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Advances in Habitat Mapping Under the Habitats Directive: Cartographic Update and Ecological Implications (Extremadura, Spain)

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Abstract

Natura 2000 habitat maps are produced under different protocols, so differences between versions may reflect methodological refinement as well as ecological change. We compared the 2005 baseline cartography of Extremadura (south-western Spain) with updates produced in 2020–2024 for six Annex I habitats (6220*, 6310, 91B0, 91E0*, 92A0 and 92D0). We assessed mapped area, subtype composition, spatial and nomenclatural correspondence and conservation status using area-weighted chi-squared tests with Crámer's V, transition matrices and intensity analysis. Mapped area differed from 21.4% decrease (6220*) *) to a 668.2% increase (91B0), and these differences were not statistically tested. Subtype composition changed substantially for 6220* (V = 0.556) but negligibly for 6310 (V = 0.053), where most "Mixed dehesa" overlapped areas previously mapped as "Holm oak". Of the 2005 extent, 71.4% (6220*) and 98.5–99.0% (riparian habitats) had no spatial match in the update. Coarse baseline mapping units may partly explain this but without documentation for the riparian updates, mapping precision, survey coverage, classification criteria and ecological change cannot be separated. For these habitats, attribution is therefore not possible, which makes documentation retrieval the first priority for monitoring. The framework shows where differences between successive maps can and cannot be read as ecological change, supporting Article 17 reporting.

Keywords: cartography; conservation; ecosystems; Europe; pastures; riparian galleries; semi-natural systems

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1. Introduction

Conserving natural habitats is essential for maintaining biodiversity and the ecosystem services that support human societies [1]. Protected areas are a key strategy for preserving and restoring biological diversity. In the European Union (EU), the Natura 2000 network (N2K) constitutes the core of biodiversity strategy and covers a substantial proportion of EU territory [1–3]. Despite progress since the implementation of the Habitats Directive (92/43/EEC), which increased the coverage of protected areas to around 18.6% of EU territory, the conservation status of these areas remains a concern. Currently, only 15% of habitats and 27% of species are assessed as favourable [4]. Regions such as Extremadura are particularly affected, with around 30.3% of the territory falling within the N2K [5],

presenting acute and extensive land management challenges [6]. Within this region, dehesas (habitat 6310) and the associated pseudo-steppe grassland (habitat 6220*) are emblematic agrosilvopastoral systems of high biodiversity and cultural value [7], while the riparian galleries (91B0, 91E0*, 92A0 and 92D0) are narrow linear habitats that support landscape connectivity and plant species richness [8]. These two groups also differ in how readily they are captured by mapping units of different precision, which makes them a useful contrast for examining cartographic effects.

The effectiveness of the N2K depends on the quality, consistency and harmonisation of the data used to assess the conservation status of habitats. Under Article 17 of the Habitats Directive, Member States must report on the status of Annex I habitats and species every six years. However, the most recent reporting cycle (2013–2018) revealed persistent gaps, most notably the absence of standardised, complete and interoperable data. This hinders comparative analysis and coordinated planning at a European scale [9]. A central source of this heterogeneity is methodological: habitat distribution has historically been mapped using expert judgement and non-harmonised protocols that vary within and between member states, which compromises the robustness and comparability of the resulting assessments [10–12]. Traditional approaches based on aerial photo interpretation and coarse vegetation mapping struggle to capture the spatial and temporal dynamics of habitats [13] and systematic, field-based validation remains limited across much of the network [14,15].

Recent cartographic updates increasingly integrate remote sensing, geographic information systems, and higher-resolution environmental data. This improves detection accuracy and reduces critical knowledge gaps in habitat distribution and dynamics [16–19]. While these advances improve inventory quality and support conservation planning, they also introduce a specific and under-examined problem: successive cartographic products for the same territory are frequently produced under different methodological protocols, classification schemes, and levels of thematic detail. Consequently, observed differences between an older and newer habitat map may reflect methodological refinement as much as, or more than, genuine ecological change [20]. Distinguishing between these two sources of apparent change is essential for using successive habitat cartographies to inform favourable conservation status assessments and favourable reference values (FRVs) [21,22], yet standardised, repeatable approaches for doing so remain scarce [23,24]. Throughout this paper, "attribution" means deciding whether an observed difference between two maps is more consistent with cartographic refinement or ecological change. Low spatial correspondence between maps, or nomenclatural change without a change in harmonised subtype composition, would be consistent with cartographic refinement. Gains or losses within areas that match spatially and are classified comparably in both maps would be consistent with ecological change. Neither pattern is conclusive on its own: attribution requires documentation of the survey protocols behind each map. Where such documentation is missing, attribution is not possible, and this limitation is itself informative for monitoring design.

