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<td><strong>Author(s)</strong></td>
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<td><strong>Publication Date</strong></td>
<td>2013-06-19</td>
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<td><strong>Link to publisher's version</strong></td>
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Ecological Variation in Response to Mass-Flowering Oilseed Rape and Surrounding Landscape Composition by Members of a Cryptic Bumblebee Complex

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Abstract

The Bombus sensu stricto species complex is a widespread group of cryptic bumblebee species which are important pollinators of many crops and wild plants. These cryptic species have, until now, largely been grouped together in ecological studies, and so little is known about their individual colony densities, foraging ranges or habitat requirements, which can be influenced by land use at a landscape scale. We used mass-flowering oilseed rape fields as locations to sample bees of this complex, as well as the second most common visitor to oilseed rape B. lapidarius, and molecular RFLP methods to distinguish between the cryptic species. We then used microsatellite genotyping to identify sisters and estimate colony densities, and related both proportions of cryptic species and their colony densities to the composition of the landscape surrounding the fields. We found B. lucorum was the most common member of the complex present in oilseed rape followed by B. terrestris. B. cryptarum was also present in all but one site, with higher proportions found in the east of the study area. High numbers of bumblebee colonies were estimated to be using oilseed rape fields as a forage resource, with B. terrestris colony numbers higher than previous estimates from non-mass-flowering fields. We also found that the cryptic species responded differently to surrounding landscape composition: both relative proportions of B. cryptarum in samples and colony densities of B. lucorum were negatively associated with the amount of arable land in the landscape, while proportions and colony densities of other species did not respond to landscape variables at the scale measured. This suggests that the cryptic species have different ecological requirements (which may be scale-dependent) and that oilseed rape can be an important forage resource for many colonies of bumblebees. Given this, we recommend sustainable management of this crop to benefit bumblebees.

Citation: Stanley DA, Knight ME, Stout JC (2013) Ecological Variation in Response to Mass-Flowering Oilseed Rape and Surrounding Landscape Composition by Members of a Cryptic Bumblebee Complex. PLoS ONE 8(6): e65516. doi:10.1371/journal.pone.0065516

Editor: Nina Farwig, University of Marburg, Germany

Received October 17, 2012; Accepted May 2, 2013; Published June 19, 2013

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Funding: This research was funded by the project SIMBIOSYS (www.simbiosys.ie, 2007-8-CD-1-51) as part of the Science, Technology, Research and Innovation for the Environment (STRIVE) Programme, financed by the Irish Government under the National Development Plan 2007–2013, administered on behalf of the Department of the Environment, Heritage and Local Government by the Irish Environmental Protection Agency (EPA). The funders had no role in study design, data collection and analysis, decision to publish, or preparation of the manuscript.

Competing Interests: The authors have read the journal’s policy and have the following conflicts: Jane Catherine Stout is an Academic Editor for PLOS ONE. This does not alter the authors’ adherence to all the PLOS ONE policies on sharing data and materials.

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Introduction

One of the most common bumblebees in North West Europe, responsible for both crop and wild plant pollination [1], is the Bombus sensu stricto group, a cryptic complex of five species: B. cryptarum, B. lucorum, B. magnus, B. terrestris and B. sporadicus [2]. Although advances in the taxonomy of this group have been made [2], most ecological studies of bumblebees and the pollination services they deliver have considered these species as a single group (e.g. [3,4]), since the workers are morphologically indistinguishable in the field [5,6,7]. However, this means that ecological differences between the species may have been overlooked [8,9] rendering the pollination services delivered by, and conservation status of the species belonging to, this cryptic complex impossible to assess [10].

