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National Park Core Zones Do Not Buffer Insect Pollinators Against Occupancy Declines

George Allen[1,2,3] | Rob Cooke[2] | Lucy Ridding[2] | Ben Woodcock[2] | Deepa Senapathi[1] | Joann Sy[3] | Michael Garratt[1]

[1]Department of Sustainable Land Management, University of Reading, Reading, UK | [2]UK Centre for Ecology & Hydrology, Wallingford, UK | [3]POLLINIS, Paris, France

Correspondence: George Allen (g.w.allen@pgr.reading.ac.uk)

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ABSTRACT

Aim: To assess the extent to which French national parks mitigate pollinator declines and support biodiversity outcomes across protection gradients.

Location: Eight French national parks, including core zones, buffer zones, and surrounding non-park landscapes.

Methods: We analysed 290,540 occurrence records representing 610 pollinator species (Lepidoptera, Hymenoptera and Diptera) from 2000 to 2023. Species-level occupancy trends were estimated across core, buffer and non-park zones using standardised occupancy modelling frameworks applied consistently across taxa and protection levels. We quantified temporal changes in occupancy, species richness and community turnover, and compared patterns between rare and common species as well as among major taxonomic groups.

Results: Pollinator occupancy decreased by −41.1% in core zones, −34.0% in buffer zones and −26.6% outside of national parks over the period. Declines were disproportionately driven by losses of common species, which became 'rarer' while rare species became more 'common'. Core zones harboured 27%–50% more rare species than non-park areas and reduced declines in common species richness by approximately 31%. Butterfly (205 sp.) occupancy declined more slowly in core zones (−0.62% year^{-1}) than outside of parks (−2.23% year^{-1}), whereas the occupancy of moths (324 sp.) declined more rapidly in core zones (−5.18% year^{-1}) than elsewhere.

Main Conclusions: French national parks partially mitigated pollinator declines by supporting rare species and slowing losses of some taxa, but they did not halt overall declines, even within strictly protected core zones. These findings suggest that protected areas alone are insufficient to reverse insect declines and that stronger or more targeted management interventions may be required if parks are to contribute meaningfully to pollinator conservation.

1 | Introduction

Protected areas are a cornerstone of global biodiversity conservation. Often implemented to reduce local species extinctions, mitigate habitat loss, and preserve ecosystem processes, they form the backbone of strategies to meet goals of the post-2020 Global Biodiversity Framework. This includes the '30-by-30' target to protect 30% of land and sea by 2030 (Klaimi and Ecosystems Division 2020; Convention on Biological Diversity 2022). Protected areas range widely in their objectives, along a gradient from strictly protected reserves (IUCN Category Ia) to areas that allow sustainable use of natural

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resources (Category VI) (Dudley 2008; Table S1). Numerous studies have shown that for native biodiversity, protected areas can support greater species richness, population stability and ecosystem functioning than surrounding areas (Rada et al. 2018; Brodie et al. 2023; Cooke et al. 2023; Nowakowski et al. 2023).

Among protected areas, national parks represent a distinct category of area-based conservation. Often large, multifunctional landscapes, national parks are typically designated under IUCN Categories II–V and aim to preserve natural and cultural systems while allowing for human habitation and sustainable development (UNDP, SCBD, and UNEP-WCMC 2021). Hence, national parks have multiple objectives. In Europe, national parks are legally protected landscapes that integrate biodiversity conservation, recreation, cultural heritage, and, in many cases, traditional land uses such as low-intensity farming or forestry (Republique Francaise 2021). Unlike smaller reserves or narrowly targeted protected areas, national parks often encompass diverse habitats, altitudinal gradients, and broad-scale land-use mosaics. These characteristics may allow them to support biodiversity not only within their borders but also within the surrounding landscape.

Pollinators such as bees, hoverflies and butterflies play a critical role in supporting both wild plant communities and agricultural systems (Klein et al. 2007; Fontaine et al. 2005). In common with many other insects, they have undergone wide-scale declines (Potts et al. 2010; Powney et al. 2019; van Klink et al. 2023) linked to drivers such as habitat loss, agrochemical use, climate change, invasive species, disease spillover and land-use intensification (Theodorou et al. 2020; Woodcock et al. 2016; Dalmon et al. 2021). National parks are uniquely positioned as multifunctional landscapes, to balance biodiversity conservation with other land uses such as agriculture, tourism and cultural preservation. This multifunctionality has the potential to support pollinators (through provisioning of suitable habitat, diverse floral resources and low pesticide input) or compromise their populations (through increased human disturbance, livestock grazing and habitat fragmentation) (Carrington 2024). However, there is evidence that such trade-offs come at a cost for insect pollinators. Field studies and trend modelling have revealed that insect populations in protected areas are declining at similar rates to those in unprotected landscapes (Rada et al. 2018; Cooke et al. 2023). Furthermore, insect pollinator communities inside protected areas are often no more diverse than those in urban green spaces (Baldock et al. 2015; Theodorou et al. 2025), meanwhile national parks are often subject to significant degradation and fragmentation (McCarthy et al. 2021; Rewilding Britain 2021). These findings have led to calls for more targeted conservation efforts within protected areas (Cheng and Bonebrake 2017; Delso et al. 2021). Understanding if and how national parks support pollinators is essential for designing conservation strategies that accommodate both ecological and socio-economic objectives.

Despite their prominence, the conservation role of national parks for pollinators remains understudied. Although some studies have assessed insect communities in individual parks (e.g., Rykken et al. 2014; Turley et al. 2022), broader evaluations of national park systems and their contributions to preventing pollinator declines are rare, although examples are now emerging (Kerdoncuff et al. 2024). Presumptions that national parks offer greater protection by virtue of their size and location have rarely been tested at large scale with empirical data, and it remains unclear whether national parks deliver measurable benefits for pollinator biodiversity compared to unprotected landscapes. Many of France's protected areas prioritise management for plants and vertebrates over insects, often in part due to a lack of knowledge and expertise for taxa such as wild bees, difficulty exercising decisions in buffer zones and until recently, the lack of a legally guiding red list for bees (Leandro et al. 2017; European Commission: Directorate-General for the Environment et al. 2026). Targeted management for pollinators within national parks therefore remains largely absent, aside from limits on beekeeping in core zones, bringing into question their contributions towards pollinator conservation.

This is particularly pressing given the Kunming-Montreal Global Biodiversity Framework (KM-GBF; Convention on Biological Diversity 2022), whose Target 3 commits 196 nations to protecting at least 30% of terrestrial and inland water areas by 2030 through ecologically representative, well-connected protected area systems. Insects are poorly served by this infrastructure: approximately 76% of globally assessed insect species fall short of minimum protected area coverage targets (Chowdhury et al. 2023), and only around 6% of Key Biodiversity Areas (KBAs)—the site-based planning tool underpinning Target 3—are triggered by invertebrates. Any expansion driven by 30×30 therefore risks failing the majority of insect diversity.

Target 11 of the KM-GBF provides a more direct mandate, requiring the restoration and maintenance of ecosystem services including pollination. Translating it into measurable outcomes, however, is hampered by an indicator gap: while GBF targets can drive the right actions for insect recovery, few proposed indicators are sensitive enough to detect whether those actions are working (Bladon et al. 2026), with headline metrics such as the Living Planet Index focused overwhelmingly on vertebrates and fewer than 1% of insect species assessed by the IUCN Red List (McRae et al. 2025). It is therefore crucial to establish empirically how national parks are functioning for pollinator communities, using long-term, standardised, insect-specific trends capable of evaluating whether parks can serve as effective KBAs for these taxa, such as a 'Global Butterfly Index' (Riva et al. 2026).