This study addresses this gap by comparing the 2005 baseline habitat cartography of Extremadura with higher-resolution updates produced independently (2020–2024) for six Annex I habitats: 6220*, 6310, 91B0, 91E0*, 92A0 and 92D0. These habitats are selected because they contrast in two properties that affect comparability: how readily they are captured by mapping units of different precision (road agrosilvopastoral formations versus narrow linear riparian galleries) and whether technical documentation of the update is available (6310 and 6220* versus the four riparian habitats).

The study therefore examines whether a diagnostic of attributability can be applied under these contrasting conditions. Rather than treating the two cartographic products as equivalent snapshots of habitat extent, we use spatial overlay and analysis of composition and intensity of change to assess how nomenclature change, mapping unit precision and

survey protocol may contribute to the observed differences. The specific objectives are: (i) to describe the difference in total mapped area between products for each habitat, without a significance test because the two products use different area measures; (ii) to characterise the spatial and nomenclatural correspondence between the 2005 subtype classification and the updated classification, distinguishing reclassification from areas without cartographic precedent; (iii) to compare conservation status with these cartographic patterns where comparable data exists; and (iv) to identify, where attribution is not possible, which documentation gaps limit interpretation and how they can guide monitoring design and data collection priorities. Naturalness (2005) baseline and conservation status (updates) rely on different indicators and are not equivalent, so they are presented side by side and not as a measure of change.

The study aims to provide a replicable diagnostic framework for interpreting successive Natura 2000 habitat cartographies and identifying where differences can be interpreted as ecological change and where they cannot.

2. Materials and Methods

2.1. Study Area

The study was conducted in Extremadura, a region in south-western Spain characterised by a Mediterranean climate and extensive agrosilvopastoral dehesa systems. It is home to a substantial network of Natura 2000 sites, which were designated under the EU Habitats Directive (92/43/EEC). The study took place in the provinces of Cáceres and Badajoz. Six different types of habitats were considered, all of which are listed in Annex I of the Directive. Two of these were linked to agrosilvopastoral systems, specifically: 6220* (Pseudo-steppe with grasses and annuals of the *Thero-Brachypodietea*), which is a priority habitat, and 6310 (Dehesas with evergreen *Quercus* spp.). The other four types of habitats were riparian forests. 91B0 (thermophilous *Fraxinus angustifolia* woods); 91E0* (alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion*, *Alnion incanae*, *Salicion albae*)), a priority habitat; 92A0 (*Salix alba* and *Populus alba* galleries); and 92D0 (southern riparian galleries and thickets (*Nerio-Tamaricetea* and *Securinegion tinctoriae*)).

2.2. Data Gathering

Each habitat was examined using two cartographic sources. The 2005 baseline was obtained from the Spanish national habitat cartography, found in the data access annex ("Cartografía de Hábitats Naturales y Seminaturales de España") [25], and delivered as polygon shapefiles in the ED50/UTM zone 30N datum. The attributes included a unique polygon identifier (HAB_LAY), a habitat code (CODUE), a common name (NOM_COMUN), a generic vernacular category (GENERICO), a phytosociological alliance (ALIANZA), a naturalness index ranging from 1 to 3 (NATURALIDAD), and the percentage of the polygon area occupied by the habitat (PORCENTAJE). This followed the standard protocol of this national inventory.