Over the past few decades, declines in both range and abundance have been documented for several bumblebee species in both North America and Europe, whilst other species have shown no decline, and in some cases have spread and become more abundant [3,4,11,12]. It is thought that the longer tongued bumblebee species, those with later starting colony development cycles, and those at range edges and with small climatic ranges, are at most risk of decline [3,13], driven largely by agricultural intensification [14,15,16]. Members of the Bombus sensu stricto (henceforth B. s. str.) cryptic complex of species are all relatively short tongued, have earlier starting colony cycles and are assumed to be ecological generalists [3,4], and therefore may not be at the same risk of decline as some other species. The B. s. str. group in Ireland contains both species which are classified according to the IUCN Red List criteria as of Least Concern (B. lucorum and B. terrestris) but also species which cannot be assigned to a threat category because they are currently Data Deficient (B. cryptarum and B. magnus) [10]. However, due to the cryptic nature of these species, it is quite possible that these classifications are inappropriate as the relative proportions of these cryptic species in both semi-natural and agricultural sites are not well known.
Individuals of the *B. s. str.* group are the most commonly observed bumblebees visiting mass-flowering oilseed rape in Ireland (followed by *B. lapidarius*; Stanley & Stout unpublished data), and as the crop benefits from insect pollination [17], are likely to be important pollinators. Although it has been grown in Europe for centuries, the distribution of oilseed rape is changing and it is becoming more common largely due to its use as a bioenergy crop [18]. Since bumblebees, including the *B. s. str.* group, have large foraging ranges in comparison to other bee species [19], for a summary of bumblebee foraging ranges see [20,21]), and are influenced by the composition of habitats and features within landscapes at both smaller [22] and larger spatial scales [23], they may be sensitive to changes in cultivation patterns of mass-flowering crops such as oilseed rape. Bumblebee foraging distances can vary with the proportion of forage habitats in the landscape [24], and landscape scale factors can also influence colony fitness [25]. Several field surveys have demonstrated that the abundance of bumblebees can be influenced by landscape features. For example, more bees of the *B. s. str.* group (as well as *B. lapidarius* and *B. pascuorum*) were observed when surrounding landscapes (up to 3 km from the sampling site) contained a high availability of mass-flowering oilseed rape in Germany [26], while oilseed rape fields had more bumblebees of all species when there was more pasture in the surrounding landscape (at an 800 m radius) in Canada [27]. Conversely, other studies have shown no relationship between the abundance of bumblebees and landscape features [17,22,20,26]. Many studies so far have grouped all bumblebee species together when looking at how bees are influenced by the surrounding landscape, when species-specific responses are likely, and have primarily focussed on the abundance and richness of bumblebees [26,27].

As colonial organisms, work on the effect of the surrounding landscape at the reproductive level (on colony densities) is important in order to predict impacts on populations, and landscape scale effects on colony densities of some distinguishable species have been investigated [24,25,29]. Since it is notoriously difficult to find and quantify colony densities using observational methods [30], molecular techniques have been developed allowing estimations of colony densities based on the relationships of bumblebee workers or sisters to each other [31,32,33]. This has allowed estimation of nest or colony density and foraging distances, which differ remarkably between different bumblebee species (for a review see [20]. As the most distinguishable of the *B. s. str.* complex, with queens and some workers having a darker buff-coloured tail, *B. terrestris* has been extensively studied (although lighter coloured individuals may have been overlooked; [3]). However, colony densities, foraging distances, landscape scale effects and even distributions of the other species in the complex are not well known. This ecological information is essential to understand how to manage, protect and conserve these important pollinator species, and may help to explain their co-existence. The aim of this study was to investigate ecological differences between species within the *B. s. str.* complex by estimating the relative abundance and colony density of each species and then relating those data to landscape composition along an agricultural landscape gradient. For comparison another non-cryptic short tongued species, *B. lapidarius* (which is second in abundance to the cryptic complex in oilseed rape fields, but designated as Near Threatened (NT) in Ireland as a whole), was also included in the study. We used mass-flowering oilseed rape fields as a sampling unit as they are commonly visited by the *B. s. str.* group and are likely to attract bumblebee colonies from the surrounding agricultural matrix. Specifically we aimed to:

1) investigate whether all of the cryptic species of the *Bombus s. str.* group are found foraging in oilseed rape fields and determine the relative abundance of each species.
2) estimate the number of colonies (colony density) of the cryptic species, and *B. lapidarius*, using oilseed rape fields as a foraging resource; and compare the number of colonies in oilseed rape fields to previously published colony densities in agricultural habitats.
3) identify ecological differences between the cryptic species of the *B. s. str.* complex in agricultural habitats by investigating whether the number of estimated colonies, and relative proportions, of the cryptic species relate differently to the composition of the landscape surrounding the oilseed rape fields.

**Methods**

**Site Selection**

Fourteen spring oilseed rape (*canola, Brassica napus L.*) fields were selected for study in an area of 114 km$^2$ in South East Ireland in 2010 (Fig. 1), where beef and dairy farming are interspersed with arable, and oilseed rape is relatively rare. All fields were privately owned, and permission to sample was obtained from relevant land owners. Fields were selected along a landscape gradient of arable to pasture dominated landscapes based on CORINE land cover data [34] (Table 1). Fields were on average 15.28 km apart (range 2.9–48.2 km). Due to the current knowledge on average foraging ranges of our focal bumblebee species (for summaries of foraging distances of *B. terrestris* and *B. lapidarius* see [20,21]; foraging distances of *B. cryptarum* and *B. hacorum* are unknown), it was assumed that given our average inter-site distance, the number of sites sharing bees from the same colony would be negligible overall.