By using opportunistic pollinator records available on GBIF (Global Biodiversity Information Facility 2024), we test in this study whether pollinator occupancy, species richness and community composition are better maintained in national park core zones compared to buffer zones and areas outside of national park borders over the period 2000–2023. Specifically, we aim to test the following hypotheses: (1) That pollinator occupancy is declining more slowly within national parks; (2) That national parks contain more species than surrounding areas and (3) That populations within parks are more stable than those outside of them. We compare how these patterns differ between common and rare species, as well as for different taxonomic groups. Our approach offers one of the first 'park-effect', landscape-scale assessments of national park efficacy in conserving pollinators and aims to contribute to the evidence base needed to evaluate national parks as area-based tools for addressing long-term insect declines.

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2 | Materials and Methods

2.1 | National Parks and Surrounding Landscapes

Metropolitan France provides an ideal case study for studying pollinator responses to national park protected areas due to its consistent park structure and availability of opportunistic pollinator records. All eight national parks in mainland France consist of two main zones: a protected core zone (IUCN II) and a surrounding buffer zone (IUCN V) which in total cover an area of 18,572 km^2 across France. These zones allow us to test whether different levels of protection within national parks confer different outcomes for pollinators. We are not, however, conducting an impact evaluation that could address what would have happened if these national parks had not been established.

France's home territories national parks were selected for analysis (Forêts, Cevennes, Pyrénées, Vanoise, Écrins, Mercantour, Calanques and Port-Cros), representing a range of altitudes, landscape compositions, habitats and park ages. We also mapped sites outside park boundaries. Shapefiles for all national parks were obtained from the open-access French Republic repository https://www.data.gouv.fr/en/ (Republique Francaise 2024—accessed 01/12/2024). All spatial layer preparation was conducted in QGIS 3.16 (QGIS Development Team 2025).

2.2 | Generating Spatial Datasets

Across each landscape, a square grid of 2 km^2 cells was generated, extending outwards from the centroid of each national park core zone to a maximum distance of 50 km (located at each grid's far corner) (Figure 1). Each national park dataset therefore consisted of 2500 grid cells, giving a total of 20,000 cells covering an area of 40,000 km^2. Grid cells were allocated a treatment group (Core Zone; Buffer Zone; Outside Park Boundary) based on which group had the highest areal coverage within the cell. A small number of grid cells fell outside French national park boundaries but within neighbouring countries (Spain and Italy). As national park designations are country-specific, core zone and buffer zone status was assigned exclusively to grid cells located within French national parks. Grid cells outside French national park boundaries, including those falling within neighbouring countries, were therefore classified as 'outside'. Consequently, our comparisons of occupancy trends were made between French national park core zones, French national park buffer zones, and areas outside of French national parks.

2.3 | GBIF Download and Record Cleaning

Insect occurrence data for the orders Hymenoptera, Lepidoptera and Diptera were downloaded from GBIF (Global Biodiversity Information Facility 2024) and cropped to the grid area for each national park landscape. These orders of insects were selected because they include most pollinating insect taxa (e.g., bees, flies, hoverflies, butterflies, wasps, moths, etc.) (Rader et al. 2015; Ollerton 2017). We acknowledge that not all species in these families are widely considered to pollinate (e.g., Ants) (Delnevo et al. 2020), but we place emphasis on these orders as they contain the major contributors to pollination services within Western Europe (Klein et al. 2018). Henceforth we refer to the species included in this study as 'pollinators'. We included the managed species *Apis mellifera* to test whether national park

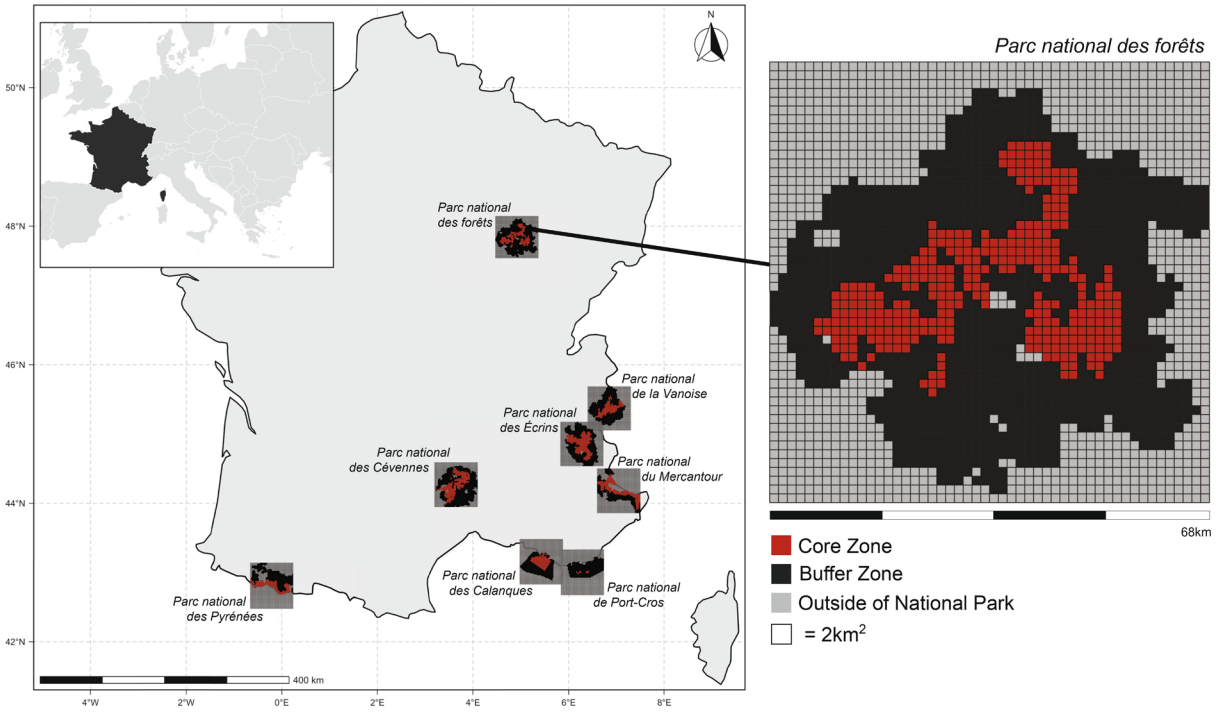


FIGURE 1 | Map of mainland France and all the national parks included in this study ($N = 8$). Coastal national parks contain island and mainland sites which benefit from the same terrestrial protected status as inland terrestrial national parks. An example of the grid layout used to conduct occupancy modelling is provided. Red grid cells indicate 2 km^2 landscapes ($N = 20{,}000$) in national park core zones (IUCN category II protection status), black cells are within buffer zones (IUCN category V) and light grey cells are areas outside of national parks.

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core zones are associated with reduced increases in honey bee occupancy over time.

Records were cleaned as per GBIF Secretariat (2022), that is, removal of fossil records, records with problematic occurrence information (incorrect country code, high coordinate uncertainty), and records outside of the time frame of interest (2000–2023). We chose this time period in order to focus on contemporary trends using the best quality data (preliminary analyses with data from the period 1980–2023 yielded poor convergence rates). Additionally, we unified 20 species that occurred under two names (pre- and post- taxonomic revision) and merged two cryptic species into one species group (Table S4). Record cleaning and quality filtering (removing species with <50 records) reduced occurrences from 1,133,263 to 290,540. Out of a total 20,000 grid cells of size 2 km^2, 8910 were retained for analysis (Table S2). Frequency of records varied in grid cells between treatments, likely due to differences in sampling effort common to unstructured citizen science datasets (Isaac and Pocock 2015). However, we found similar rates of increasing recording effort throughout time in all three zones (Figure S1), as well as higher species discovery rates in core zones, followed by buffer zones and then areas outside of parks. Relative changes in sampling effort may be considered more important than absolute numbers of records when assessing temporal trends, as relative changes can drive temporal biases between regions (Boyd et al. 2022). We therefore chose not to correct for sample sizes between zones.