Updated cartography was obtained from three independent, habitat-specific sources, all of which were delivered in the ETRS89/UTM Zone 30N datum (EPSG: 25830). For the 6310, a dedicated polygon layer was used (updated in 2020), which included the dehesa subtype (TipoDehesa) and EU conservation status (Favourable/Inadequate/Unfavourable). For the 6220*, a regional habitat geopackage was used (updated in 2024), which included the subtype and conservation status attributes. For the four riparian habitats, two complementary polygon layers (updated in 2023) covering different survey extents were combined for the updated period. The conservation status methodology for the 6310 and 6220* habitats was cross-checked against the corresponding official technical reports. The updated year, source and interval from 2005 for each habitat are summarised in Table 1.

Table 1. Update source, survey year, time interval from the 2005 baseline (Yt) and uniform annual intensity of change (St) under spatially uniform change used in the intensity analysis for each habitat. St values are from Table S1.

Habitat	Update Source	Update Year	Interval Yt (Years)	St (% yr^{-1})
6220*	Regional habitat geopackage (HIC 6220)	2024	19	2.59
6310	Dedicated dehesa polygon layer (HB6310)	2020	15	1.83
91B0	SERMA2023 + River_Comp (provisional 2023)	2023	18	3.41
91E0*	SERMA2023 + River_Comp (provisional 2023)	2023	18	3.01
92A0	SERMA2023 + River_Comp (provisional 2023)	2023	18	3.28
92D0	SERMA2023 + River_Comp (provisional 2023)	2023	18	2.81

A small number of polygons (42.0 ha, two features) lacked a valid habitat code and were excluded from all analyses. A further small proportion of updated period polygons for riparian habitats lacked a subtype attribute (91B0: 686.0 ha; 91E0*: 840.0 ha; 92A0: 231.0 ha; 92D0: 30.8 ha) and were excluded from subtype-level analysis only, while being retained in total area calculations.

The documentation status of each update was recorded as an input to the diagnosis. Here "documentation" means a technical report or metadata record describing the survey extent, the classification criteria, the mapping unit and the validation procedure.

2.3. Data Analyses

All spatial data were reprojected to a common datum (ETRS89/UTM zone 30N) prior to analysis, geometries were validated (st_make_valid) to correct topological errors arising from the harmonisation of independently produced layers. Baseline (2005) habitat area was calculated as a polygon area weighted by the PORCENTAJE attribute, rather than gross polygon area, to account for the mosaic (landscape unit) nature of the original mapping units. Updated period area was taken from the area attribute of each updated polygon (SupTesHa, SUP_HA and AREA_Ha for 6310, 6220* and the riparian habitats, respectively), i.e., full polygon area without weighting, because the updated layers delineate the habitat itself. The two riparian layers were combined by concatenation and polygons of the two layers that overlap were counted but not dissolved or clipped. Because the baseline area is a percentage-weighted estimate within landscape mosaic units whereas the updated area is a direct delineation, the two periods use different area measures. The reported area differences therefore describe differences between cartographic products and cannot be read as measured change in habitat extent. The spatial transition matrices and the intensity analysis were calculated from the geometric (unweighted) polygon areas of the intersected layers, so their absolute areas differ from those in Table 2 and area used for proportions and relative rates.

The nomenclature of subtypes differed substantially between the 2005 baseline, which comprised a mixture of vernacular and full phytosociological association names, and the updated sources, which comprised more generic vernacular categories. To statistically test for compositional change, a harmonisation dictionary was applied to group nomenclaturally distinct but conceptually equivalent categories (for example, phytosociological association names that were subsumed under their corresponding vernacular subtype). The vernacular categories follow the national cartography for the baseline [25] and the technical report of the dehesa update [26]. The dictionary is a conceptual grouping of nomenclatures, not a claim that boundaries between plant communities are sharp or invariant, vegetation varies continuously along gradients, and community boundaries are placed differently under different classification approaches, so harmonised categories are approximations (see Section 4.5).

Table 2. Mapped total area, direction and magnitude of difference between products, and results of the chi-squared test of subtype composition by survey period for each of the six habitat types.