**Sample Collection**

Fields were visited once between 13$^{th}$ June and 12$^{th}$ July 2010 during the peak flowering of spring oilseed rape. Firstly, two 100 m transects were walked at a slow steady pace in each site to identify the main bumblebees present. Then, individuals of the *B. s. str.* group were sampled qualitatively in each field by walking around the entire perimeter of the field on the outermost tramline (approximately 20 m into the crop) and catching individuals until a target of 60 were caught. A non-lethal sample of tarsal segment was taken from the mid-leg and stored in 75% ethanol for later DNA analysis [35]. In the seven sites where *B. lapidarius* occurred in the transects, a target of 50 *B. lapidarius* individuals were also sampled in the same way. Sampled individuals were marked using a non-toxic marker pen to avoid re-capture of the same individuals. A similar amount of time was also spent catching bees around the perimeter (field margin) of a non mass-flowering field directly adjacent to the oilseed rape field. This was initially done to investigate patterns of worker distribution from individual colonies, but very few bees from the same colonies (sisters) were found in either field, which meant that we could not investigate this further. It was only possible to sample *B. s. str.* workers in nine adjacent fields as there were no suitable sampling areas (flowering vegetation) at the other five fields. Most *B. lapidarius* individuals were primarily attracted to oilseed rape fields; individuals were rarely present in adjacent fields and so numbers were too low to include in estimates of colony density. A total of 1362 individuals from the *B. s. str.* group and 330 *B. lapidarius* individuals were sampled overall.
Landscape Characterisation

A detailed map of the landscape surrounding each oilseed rape field up to a 700 m radius from the centroid of each field was also produced (Fig. 1). This radius was chosen based on the estimated foraging distance of the focal bee species (as reviewed in [20]). Land cover was divided into the following categories (Tables S1 & S2 in supporting information): mass-flowering crops (average across all sites: 79% oilseed rape, 14% potatoes, 7% field beans), non-mass-flowering arable land, grassland, forestry and extensively modified human surfaces (including buildings, yards and gardens). Land cover types were ground-truthed for each landscape as accurate distinction using aerial photographs was not possible. The length of field boundaries and area of fields sampled were also quantified using ortho-photographs and Ordnance Survey maps. All landscape analyses were carried out in ESRI ArcGIS 9.3.

Species Differentiation and Microsatellite Genotyping

DNA was extracted from tarsal segments by pulverising each sample after cooling in liquid nitrogen, and adding 300 µL 10% solution of Chelex 100 heated to 80°C to each sample. Samples were then heated to 100°C for 15 minutes before centrifuging and finally cooling to 4°C. We used a Restriction Fragment Length Polymorphism (RFLP) method developed by Murray et al. [8] to definitively assign each sample to one of the cryptic species. Samples were amplified using a polymerase chain reaction (PCR), digested using specific restriction enzymes (EcoNI and HinFI) and then visualised using electrophoresis in 2% agarose gels, where each cryptic species has a unique banding pattern [8]. Sixteen individuals (including a mixture of ambiguous and confirmed banding patterns) were also sequenced at partial mitochondrial COI gene to confirm RFLP identities [36].

Samples from all four species were subsequently genotyped at 14 microsatellite loci in two multiplex reactions (all: B10, B11, B96, B100, B118, B124, B126, B132, BT08, BT11, BL02, BL06, BTERN01, B. terrestris and B. lucorum; BL03, B. lapidarius and B. cryptarum; BL11, Table 2) [37,38,39]. PCR products were visualised on an ABI 3730xl automated sequencer (Applied Biosystems) using GeneScan™ 500 LIZ® size standard, and alleles were sized using GENEMAPPER® software (Applied Biosystems). Where a sample failed to amplify at any locus on the first attempt, or where there was any case of scoring ambiguity, a new PCR was run and all loci were re-amplified. This also allowed calculation of scoring and allelic drop-out error rates for...
loci that were amplified twice. After two attempts, B100 still failed to amplify sufficiently for accurate scoring in both *B. terrestris* and *B. lucorum*, and so these loci were omitted from any further analyses.