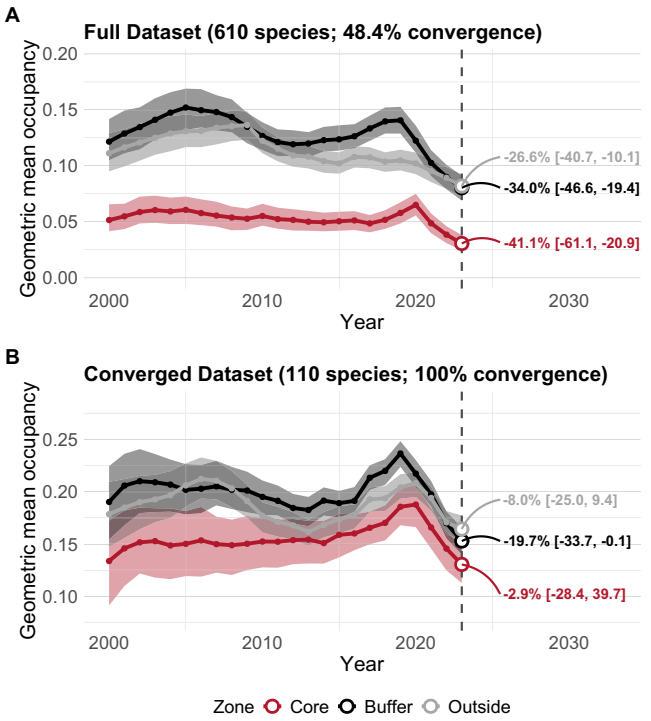


FIGURE 2 | (A, B) Trends in geometric mean species occupancy of insect pollinators across national park zones in France from 2000 to 2023. Lines show posterior median occupancy and shaded ribbons indicate 95% credible intervals for core zones (red), buffer zones (black), and areas within 50 km outside park boundaries (grey). (A) Full dataset of 610 species (48.4% model convergence; $N = 43{,}920$ species:zone:year combinations). (B) Converged dataset of 110 species (100% convergence). Open circles mark 2023 occupancy estimates, with labels indicating the plotted values and percent change from 2000 to 2023 for each zone.

2.4 | Occupancy Modelling

We fitted multi-season Bayesian hierarchical site-occupancy models (MacKenzie et al. 2002) for each species ($n = 610$) using the occDetFunc function in the sparta package (Isaac et al. 2024) in R v4.5.2 (RStudio Team 2025), following the approach of Outhwaite et al. (2019) and Cooke et al. (2023). Occupancy was estimated annually from 2000 to 2023 at a 2 km^2 grid-cell resolution, with year effects estimated separately for each zone (core, buffer, outside) using a common detection process, and a grid-cell random effect. Detection was modelled as a function of year and list length (1, 2–3, 4+ species), using records of other species to infer non-detections (pooled across all species groups) (Kéry et al. 2010). We modelled only species with ≥50 records (Pocock et al. 2019) and retained estimates meeting convergence criteria ($R_{hat} < 1.1$). Full model structure, priors and MCMC settings are provided in Section S1.

2.5 | Model Convergence and Data Filtering

To ensure robustness and transparency in our occupancy trend estimates, we generated and explored models using the full set of species: year occupancy estimates ($n = 43{,}920$; 610 species), as well as a filtered dataset including only those species: year estimates of species which fully converged in all years, as a sensitivity analysis ($n = 7920$; 110 species). Although the converged dataset may offer insights into the trajectories of a subset of species (mostly charismatic butterflies, moths and bees), we only present this result once (Figure 2B) for comparison with the full dataset (Figure 2A), which we use for all subsequent analyses. We argue that uncertain but well-sampled directionality is more informative of overall pollinator trends than the use of a well-converged subset of species which may be more subject to non-random taxonomic patterns.

For the full dataset, 48.4% of all species: year estimates converged (21,244 out of 43,920 species: year estimates). We also found that 29.6% of species converged in both 2000 and 2023 (181 out of 610 species), and 55.57% converged in at least 1 year (339 out of 610 species). Encouragingly, a relatively large number of species converged in all years and zones (18%; 110 out of 610 species).

2.6 | Species Richness, Occupancy Trends and Effect Sizes

We summarised temporal trends in pollinator occupancy for each national park zone (core, buffer, outside) using the composite geometric mean across species, following Cooke et al. (2023). For each species:zone:year we approximated the posterior occupancy distribution with a beta distribution (parameterised by its posterior mean and standard deviation) and drew 999 Monte Carlo

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samples; for every draw we calculated the geometric mean occupancy across species within each zone and year, summarising these composites as posterior medians with 95% highest-density credible intervals (HDIs; HDInterval package, Meredith and Kruschke 2018). Species richness was defined as the summed posterior mean occupancy of species per zone and year (Dorazio et al. 2006), averaged across years for the study period.

To compare zones, we calculated an absolute effect size as the median pairwise difference between zones (in period-mean composite occupancy, period-mean richness, or annual growth rate), with 95% HDIs from posterior draws (or bootstrap replicates for richness), and a relative effect size as Hedges' g with 95% confidence intervals (cohen.d function, effsize package; Torchiano 2016), where $g = 1$ indicates zones differing by one standard deviation. Monte Carlo draws of the period-mean composite occupancy and growth-rate parameter were treated as posterior samples of each.

2.7 | Rare and Common Species

Protected areas have been suggested to benefit rare and specialist invertebrate species more than common species. To test whether occupancy dynamics differed between species of contrasting rarity, we classified species as rare or common based on each species' median annual occupancy across the study period (Outhwaite et al. 2020) and repeated the same richness, trends and growth rates analysis as for the full dataset. In this context, we describe localised species as 'rare' (median occupancy across all years was in the lower quartile: occupancy ≤ 0.10; range size $\leq 4116\,km^2$; $N = 153$ species), and widespread species as 'common' (median occupancy across all years was in the upper quartile: occupancy ≥ 0.48; range size $\geq 19{,}311\,km^2$; $N = 153$ species). This split defines rarity from the quartiles of the species pool's own occupancy distribution, rather than from arbitrary limits. As only species with ≥ 50 records were modelled, 'rare' here denotes localised species among those sufficiently recorded, rather than the rarest taxa in France's fauna. To confirm that our findings were not sensitive to this threshold, we repeated the classification using a stricter 10th/90th-percentile split; the direction of rare and common trends was consistent across both definitions in each zone (Figure S2).

2.8 | Pollinator Groups

To address taxonomic biases in our data, we conducted a group-level trend analysis by subsetting species: year occupancy estimates into broad taxonomic groups. These groups were: Moths (number of species = 327); Butterflies (205); Ants (20); Hoverflies (18); Bumblebees (16); Wild Bees (12); Other Flies (4); Social Wasps (3); Solitary Wasps (2); Bee Flies (1); Honey Bees (1); and Micro Wasps (1). We emphasise here the disparity in recording effort between orders (Lepidoptera comprising > 90% of total records) and suggest that our data are more representative of Lepidoptera than any other taxonomic group. This is unsurprising given the significant bias towards butterfly recording in France and the GBIF database in general (Hughes et al. 2024; Rahimi and Jung 2025). This lends strength to our models for those taxa, but suggests that our models less well represent other important pollinating taxa (e.g., bees, hoverflies), partially reflected in the greater uncertainty associated with these groups.

