declines of managed honey bees and beekeepers in Europe. *Journal of Apicultural Research*, **49***,* (1), pp.15-22.

Potts, S.G., Biesmeijer, J.C., Kremen, C., Neumann, P., Schweiger, O., Kunin, W.E. (2010 b). Global pollinator declines: trends, impacts and drivers. *Trends in Ecology & Evolution*, **25**, (6), 345-353.

R Core Team (2017). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/

Rosenkranz, P., Aumeier, P., Ziegelmann, B. (2010). Biology and control of *Varroa destructor*. *Journal of Invertebrate Pathology*, **103**, pp.96-119.

Santorum, V., Breen, J. (2005). Bumblebee diversity on Irish farmland. *Tearmann: Irish Journal of Agri-environmental Research,* **4**, pp.79-90.

Stanley, D.A., Stout, J.C. (2013). Quantifying the impacts of bioenergy crops on pollinating insect abundance and diversity: a field-scale evaluation reveals taxon-specific responses. *Journal of Applied Ecology*, **50***,* (2), pp.335-344.

Steffan-Dewenter, I., Potts, S.G., Packer, L. (2005). Pollinator diversity and crop pollination services are at risk. *Trends in Ecology & Evolution*, **20***,* (12), pp.651-652.

Stout, J.C., Morales, C.L. (2009). Ecological impacts of invasive alien species on bees. *Apidologie*, **40**, (3), pp.388-409.

Vickery, J.A., Tallowin, J.R., Feber, R.E., Asteraki, E.J., Atkinson, P.W., Fuller, R.J., Brown, V.K. (2001). The management of lowland neutral grasslands in Britain: effects of agricultural practices on birds and their food resources. *Journal of Applied Ecology*, **38**, (3), pp.647-664.

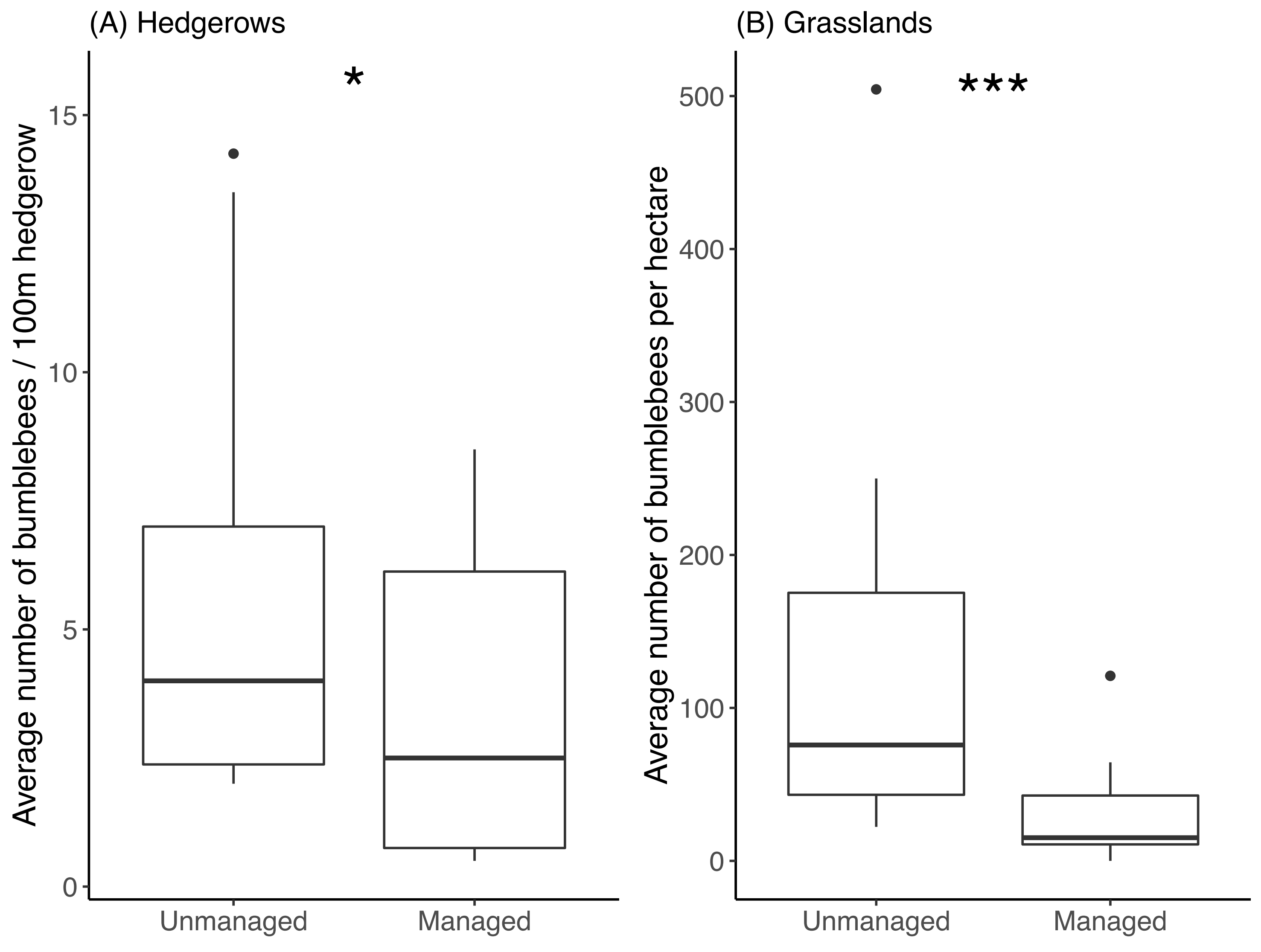
Williams, P.H., Araújo, M.B., Rasmont, P. (2007). Can vulnerability among British bumblebee (*Bombus*) species be explained by niche position and breadth? *Biological Conservation*, **138**, (3-4), pp.493-505.

Williams, P.H., Osborne, J.L. (2009). Bumblebee vulnerability and conservation world-wide. *Apidologie*, **40**, (3), pp.367-387.

Winfree, R., Aguilar, R., Vázquez, D.P., LeBuhn, G., Aizen, M.A. (2009). A meta-analysis of bees' responses to anthropogenic disturbance. *Ecology*, **90**, (8), pp.2068-2076.

Figure legends

Fig. 1. Bumblebee abundance recorded in hedgerows (A) and grasslands (B) that were managed or unmanaged. Values represent the average of bees recorded during four different visits to each site. For each boxplot, the bar within each box represents the median, each box represents the first and third quartiles (or 25th and 75th percentiles), the two whiskers represent the maximum values that are within 1.5 * IQR of the box (where IQR or inter-quartile range is the distance between the first and third quartiles), and points beyond the whiskers represent outliers. Significant differences between managed and unmanaged habitats were obtained using generalised linear mixed models (* denotes $p < 0.05$; *** denotes $p < 0.0005$).


(A) Hedgerows
*
Average number of bumblebees / 100m hedgerow
15
10
5
0
Unmanaged
Managed
(B) Grasslands
***
Average number of bumblebees per hectare
500
400
300
200
100
0
Unmanaged
Managed