**Table 2.** Characteristics of the microsatellite loci used in each species.

<table>
<thead>
<tr>
<th>Locus</th>
<th><em>B. cryptarum</em></th>
<th><em>B. terrestris</em></th>
<th><em>B. lucorum</em></th>
<th><em>B. lapidarius</em></th>
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<tbody>
<tr>
<td></td>
<td>AR</td>
<td>Fis ± SE</td>
<td>N</td>
<td>AR</td>
</tr>
<tr>
<td>B10</td>
<td>1.92</td>
<td>0.029 ± 0.032</td>
<td>22</td>
<td>5.35</td>
</tr>
<tr>
<td>B100</td>
<td>1.78</td>
<td>0.088 ± 0.072</td>
<td>10</td>
<td>na</td>
</tr>
<tr>
<td>B11</td>
<td>1.45</td>
<td>0.137 ± 0.046</td>
<td>8</td>
<td>3.13</td>
</tr>
<tr>
<td>B118</td>
<td>1.72</td>
<td>0.138 ± 0.050</td>
<td>7</td>
<td>3.48</td>
</tr>
<tr>
<td>B124</td>
<td>1.67</td>
<td>0.008 ± 0.043</td>
<td>8</td>
<td>4.94</td>
</tr>
<tr>
<td>B126</td>
<td>1.89</td>
<td>0.099 ± 0.037</td>
<td>19</td>
<td>4.30</td>
</tr>
<tr>
<td>B132</td>
<td>1.8</td>
<td>0.013 ± 0.027</td>
<td>15</td>
<td>4.34</td>
</tr>
<tr>
<td>B96</td>
<td>1.51</td>
<td>0.247 ± 0.096</td>
<td>5</td>
<td>3.00</td>
</tr>
<tr>
<td>B02</td>
<td>1.92</td>
<td>0.026 ± 0.025</td>
<td>22</td>
<td>4.59</td>
</tr>
<tr>
<td>B03</td>
<td>−</td>
<td>−</td>
<td>−</td>
<td>4.51</td>
</tr>
<tr>
<td>B06</td>
<td>1.93</td>
<td>−0.000 ± 0.031</td>
<td>25</td>
<td>2.65</td>
</tr>
<tr>
<td>BL11</td>
<td>1.93</td>
<td>0.040 ± 0.045</td>
<td>24</td>
<td>−</td>
</tr>
<tr>
<td>BT08</td>
<td>1.79</td>
<td>0.168 ± 0.046</td>
<td>14</td>
<td>6.05</td>
</tr>
<tr>
<td>BT11</td>
<td>1.85</td>
<td>−0.010 ± 0.005</td>
<td>13</td>
<td>3.41</td>
</tr>
<tr>
<td>BTERN01</td>
<td>1.82</td>
<td>−0.007 ± 0.051</td>
<td>13</td>
<td>4.67</td>
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*population N was removed from analysis of *B. cryptarum* as there were no allele scores in that population for BT11. AR = average allelic richness across populations, Fis = inbreeding coefficient jack-knifed across populations (calculated in FSTAT) and N = total number of alleles. Minimum sample sizes for AR calculations are as follows: *B. cryptarum* = 1, *B. terrestris* = 6, *B. lucorum* = 12, *B. lapidarius* = 20. Loci identified by Estoup et al. [37], Estoup et al. [38] & Funk et al. [39].

doi:10.1371/journal.pone.0065516.t002

**Genetic Data Analyses**

For analysis, we included any samples with a minimum of seven of the 13 loci scored for *B. terrestris* and *B. lucorum*, and a minimum of six loci for *B. cryptarum* and *B. lapidarius*. All data were analysed on a per site basis. Genotypes were checked for typographic error.
and null alleles using MICRO-CHECKER [40]. We then used the program COLONY [41] to identify the number of colonies sampled in each site (colony density), using allelic drop out and scoring error rates calculated from re-scoring. COLONY implements a maximum likelihood sibship reconstruction method [42] and has been shown to give the most accurate sibship reconstruction when compared with other methods [43]. Due to the assumptions of COLONY, GENEPOP 4.1 [44] was used to test for deviations from Hardy Weinberg equilibrium (HWE) of individual loci by site using a probability test, and linkage disequilibrium between loci across all sites, using Bonferroni corrections for multiple comparisons of loci per individual. Summary characteristics of microsatellite loci were calculated using FSTAT [45] (Table 2). For GENEPOP and FSTAT analyses only one individual per colony was retained, as inclusion of closely related family members will inevitably lead to inflated homozygosity estimates that can lead to spurious deviations from HWE expectations [32,46].