2.9 | Community Composition and Temporal Change

To assess the temporal stability of pollinator communities across zones, we analysed changes in community composition using a Non-metric Multidimensional Scaling (NMDS) ordination based on Bray–Curtis dissimilarities of species occupancy estimates aggregated by year and zone. Uncertainty from the occupancy models was propagated by reconstructing beta distributions from posterior means and standard deviations and generating 300 Monte Carlo draws; NMDS ordinations were fitted to each draw and Procrustes-aligned to a reference ordination based on posterior mean occupancies to allow for comparability between repeat ordinations. Longer-term compositional change was assessed by calculating Bray–Curtis dissimilarity between initial and final communities within each zone, comparing mean occupancy across the first 3 years (2000–2002) against the last 3 years (2021–2023) to avoid sensitivity to interannual variability in any single year. This net change was further decomposed into proportional losses and gains in species-level occupancy (losses: % total decrease in occupancy from the initial to the final period; gains: % total increase in occupancy from the initial to the final period). Differences between zones in each metric were assessed as the posterior distribution of pairwise zone differences across draws, treating a contrast as credible where its 95% credible interval excluded zero. All analyses were conducted in R using the vegan package (Oksanen et al. 2020).

3 | Results

3.1 | Occupancy Trends and Growth Rates

Our final dataset, based on 290,540 pollinator records, comprised 43,920 species: year estimates for 610 pollinator species across 24 years (Table S3). These estimates spanned 8910 grid cells within three spatially explicit areas covering eight national parks, together totalling $17{,}820\,km^2$.

When considering the full dataset of species, pollinators declined in core zones (−41.1% [−61.1, −20.9]), buffer zones (−34.0% [−46.6, −19.4]) and areas outside of national parks (−26.6% [−40.7, −10.1]) over the study period (Figure 2A). Although absolute occupancy declines were lower in core zones, percentage declines were higher. Annual growth rates were negative in all zones but were more severe in core zones (−2.28%/year [−3.63, −0.77]), followed by buffer zones (−1.79%/year [−2.69, −0.93]) and areas outside of parks (−1.34%/year [−2.25, −0.46]). Pairwise comparisons of posterior growth rate distributions indicated a large effect size between outside areas and core zones (Hedges' $g = 1.55$ [1.45, 1.65]), a medium effect between outside areas and buffer zones ($g = 1.01$ [0.92, 1.10]) and a smaller effect between buffer zones and core zones ($g = 0.79$ [0.70, 0.88]). This trend followed a steady decline in core zones, marked by a peak in 2020 and then a steep decline towards 2023. For buffer zones and areas outside of parks, populations fluctuated along a declining trajectory.

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Conversely, when considering the subset of species that converged in all years ($N = 110$), we observed substantially reduced, though highly variable, declines in core zones (−3.4% [−30.5, +36.5]) compared to buffer zones (−20.5% [−34.7, −0.3]). Trends outside of parks were similarly variable, with approximately 80% of the posterior distribution below zero (−8.7% [−24.2, +10.4]) (Figure 2B). Annual growth rates mirrored these patterns across zones (core: −0.15%/year [−1.53, +1.40]; buffer: −0.99%/year [−1.83, −0.01]; outside: −0.40%/year [−1.20, +0.43]), though credible intervals for core and outside zones broadly overlapped zero, indicating considerable uncertainty in the direction of change for these zones in the converged subset. Effect sizes between zones were correspondingly smaller and less certain in this subset, reflecting the reduced species pool and wider credible intervals.

3.2 | Species Richness and Rare Versus Common Species

Buffer zones hosted the greatest species richness over the period (buffer: 215 species [213, 216]) and richness in core zones (core: 199 species [197, 200]) was no different to that of areas outside of national parks (outside: 198 species [197, 200]). Richness of rare and common species varied credibly between treatments, core zones hosting around 27%–50% more rare species on average (core: 8.99 species [8.67, 9.44]; 153 total rare species) than areas outside of parks (outside: 6.55 species [6.29, 6.83]; Hedges' $g = 14.19$ [13.74, 14.64]). Buffer zones also had high rare species richness (buffer: 8.43 species [8.10, 8.78]; Buffer − Outside $g = 12.22$ [11.84, 12.61]; Core − Buffer $g = 3.13$ [3.00, 3.26]) and national parks therefore contained more rare species than surrounding landscapes. By contrast, common species richness was lowest in core zones (89.9 species [88.9, 91.0]; 153 total common species) and highest in buffer zones (98.5 species [97.5, 99.3]; Core − Buffer $g = -17.24$ [−17.78, −16.70]), with outside areas containing a moderate richness of common species (94.0 species [93.0, 94.9]; Core − Outside $g = -8.03$ [−8.29, −7.76]; Buffer − Outside $g = 9.58$ [9.27, 9.89]) (Figure 3C). All differences in species richness between zones were credible for both rare and common species (95% credible intervals excluding zero).

Occupancy trends were markedly different for rare and common species between 2000 and 2023. Common species declined in all zones (core: −47.5% [−63.2, −26.2]; buffer: −47.7% [−62.6, −30.9]; outside: −49.6% [−63.8, −31.3]), whereas rare species occupancy trends were highly varied but indicated overall growth (core: +87.5% [−37.0, +293.5]; buffer: +94.1% [+2.3, +208.0]; outside: +129.1% [+27.4, +261]). Common species were therefore responsible for the majority of occupancy declines and these declines were no different between national parks and surrounding areas, with negligible effect sizes between all zone pairs for common species (all Hedge's $g < 0.20$; Figure 3A).

Annual growth rates mirrored these trends. Common species exhibited consistently negative growth across all zones per year (core: −2.76%/year [−4.25, −1.31]; buffer: −2.78%/year [−4.02, −1.46]; outside: −2.94%/year [−4.18, −1.52]), whereas rare species showed signs of growth in all zones (core: +2.77%/year [−1.07, +6.62]; buffer: +2.93%/year [+0.93, +5.55]; outside: +3.67%/year [+1.54, +6.09]). Rarity class was a substantially stronger predictor of occupancy trajectory than zone, with large effect sizes between rare and common species within all three zones (core: $g = 3.67$ [3.53, 3.82]; buffer: $g = 5.94$ [5.74, 6.14]; outside: $g = 6.96$ [6.73, 7.20]), and the posterior probability that rare species were declining faster than common species was near zero in all zones (≤ 0.001). Rare species population trajectories were therefore more positive than those of common species in buffer zones and outside of parks, whereas core zone trends showed the most uncertainty but were still largely positive (Figure 3D).

Species richness declined everywhere between 2000 and 2023 but differed significantly between rarity classes and zones. Rare species richness declined sharply within core zones (−60.6% [−71.0, −49.0]), compared to smaller declines in buffer zones (−25.8% [−38.0, −9.7]) and areas outside of parks (−19.4% [−34.0, −2.5]), with large effect sizes between core and both buffer ($g = 5.30$ [5.17, 5.44]) and outside zones ($g = 6.05$ [5.91, 6.20]). By contrast, common species richness declined substantially less in core zones (−21.5% [−25.9, −16.6]) than in buffer zones (−31.0% [−36.1, −26.6]; Hedges' $g = 3.89$ [3.74, 4.03]) or areas outside of parks (−31.1% [−36.2, −26.5]; $g = 3.94$ [3.79, 4.09]), whereas buffer and outside zones declined similarly ($g = 0.08$ [−0.00, 0.17]). The rare versus common contrast was large in buffer zones ($g = 0.98$ [0.92, 1.05]) and outside areas ($g = 2.14$ [2.07, 2.22]), though the core zone contrast was in the opposing direction ($g = -8.88$ [−9.09, −8.68]), reflecting the divergence between rare species richness loss and occupancy increase in the most strictly protected zone. Species rarity therefore determined richness responses to national park zones of protection, but common species declined 31% [2, 54] less in core zones than areas outside of parks (Figure 3B).