However, sampling was not exhaustive, and the COLONY estimates are based only on workers successfully sampled from each site. Therefore, to account for the number of colonies we did not sample at each site, we also estimated the total number of colonies (total colony density) present in each site for each species. To do this we used the CAPWIRE programme [47] which is a mark-recapture software that allows for multiple sampling of an individual and can also be used for estimating number of bumblebee colonies [25]. CAPWIRE implements two different estimation methods; the Even Capture Model (ECM) assumes equal chances of sampling bees from the same colony [47] and provides very similar estimates to those obtained in previous studies using the truncated Poisson method [25], while the Two Inmate Rate Model (TIRM) assumes unequal rates of capture of different colonies. Although the TIRM method has been shown to be most useful for estimating the number of bumblebee colonies in other work [25], a likelihood ratio test (LRT) on our data found the ECM method to be preferable in the majority of cases (Table S3) and so we use this estimate in landscape analyses here. However, results of TIRM estimations can be found in the supporting information (Table S3). CAPWIRE models were run in 0.1 increments with capturability ratios of minimum 1, maximum 20; 95% confidence intervals for the estimate on population size based on 1000 bootstrap replicates; a largest population size of 750 for dimensioning; and a likelihood ration rejection region of 0.2 when conducting likelihood ratio tests. We estimated total colony density in each field two ways: firstly we used sisters identified in the oilseed rape field only to get an estimate of colonies using that resource. Secondly, since very low numbers of sister pairs were identified within the oilseed rape fields for both B. cryptarum and B. lucorum (and sister pairs or recaptures are necessary for further estimation using CAPWIRE or truncated Poisson methods), we pooled these data with those from the adjacent field (these pooled data henceforth referred to as “site”) for each species to increase sample sizes and number of sister pairs. This allowed us to get a more accurate estimate of the numbers of colonies foraging in the area.

We also calculated the total number of colonies per km² for the two species (B. terrestris and B. lapidarius) for which estimated foraging distances have been published (B. terrestris 758 m, B. lapidarius 450 m, [33]). These calculations per km² from our study (based on B. terrestris from 12 fields and B. lapidarius from 7 fields) were then compared to those from other studies (B. terrestris previous estimates data from a number of sources (5 data points), summarised in [20]; B. lapidarius previous estimates data (12 data points) from [25,23]) using non-parametric Kruskal-Wallis tests.

Landscape Analyses

Both proportions and colony density estimates for each species were initially investigated for correlative relationships with geographic location (TIXm and y co-ordinates), while colony density estimates were also investigated for relationships with the area sampled and the number of individuals sampled using Spearman’s rank correlation.

To test whether the cryptic species responded differently to the composition of the landscape surrounding the sites, generalized linear modelling was used. Proportions and total estimated colony densities (from CAPWIRE, using ECM) of each species were modelled as separate response variables, and landscape composition variables were predictors. For the proportions of each cryptic species, binomial GLMs were used to account for proportional data, and corrected for overdispersion using quasi-binomial GLMs if necessary. For models of the colony density estimates of each species, Poisson GLMs were used and standard errors corrected for overdispersion using quasi-poisson errors. Landscape variables were first normalised (variable-mean)/standard deviation), and after removing variables that were highly co-linear (see Tables S1 & S2), area of arable land, artificial surfaces, mass-flowering crops, forestry and length of field boundaries were used in the models. Significance of terms was assessed using Z tests in the multicomp package [48] which corrects for multiple comparisons, and full models are presented (simplified models are also available in Table S4). All models were validated by plotting deviance and Pearson residuals against fitted values and explanatory variables, and by normal QQ-plots [49]. For colony densities, only sites where an estimate was obtained were used in analyses (i.e. sites where no sisters were found were not used as no accurate estimate could be calculated, thus perhaps excluding those with the highest colony densities). B. cryptarum colony density estimates had one outlier (site T) that had a much higher estimate than all other sites (with only one sister pair found in the largest sample of this species), and so landscape analyses were carried out both including and excluding this site. All analyses were carried out using the stats package in R version 2.15.2 [50].

Results

Proportions of Cryptic Species

Three of the four species of the B. s. str. group found in Ireland were found foraging in oilseed rape fields – B. cryptarum, B. lucorum and B. terrestris. No B. magnus individuals were found in any of the fields studied. The most abundant species was B. lucorum (mean proportion of individuals per field 0.47±0.04 standard error), followed by B. terrestris (0.39±0.06), B. cryptarum was also present in all but one of the fields, but in lower numbers that the other two species (0.14±0.03, Fig. 2). Proportions of all three species varied among fields (Fig. 2).

Microsatellite Genotyping

Thirteen loci were scored for B. lucorum and B. terrestris, 14 for B. cryptarum and 11 for B. lapidarius, with variation between species in terms of loci characteristics (Table 2). There were no typographical errors found using Microchecker.

For B. lucorum and B. cryptarum a global test showed no overall deviations of any loci from HWE (global Fishers test: B. lucorum \( \chi^2 = 373.24, df = 364, p = 0.36, B. cryptarum \chi^2 = 271.16, df = 316, p = 0.97 \)). There was no significant linkage disequilibrium detected between loci for B. cryptarum. Although a number of loci showed significant linkage disequilibrium using a global test across all populations (sites) for B. lucorum, on further investigation of each case this was driven by linkage found on one population only. Due

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**Cryptic Bumblebees in Oilseed Rape**
to the small number of populations affected (1 out of 14) all loci were retained in further analyses.

For B. lapidarius, BT11 showed significant deviation from HWE in 3 of the 6 populations, possibly due to a deficit of heterozygotes suggesting the presence of a null allele. Therefore this locus was removed from analysis and COLONY sibships re-run without it. There were no deviations from HWE for any of the remaining loci removed from analysis and COLONY sibships re-run without it, suggesting the presence of a null allele. Therefore this locus was in 3 of the 6 populations, possibly due to a deficit of heterozygotes.

There were no significant deviations from HWE in any of the remaining loci (global Fishers test: $\chi^2 = 101.18$, df = 140, $p = 0.99$), and no significant linkage disequilibrium between any loci.

For B. terrestris, a global Fishers test showed significant deviation from HWE ($\chi^2 = \infty$, df = 310, $p < 0.001$). However, on further investigation this was caused by a small number of loci in four populations only: BT08 in site A, B132 in site B, B126 in site F and BT11 and BT08 in site M. The following loci showed significant linkage disequilibrium but again in one population only: B132 and B124 in site B, BL03 and BL06 in site D and B126 and B118 in site E. Due to the small number of populations affected (1 or 2 out of 14 in all cases) all loci were retained in further analyses.

**Colony Estimation**

Low numbers of sister pairs of all species were found within each field, suggesting that high numbers of colonies were using this mass-flowering resource (Table 1, Table S3). Most colonies were represented by a single worker, and the maximum numbers of sisters from any one colony was four (from an average of 31 individuals per site). To examine the total estimated density of colonies using mass-flowering oilseed rape fields as a resource, we first estimated colony densities using bees caught only in the oilseed rape. Colony estimations were not possible in fields where all sampled colonies are represented by one individual only, as this represents a potentially endless population; therefore estimations of total colony densities were possible in 12 fields for B. terrestris (mean 131, range 19–303 colonies), but only in one field for B. cryptarum (41 colonies), and four fields for B. lucorum (mean 214, range 100–303 colonies); despite similar sample sizes to previous studies (e.g. [20,25]). B. lapidarius was only found in sufficient numbers in oilseed rape fields and colony estimates ranged from 38–84 colonies per field (Table 1).

Given these limitations, we also pooled data from both the oilseed rape field and adjacent field (henceforth “site”) to estimate total colony densities, as sample sizes (and number of sister pairs) were larger, allowing estimates of total colony density using a larger number of sites (Table S3). Using an average across all sites where estimations were possible, the highest number of colonies found were of B. lucorum, then B. terrestris and then B. cryptarum (Fig. 3, Table 1). Colony densities of all species were not significantly related to the size of the fields sampled. Colony densities of B. lucorum and B. lapidarius (using only sites where colonies were sampled) were not significantly related to the number of individuals sampled, but colony densities of B. cryptarum (Spearman’s rank correlation: $Rho = 0.86$, $S = 8$, $p = 0.02$) and B. terrestris (Spearman’s rank correlation: $Rho = 0.85$, $S = 44$, $p < 0.001$) were, suggesting that larger sample sizes may have detected more colonies. In addition, differences in allelic richness between species may indicate differences in power of resolution between species (Table 2). Given these factors, actual values of estimates (rather than patterns) should not be over-interpreted.

Using total colony densities calculated per km² (Table S3), we found density of colonies of B. terrestris from our samples using oilseed rape fields to be significantly higher than previously published estimates (Kruskal-Wallis $\chi^2 = 4.9$, df = 1, $p = 0.027$, n (our study) = 12, n (previous studies) = 5, Fig. 3), while colonies of B. lapidarius were not (Kruskal-Wallis $\chi^2 = 3.46$, df = 1, $p = 0.063$, n (our study) = 7, n (previous studies) = 12, Fig. 3). Comparisons for B. cryptarum and B. lucorum were not possible as, to our knowledge, there are no previously published estimates of colony densities for these species.