3.3 | Pollinator Groups

Taxonomic groups varied in their responses to national park zones of protection (Figure 4). The most speciose group, Moths ($N = 23{,}544$ species:zone:year combinations; 327 species), declined in occupancy across all zones, and this was most pronounced in core zones (−5.18%/year [−6.51, −3.85]). By contrast, the most well-sampled group, butterflies ($N = 14{,}760$; 205 species), declined only minorly or slightly increased in core zones (−0.62%/year [−1.76, +0.54]) but clearly declined in buffer zones (−2.54%/year [−3.30, −1.80]) and areas outside of parks (−2.23%/year [−2.89, −1.56]). Butterflies increased in occupancy from 2000 to 2019, then declined in all zones towards 2023.

We observed overall gains for the few species of hoverflies we included ($N = 1296$, 18 species) and found that core zones consistently hosted the highest occupancy of bumblebees ($N = 1152$, 16 species), with core zone occupancy substantially exceeding both buffer zones and outside areas across the study period (core: 0.052 [0.045, 0.061]; buffer: 0.016 [0.014, 0.019]; outside: 0.018 [0.015, 0.021]; Buffer versus Core: $g = -11.99$ [−12.38, −11.61]; Outside versus Core: $g = -11.39$ [−11.75, −11.02]; both $p < 0.001$). Over the period, honey bees ($N = 72$, 1 species) substantially increased in occupancy everywhere, but in 2023 honey bee occupancy was ~70% lower in core zones than in buffer zones or areas outside of parks (core: 0.05 [0.02, 0.09]; buffer: 0.18 [0.12, 0.24]; outside: 0.16 [0.11, 0.21]). Across the full study period, honey bee occupancy was consistently higher outside parks than within them (outside: 0.054 [0.049, 0.059]; buffer: 0.042 [0.038, 0.047]; core: 0.019 [0.016,

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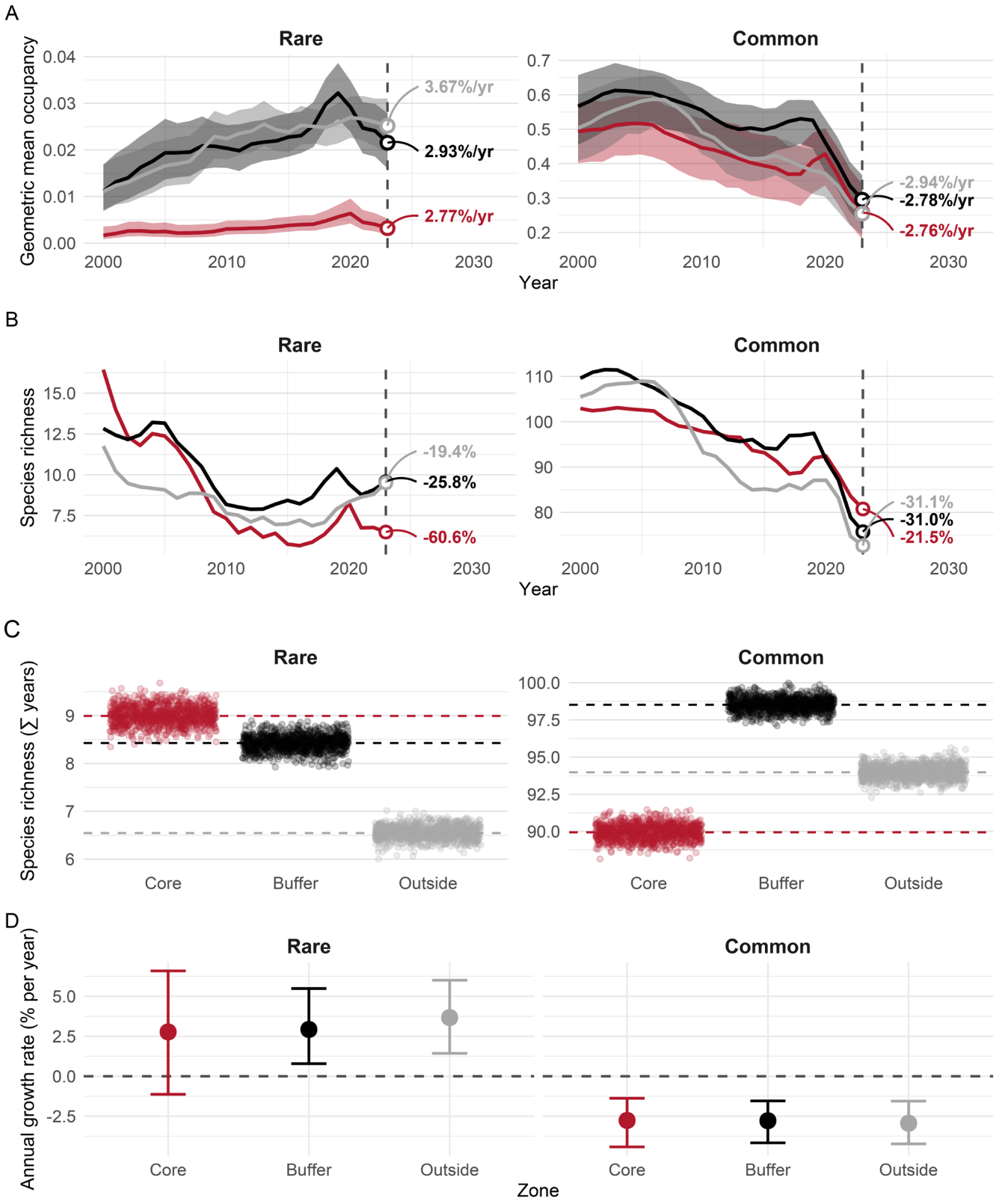


FIGURE 3 | (A–D) Trends and summaries for rare and common insect pollinators across French national park core zones (red), buffer zones (black) and areas outside of parks (grey) from 2000 to 2023. (A) Composite occupancy trajectories, calculated as the geometric mean across species within each zone and year; lines show posterior medians and ribbons show 95% credible intervals. Open circles mark the 2023 values and labels give the median annual growth rate (% year^{-1}) from 2000 to 2023 for each zone. (B) Expected species richness through time (annual sum of species' occupancy probabilities, $\Sigma\psi$) for each rarity group and zone, endpoint labels show percentage change from 2000 to 2023. (C) Total expected richness over the study period for each zone (points show Monte Carlo draws; dashed horizontal lines indicate medians). (D) Annual growth rates of composite occupancy for each rarity group and zone.