**Landscape Analyses**

Proportions of B. cryptarum were positively related to longitude (Spearman’s rank correlation: $S = 206$, $p = 0.046$, $Rho = 0.55$); higher proportions of the species were found in the eastern part of the study area (Fig. 1), where there were also fewer B. terrestris found (Spearman’s rank correlation: $S = 686$, $p = 0.07$, $Rho = -0.51$). Proportions of B. cryptarum were negatively associated with the amount of arable land (Table 3, Fig. 4), while proportions of the other cryptic species were not related to landscape compositional variables at the 700 m radius studied (Table S5). Colony densities of B. lucorum were also negatively related to the amount of arable land in the landscape (Table 3, Fig. 4), while other colony densities were not related to landscape variables (Table S5).
Discussion

Bumblebee communities are often composed of a number of morphologically similar species, including members of the cryptic *B. s. str* complex. However, subtle ecological differences may exist among species, which have been overlooked in the majority of previous studies which have grouped them together. We have shown that three members of the *B. s str.* complex co-exist in mass-flowering fields in Irish farmland, with *B. lucorum* most ubiquitously abundant (in accordance with other studies in different habitats; [8,31]), but with *B. terrestris* and *B. cryptarum* also common. The fourth species of the complex, *B. magnus*, was not observed, but previous work has suggested that *B. magnus* is an upland species.

**Table 3.** Results of multiple comparisons of variables from generalized linear models investigating the effects of landscape composition measured within a 700 m radius on proportions and colony densities of each species.

|                      | estimate | z-value | Pr(>|z|) | df  | Model fit |
|----------------------|----------|---------|---------|-----|-----------|
| **Proportion *B. cryptarum*** |          |         |         |     |           |
| Arable land (area)   | -0.85    | -2.94   | 0.02*   | 2   |           |
| Forestry (area)      | 0.18     | 0.58    | 0.97    | 2   |           |
| Mass-flowering crops (area) | 0.062    | -1.91   | 0.22    | 2   |           |
| Artificial surfaces (area) | -0.85   | -1.74   | 0.31    | 2   |           |
| Field boundaries (length) | -0.20   | -0.69   | 0.95    | 2   |           |
| **Colony density *B. lucorum*** |          |         |         | 5   |           |
| Arable land (area)   | -0.54    | -2.64   | 0.04*   | 2   |           |
| Forestry (area)      | -0.09    | -0.31   | 0.99    | 2   |           |
| Mass-flowering crops (area) | 0.12     | -0.69   | 0.93    | 2   |           |
| Artificial surfaces (area) | -0.38   | -1.33   | 0.54    | 2   |           |
| Field boundaries (length) | 0.11    | 0.47    | 0.98    | 2   |           |

Only models with significant components (marked with a *) are displayed. Proportion estimates are on the logit scale, and colony estimates on log scale. Model fit is calculated as follows: ((null deviance – residual deviance)/null deviance) [49].

doi:10.1371/journal.pone.0065516.t003

**Figure 3.** Mean numbers of colonies estimated foraging per field (a) and mean colony density per km$^2$ in comparison with previous work (b). Mean colony density per km$^2$ of *Bombus terrestris* was calculated using a foraging range of 758 m (n (our study) = 12, n (previous studies) = 5), and *B. lapidarius* using a foraging range of 450 m (n (our study) = 7, n (previous studies) = 12), and are provided in comparison to previous estimates (*B. lapidarius* previous estimates data from Goulson et al. [25] and Knight et al. [31]). *B. terrestris* previous estimates data from a number of sources, summarised in Charman et al., [20]). Letters indicate significant differences (p<0.05) determined using Kruskal-Wallis tests.

doi:10.1371/journal.pone.0065516.g003

**Figure 4.** Negative relationships between amount of arable land in a 700 m radius and a) proportions of *B. cryptarum* and b) colony densities of *B. lucorum*. Points show normalised measured values, and lines show model predictions.

doi:10.1371/journal.pone.0065516.g004
associated with heathlands and Ericaceous species [3,9]. The four species observed differed in colony densities and responses to surrounding landscape, suggesting the species are ecologically distinct and possibly explaining their ability to co-exist. In addition, we found that the numbers of colonies of bumblebees using resources in oilseed rape fields was particularly high. Our data indicate that *B. cryptarum* is widespread in Ireland and is certainly under-recorded due to its morphological similarity to the other cryptic species. Meanwhile, the distribution of *B. lapidarius* in agricultural areas in Ireland appears to be patchy; it was locally abundant in some fields, and absent from others.

As resources for pollinators become increasingly sparse in agricultural areas, mass-flowering crops can have positive impacts on bumblebee abundance [26,52] and on colony growth [53,54,55]. Here we also show that not only large numbers of individual bees use mass-flowering fields, but that they come from a large number of different colonies; we estimated between 648–831 colonies of the species studied using a site containing oilseed rape. Assuming all nests are located within the 700 m landscape radius measured, a colony would be located on average every 20–45 m of field boundary (although some colonies may be coming from further afield).