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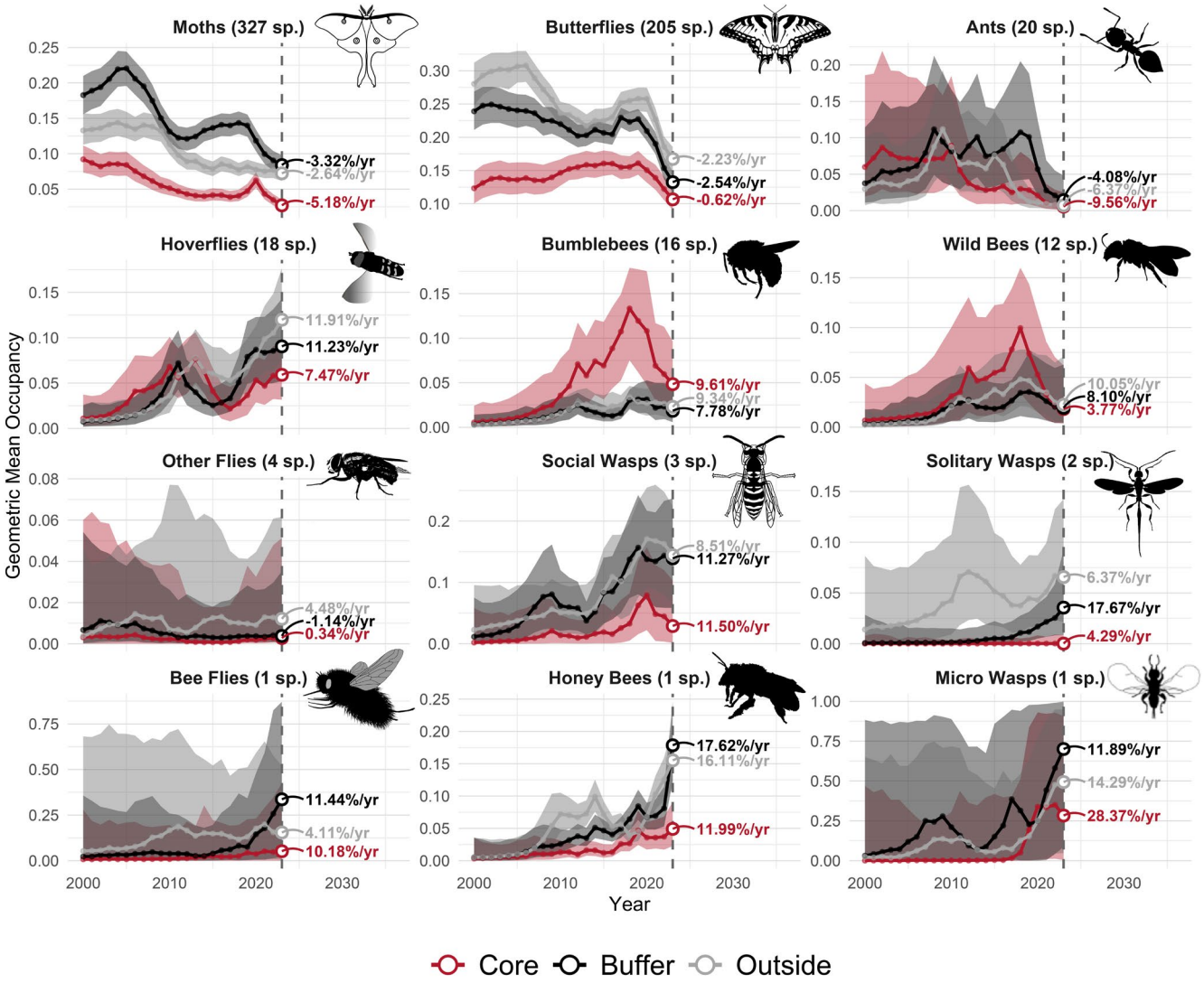


FIGURE 4 | Group-level trends in insect occupancy across French national park zones (2000–2023). Panels show composite occupancy trends for major insect pollinator groups. For each group and zone, species-level occupancy estimates were combined annually using the geometric mean to summarise multispecies occupancy. Uncertainty was propagated by reconstructing 999 beta distributions from posterior means and standard deviations and drawing repeated Monte Carlo samples for each species:zone:year combination. Solid lines show posterior median composite occupancy and shaded ribbons indicate 95% credible intervals. Vertical dashed lines mark the end of the study period (2023). Open circles indicate end-year composite occupancy values, and labels show median annual growth rate (%/year). All silhouettes were obtained from PhyloPic (www.phylopic.org) under 1.0 universal public domain dedication licence.

0.023]; Outside versus Core: $g = 15.48$ [14.99, 15.97]; Buffer versus Core: $g = 10.79$ [10.44, 11.14]). We present the remaining trends for transparency, but do not comment on them here as low species richness and high uncertainty suggest they may reflect the constituent species rather than their phylogenetic groups.

3.4 | Community Composition and Temporal Change

National park zone of protection ('Zone') explained 52.4% of community variance across all years and sampling 'Year' explained 27.1% (PERMANOVA), with zones forming distinct, well-separated clusters in NMDS ordination space (Figure 5A). When comparing mean occupancy across the first three (2000–2002) and last three (2021–2023) years, all zones experienced similar percentage community change (Bray–Curtis dissimilarity; ~31.4% [29.6, 32.9]), which was dominated by losses rather than gains (losses: ~43% [40, 45]; gains: ~9% [7, 11]) (Figure 5B,C). Compositional change did not differ detectably among zones, though gains in core zones tended to be greater than those in buffer zones. Thus, insect communities within national parks were subject to similar compositional change as those outside of their borders.

4 | Discussion

Our study contributes to a growing body of evidence for insect declines within protected areas (Hallmann et al. 2017; Rada et al. 2018; van Klink et al. 2020; Cooke et al. 2023). Crucially, these declines occurred in highly natural landscapes often considered 'wild' or 'untouched', indicating that even large, strictly

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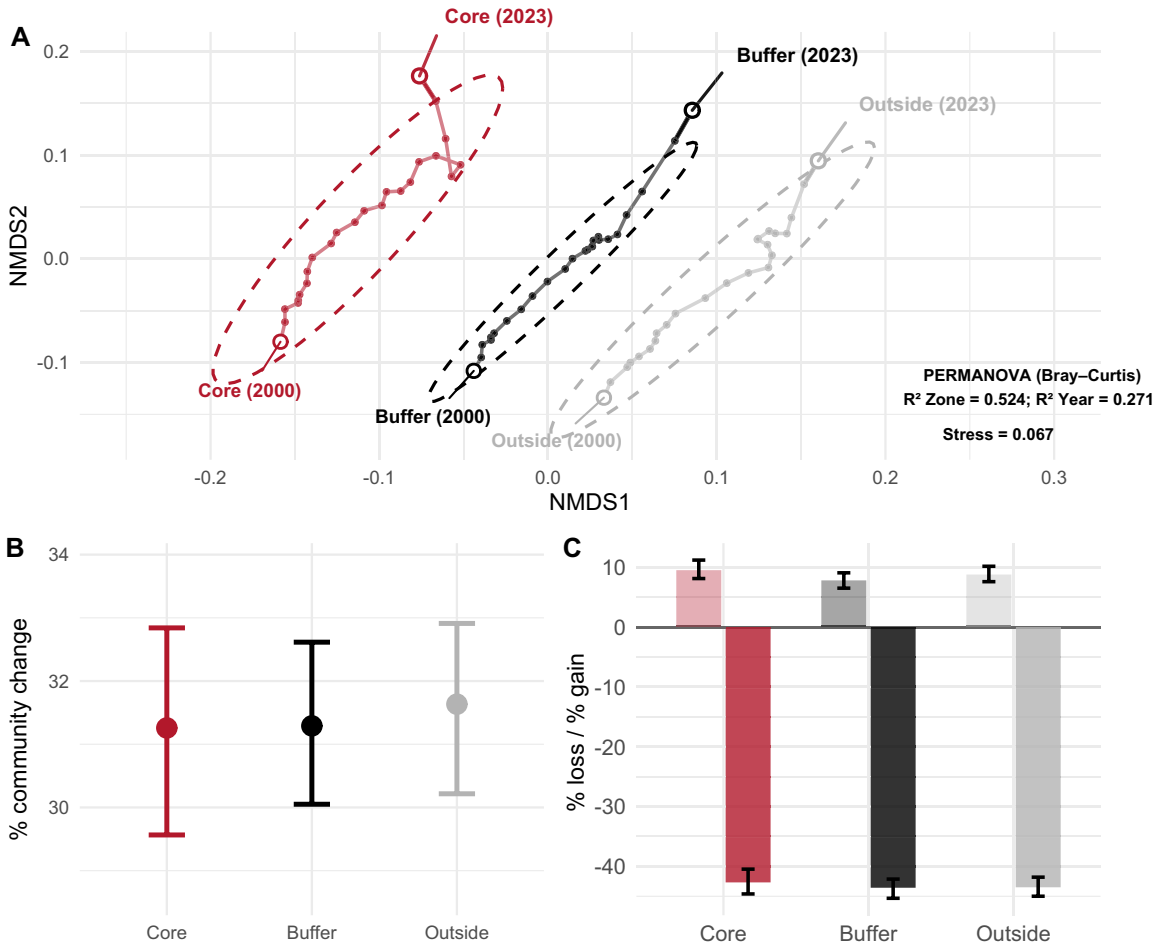


FIGURE 5 | (A–C) Community-level analysis of species: Year occupancy estimates. (A) NMDS ordination plot representing directionality of insect community change from 2000 to 2023 and dissimilarity between zones. Points represent yearly NMDS values for each zone and are connected in chronological order. Start (2000) and end (2023) positions are labelled for each zone, and dashed ellipses indicate the 95% t-ellipse around coordinates for each zone. (B) Overall community change between 2000 and 2023 within each zone, quantified as Bray–Curtis dissimilarity between start and end year. (C) Decomposition of net change into proportional loss and gain in species-level occupancy between 2000 and 2023 within each zone. Bars show posterior medians and 95% credible intervals (loss plotted as negative values, gain as positive).

protected areas cannot in themselves prevent insect declines. Our data reinforce 2006 onwards as a period of accelerating insect decline in Western Europe (Cooke et al. 2023), particularly for moths in core zones and butterflies in buffer and non-park areas, although butterfly and bumblebee occupancy increased in core zones until 2019 before falling towards 2023. Such declines likely began well before our study period (Zattara and Aizen 2021), with recent work identifying the 1930s–1960s as a period of significant butterfly loss in Switzerland (Neff et al. 2026). In aiming to deliver diverse cultural, geological and biodiversity endpoints simultaneously, national parks may therefore be failing at least one of these goals.

This has direct implications for the Kunming-Montreal Global Biodiversity Framework (Convention on Biological Diversity 2022). Target 11 explicitly names pollination as an ecosystem service requiring restoration and maintenance, whereas Target 3 commits nations to protecting 30% of land by 2030, yet if the most strictly protected landscapes cannot prevent pollinator declines, expanding areal coverage alone will deliver neither target without pollinator-specific management and monitoring within protected areas. This concern is compounded by the indicator gap raised by Bladon et al. (2026): GBF indicators, focused predominantly on vertebrates, cannot detect whether conservation actions are working for insects. The spatial planning infrastructure underpinning 30×30 therefore remains misaligned with the real-world needs of insects.

In our study, species responses were complex and difficult to generalise between groups, a finding common to multi-taxa trend studies (Outhwaite et al. 2020). Specifically, responses are likely related to the functional and behavioural strategies of different species, many of which are phylogenetically conserved within major taxonomic groups. For example, we found that butterfly trajectories were more positive within core zones until 2019, possibly reflecting reduced agricultural intensification and chemical input, and an abundance of montane habitats (Habel et al. 2019; Warren et al. 2021; Pinkert et al. 2025). Moth declines in core zones are harder to attribute to well-established causes such as light pollution and habitat degradation and instead may be a product of ubiquitous stressors such as atmospheric nitrification and elevational range shifts (Bobbink et al. 1998; Wagner et al. 2021; Neff et al. 2025). Indeed, five of the eight national parks we studied are montane and among the most intensively recorded, so the

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amplified warming and consequent upslope range shifts characteristic of high-elevation core zones, or the 'escalator to extinction' (Urban 2018), may be a substantial driver of the core zone declines we observe.

Species within different rarity classes seemed to respond consistently to national park protection. Common species declined at similar rates in all areas and benefited little from core zone protection, effectively becoming 'rarer' overall. Conversely, rare species showed fewer signs of declines overall, becoming more 'common' and increasing in overall occupancy across all zones. However, species richness metrics revealed other trends. We found that the richness of common species declined ~30% less in core zones over the period, and the richness of common species in 2023 was highest in core zones, suggesting that national parks may aid in reducing local extinctions of common species. As with rare species, this pattern was influenced by starting richness, with non-core zones hosting greater common species richness than core zones in the year 2000. By contrast, rare species richness declined more than twice as much in core zones as it did in buffer zones or areas outside of national parks, but this was in part due to core zones hosting greater rare species richness in the year 2000. Core zones may therefore fail to prevent losses of rare species richness over time. Despite this, core zones still contained 27%–50% more rare species on average than areas outside of national parks over the period, emphasising their importance for rare insect biodiversity and likely reflecting their siting within less-disturbed landscapes (Joppa and Pfaff 2009).

We expected pollinator populations in national parks to be more resilient to community reorganisation because national park designation can reduce the rate of land-use change (Mingarro and Lobo 2023), which is one of the main drivers of community homogenisation in insects (Gossner et al. 2023), but we found no evidence of this. All zones experienced ~31% total compositional change between 2000 and 2023. We also found that the directionality of this change was similar across zones, suggesting that the subset of species being reorganised between zones was similar, matching our observation that common or widespread species experienced the greatest declines (Van Dyck et al. 2009). Furthermore, our results concur with the previous findings of van Klink et al. (2023), including the reorganisation (but relatively reduced declines) of rare species and the disproportionate decline of previously abundant (common) species. This compositional turnover may reflect structured, directional change within environments that may have previously been considered stable, likely driven by processes such as climate tracking, species sorting, or ecological succession (Pecl et al. 2017; Rubenstein et al. 2023).

Core zones may promote the persistence of pollinators by enabling successful adaptive reassembly rather than by preventing change entirely (Burkle and Alarcon 2011). They may also benefit pollinators undergoing range shifts and may help prevent species declines associated with climate change (Lehikoinen et al. 2021). For butterflies, we observed similar community turnover outside of core zones, but greater net losses in areas outside of national parks, suggesting that species loss in these areas may be driven by an inability for species shifting their ranges to complete their life cycles in new habitats and fail to establish long-term populations (Kingsolver et al. 2011). This supports the 'landing-pad' hypothesis (Hiley et al. 2013), whereby protected areas act as establishment centres for species shifting their ranges.

Our results provide mixed support for the findings of Rada et al. (2018), whereby the benefits of protected areas for pollinating insects are greatest within protected areas and diminish outwards from their borders: the proportion of rare species in the overall community decreased from core zones to buffer zones to outside of parks, and butterflies declined the least in core zones, though most other taxa did not follow this pattern. We did not include 'distance from border' as a factor in our models and thus could not discern the role of core zones as a source of insect biodiversity spillover into surrounding landscapes, as has been shown for birds (Blitzer et al. 2012; Brodie et al. 2023). National parks, by virtue of their large size and high interior-to-edge ratio, may yet be more robust to edge-driven threats and the encroachment of pressures from surrounding unprotected land than smaller protected areas (Ewers and Rodrigues 2008; Coetzee et al. 2014), but further work is required to evidence this.