Bumblebees have been shown to vary their foraging distance based on the availability of resources in their environment [24,56]. We estimated significantly more colonies of *B. terrestris* per km² in our study than in previous work, suggesting that a) there is a higher background number of *B. terrestris* colonies in Ireland than in the UK and Germany where previous studies have been carried out (perhaps as the intensity of land use in Ireland is less, which can support higher densities of *B. terrestris* [57]), or b) that *B. terrestris* will fly longer distances to exploit a mass-flowering crop, therefore inflating the colony density estimates using the site. *B. terrestris* has been found to be able to fly long distances on occasion [58] and to quickly complete its colony cycle when growing next to a mass-flowering crop [55], suggesting that mass-flowering crops can be exploited by this species and at long distances. Interestingly, we found no difference between our estimates of colony density of *B. lapidarius* and previous estimates from other studies. *B. lapidarius* was only found foraging in oilseed rape at most sites, and not in the surrounding landscape. This suggests that *B. lapidarius* individuals within the vicinity of a mass-flowering resource will use that resource relatively exclusively.

Although mass-flowering oilseed rape fields can provide forage resources for a large number of bumblebee colonies in agricultural areas, not all bumblebee species will respond in the same way. As species studied here are the most abundant visitors to oilseed rape, it is possible that they may have different ecological requirements. For example, *B. cryptarum* did not respond to any landscape measures. This suggests differences among the species in terms of their foraging ranges, and that both *B. lucorum* and *B. terrestris* may utilise the landscape at larger spatial scales than measured in this study [23], or fly further distances to access a mass-flowering resource [24,56]. Therefore co-existence in these cryptic bumblebees may also be driven by spatial resource usage patterns [23].

More colonies of *B. lucorum* were also found with decreasing amount of arable land although proportions of this species were not related to this character. However, the colony density estimates of *B. terrestris* and *B. cryptarum* in this study were correlated with the number of individuals sampled; this suggests that adding more individuals to the sample would increase estimates, and therefore the lack of landscape associations with estimates for these species in particular should be interpreted with caution.

Conclusions

Although species of the *B. s. str* cryptic complex co-exist on Irish farmland, we found differences in their relative proportions and colony densities possibly related to the differential impacts of landscape on these species and *B. lapidarius*, and in turn suggesting that they may have different ecological requirements. For example, *B. cryptarum* was less common in landscapes with more arable land. This knowledge may help conservation efforts targeted to conserve this species, or may help to predict the distributions of the cryptic species which are not well known. We also found large numbers of bumblebee colonies using oilseed rape fields as a resource. This suggests that mass-flowering crops provide important forage for pollinators within agricultural areas, but it also highlights the possible severity of any negative effects of pesticides on bumblebee populations [14], and the need for sustainable management of this crop.

Supporting Information

Table S1 Summary of landscapes variables calculated surrounding each of the 14 fields.

<table>
<thead>
<tr>
<th>Landscape Variable</th>
<th>Value</th>
</tr>
</thead>
</table>
| Arable land (%)     | 70%
| Woodland (%)        | 10%
| Urban (%)           | 5%
| Agriculture (%)     | 15%

Table S2 Summary of Spearman’s rank correlations between compositional landscape variables describing landscapes within a 700 m radius from the focal oilseed rape field.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Spearman’s Rank Correlation</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>MFC</td>
<td>-0.50</td>
<td>0.006</td>
</tr>
<tr>
<td>FB</td>
<td>0.40</td>
<td>0.03</td>
</tr>
</tbody>
</table>

MFC = mass flowering crops, FB = length of field boundary.

(DOC)
Table S3 Colony density estimations for a) *Bombus terrestris*, b) *B. lucorum*, c) *B. cryptarum* and d) *B. lapidarius* at each site. \( N_{\text{obs}} \) = number of individual worker bees sampled, \( N_{\text{sib}} \) = total number of sister pairs found within the sampled individuals using COLONY analyses of microsatellite data; and in the oilseed rape field only (OS), \( N_{\text{adj}} \) = n/a. \( N_{\text{adj}} \) was estimated per km\(^2\) based on ECM estimations and foraging distances from Knight et al. [33].

<table>
<thead>
<tr>
<th>Site</th>
<th>Total Colonies</th>
<th>Adjacent Field</th>
<th>OS Field</th>
<th>OS Adjacent Field</th>
</tr>
</thead>
<tbody>
<tr>
<td>Site A</td>
<td>52</td>
<td>44</td>
<td>10</td>
<td>2</td>
</tr>
<tr>
<td>Site B</td>
<td>38</td>
<td>32</td>
<td>6</td>
<td>2</td>
</tr>
<tr>
<td>Site C</td>
<td>41</td>
<td>35</td>
<td>6</td>
<td>1</td>
</tr>
</tbody>
</table>

**References**