Our study drew on over a quarter of a million opportunistic records from eight national parks across 24 years. Although our modelling approach is designed to accommodate the biases inherent in such unstructured data, it cannot substitute for standardised monitoring such as transect walks (Warren et al. 2021). More precise estimates of decline would come from long-term structured schemes, but these do not yet exist in all parks, particularly for under-sampled taxa such as wild bees. Our $\geq$50-record threshold likewise excluded the most poorly recorded species, which likely include some of the genuinely rarest taxa; our rare-species conclusions therefore apply to localised but adequately recorded species rather than to the rarest members of France's fauna. Notably, however, the declines we report were driven overwhelmingly by common, well-recorded species, which are unaffected by this threshold, so the exclusion is unlikely to alter our central conclusions regarding pollinator decline within national parks. Relatedly, we based our analyses on the full species pool rather than the subset of species whose estimates fully converged, since convergence is tied to recording frequency and the converged subset is therefore biased towards a small number of well-sampled, atypical taxa. Our trends should accordingly be read as community-level estimates rather than as trends for the best-recorded species specifically. Continued structured monitoring in and around these parks is needed to better inform future models. Core zones, buffer zones and areas outside parks may also differ systematically in climate, topography and land cover, so occupancy trends could partly reflect environmental gradients rather than protection status, although reporting temporal trends somewhat mitigates this. Our comparisons are therefore descriptive rather than causal. Covariate matching, pairing protected and unprotected grid cells on similar environmental values (Santangeli et al. 2023), represents best practice for counterfactual assessment and would more rigorously isolate the effect of protection. We note, however, that restricting analyses to environmentally stable cells risks excluding landscapes that have undergone land-cover change, itself a primary driver of pollinator decline, potentially underestimating the breadth of decline across the network.

Finally, national parks are neither established nor managed consistently between countries, so our results may not generalise beyond France. The UK's parks (Category V) are often designated

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and managed below the threshold for strict (Category II) protection (Starnes et al. 2021), whereas Italian parks are increasingly viewed as socioeconomic assets (Romano et al. 2021). Parks worldwide span all IUCN categories (Dudley 2008) and afford differing protection, complicating cross-border assessment of efficacy. They nonetheless face common threats—tourism, recreation, underfunding and understaffing—which are likely to have similar impacts on insect biodiversity (Kubacka et al. 2022; Cox and Natali 2024; Kerdoncuff et al. 2024). Multi-country assessments would be valuable but will depend on record availability, which remains low for non-Lepidopteran groups.

Pollinators represent one of the most ecologically and economically important functional groups of insects and serve as effective indicators of broader trends in insect biodiversity. As such, studying their status within national parks simultaneously offers insights into the status of these populations and the effectiveness of national parks in preventing biodiversity loss. Although efforts to catalogue pollinator diversity in national parks are growing globally (Hinson et al. 2015; Carril et al. 2018; Meiners et al. 2019; D'Souza et al. 2021; Kang et al. 2024), we find that published inventories remain incomplete compared to the diversity of species requiring assessment (Ichter et al. 2022), and even rarer are longitudinal or 'park-effect' studies. This scarcity is troubling given Europe's position as a global leader for pollinator research (Millard et al. 2019) and the ambitions of the EU Pollinators Initiative to reverse pollinator declines by 2030 (European Commission 2023). Our study contributes to filling this gap but also reveals the limitations of relying on protected area designation alone as a method of preventing insect declines.

4.1 | Conservation Implications

Our findings suggest that area-based protection alone is insufficient to prevent large-scale pollinator declines, even within IUCN-II national park core zones. From a biogeographic perspective, spatial designation does not necessarily buffer species against landscape-scale drivers of decline, particularly for the widespread, common taxa that dominate occupancy trends: protected areas may retain rare species, but without targeted management they do not inherently stabilise pollinator distributions. Effective conservation will therefore require moving beyond static protection boundaries towards strategies that address habitat quality, floral resource continuity, disturbance pressure, and connectivity between core, buffer and non-park areas. Protected areas should be evaluated not only by extent, but by their capacity to sustain pollinator diversity over the long term, which will require standardised monitoring inside and outside parks, coupled with adaptive management, if national parks are to contribute meaningfully to reversing pollinator declines in Europe and beyond.

5 | Conclusion

Multifunctional national parks, while valuable, may not be sufficient to safeguard insect biodiversity, particularly where their mandates also encompass tourism, recreation and resource extraction. If declines persist within them, strategic changes to park design, zoning and active management will be essential, yet parks are often underfunded, and investment in pollinator management risks being overshadowed by competing priorities. We therefore urge policymakers and researchers to identify more precisely where and how national parks succeed or fall short for pollinators, as only that evidence can guide the integrated conservation strategies needed to improve the state of Europe's pollinator populations.

Author Contributions

George Allen conceived the study, conducted the analyses, and led the writing of the manuscript. Rob Cooke contributed to data analysis, modelling and manuscript development. Lucy Ridding, Ben Woodcock, Deepa Senapathi, Joann Sy and Michael Garratt contributed to study design, interpretation of results and revision of the manuscript. All authors approved the final version for publication.

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Artificial intelligence use statement: ChatGPT (OpenAI, GPT-5.2) was used in the methods stage for assistance with code editing and script organisation to support pipeline management. All analyses were executed, verified and interpreted by the authors. AI use did not generate data, influence analytical decisions, or affect results or conclusions.

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Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

All data and code supporting the findings of this study are openly available at the anonymous public data repository https://github.com/manuscriptsubmission2000/FR_Occ_v1.0.1. Occurrence data were originally obtained from the Global Biodiversity Information Facility (GBIF; https://www.gbif.org). Remaining data, code, outputs and figures will be archived in a public repository upon acceptance. Species occurrence records were obtained from the Global Biodiversity Information Facility (GBIF) and are available at https://doi.org/10.15468/dl.ntnm63. All R code used to clean the data, fit the occupancy models, and produce the analyses and figures is archived on Zenodo at https://doi.org/10.5281/zenodo.18772435 (v1.0.1).

Peer Review

For transparency, the peer review documents associated with this article are available at https://doi.org/10.1111/ddi.70283.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Figure S1:** Sampling rate versus species discovery rate in 2 km^2 sites in French core national park zones, buffer zones and surrounding areas up to 50 km from park centroids (1960–2024). (A) Sampling rate versus (B) species richness gain per grid cell over the period. More species were discovered per sample in core zones than in buffer zones or sites outside of national parks, despite a relatively uniform increase in sampling rate over time. **Figure S2:** Sensitivity of rare and common pollinator occupancy trends to the rarity-classification threshold. Annual growth rates in composite (geometric-mean) occupancy from 2000 to 2023 are shown for each national park zone (core, buffer, outside), under two rarity definitions: the quartile split used in the main analysis (25th/75th percentiles of species' median occupancy; 153 species per class, red) and a stricter decile split (10th/90th percentiles; 61 species per class, grey). Points are posterior medians and error bars 95% credible intervals across 999 Monte Carlo draws. Under both thresholds and in every zone, rare species show positive growth (increasing occupancy) and common species negative growth (declining occupancy), indicating that the contrasting trajectories of rare and common species are not sensitive to the choice of classification boundary. **Table S1:** List of IUCN protected area categories (Dudley 2008), their main objectives and their restrictions on human interaction. **Table S2:** Summary of total spatial extent covered in this study. **Table S3:** Number of Records, Species and Occupied Grid Cells for each group of pollinators. **Table S4:** Corrections applied to the problematic/resolved taxa included in our models. **Section S1:** Occupancy modelling and parameterisation.

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