Habitat	2005 Area (ha)	Update Area (ha)	Change (%)	χ^2	df	Cramér's V
6220*	322,364.0	253,407.0	−21.4	177,912.7	1	0.556
6310	750,900.0	1,452,229.0	93.4	6213.2	1	0.053
91B0	876.0	6729.0	668.2	n.a.	n.a.	n.a.
91E0*	1449.0	4566.0	215.1	n.a.	n.a.	n.a.
92A0	1517.0	8920.0	488.0	199.6	2	0.140
92D0	7147.0	7337.0	2.7	1096.6	3	0.275

χ^2: chi-squared statistic; df: degrees of freedom; n.a.: not applicable (fewer than two comparable subtype categories after nomenclature harmonisation). All reported chi-squared values are significant at $p < 0.001$. Area change refers to differences between products and is descriptive, chi-squared test refer to subtype composition only.

Categories representing less than 2% of the total habitat area after harmonisation were pooled into a residual 'other' class. Compositional change between periods was tested for each habitat using a chi-squared test of independence on the area-weighted subtype-by-period contingency table. Cramer's V [27] was used as an effect size independent of sample size. Hectares are not independent counts and neighbouring polygons are spatially autocorrelated, so the independence assumption of the chi-squared test is violated and the χ^2 statistics and p-value are strongly inflated by large area-based totals. The tests are therefore used as descriptive screens of compositional difference, and the interpretation rests on Cramér's V, not the p-values as formal inference.

To characterise the correspondence between the 2005 nomenclature and the updated nomenclature independently of the harmonisation used for statistical testing, a partial transition matrix was built for each habitat by intersecting the baseline and updated polygons and their non-harmonised subtype labels. Baseline areas with no spatial match in the updated layer and updated areas with no baseline precedent were reported as separate categories. The outputs were interpreted jointly as: (a) spatial correspondence, the share of the 2005 extent with no spatial match the update, indicates how far the two products delineate the habitat in the same places; (b) nomenclatural correspondence, from the transition matrices and the harmonised composition, indicates whether apparent compositional change is a change in vocabulary or of classified area; (c) documentation status determines whether attribution is possible and this is reported as the diagnostic result.

Category level intensity analysis [28] was applied to the harmonised subtype categories to test whether the annual loss or gain rates of individual subtypes departed from the rate expected under spatially uniform change (St), using the time interval between the 2005 baseline and the respective update survey (Table 1). A subtype was classified as "active" in loss (or gain) when its annualised rate exceeded St, and 'dormant' otherwise. All analyses were conducted in R (version 4.4.1) [29] using the sf, dplyr, tidyr and ggplot2 packages.

3. Results

3.1. Habitat Area Change and Typological Composition

The total mapped area for all six habitat types differed between the 2005 baseline and the most recent cartographic update, with differences ranging from a 21.4% decrease for habitat type 6220* to a 668.2% increase for habitat type 91B0 (Table 2). Habitat 6220* was the only type for which the updated survey recorded a smaller total area than in 2005 (253,407.0 ha versus 322,364.0 ha). The total area of all other habitats was higher in the updated survey, most notably for habitats 91B0, 92A0 and 91E0*, which increased by 668.2%, 488.0% and 215.1%, respectively. Habitat 92D0 showed comparatively little

change (+2.7%). These area differences are descriptive. No statistical test was applied to total area, and, because the two products use different area measures (see Section 2.3), they reflect differences between cartographic products rather than measure change. The chi-squared test of independence assessed harmonised subtype composition only. All were significant ($p < 0.001$) for habitats with more than one comparable subtype category after harmonisation, but given the large area-based tools, are interpreted though Cramér's V (Table 2).

Effect sizes (Cramér's V), which are less sensitive than *p*-values to large sample sizes, differ markedly among habitats. Habitat 6220* showed a strong effect size (V = 0.556), reflecting a substantial redistribution between its two subtypes. Perennial communities ("Majadales") accounted for the vast majority of the 6220* area in 2005, but only around half of the updated area, with a corresponding increase in annual grasslands (Figure 1A). In contrast, despite its significant increase in total area, habitat 6310 showed a negligible effect (V = 0.053), indicating that the near doubling in mapped area was not accompanied by a comparable change in subtype composition at this level of resolution. Habitats 92A0 and 92D0 showed small to moderate effects (V = 0.140 and V = 0.275, respectively). For habitats 91B0 and 91E0*, only one subtype category remained after harmonisation in each period. Therefore, no compositional test could be performed for these two habitats, and only changes in total area are reported (Figure 1B).

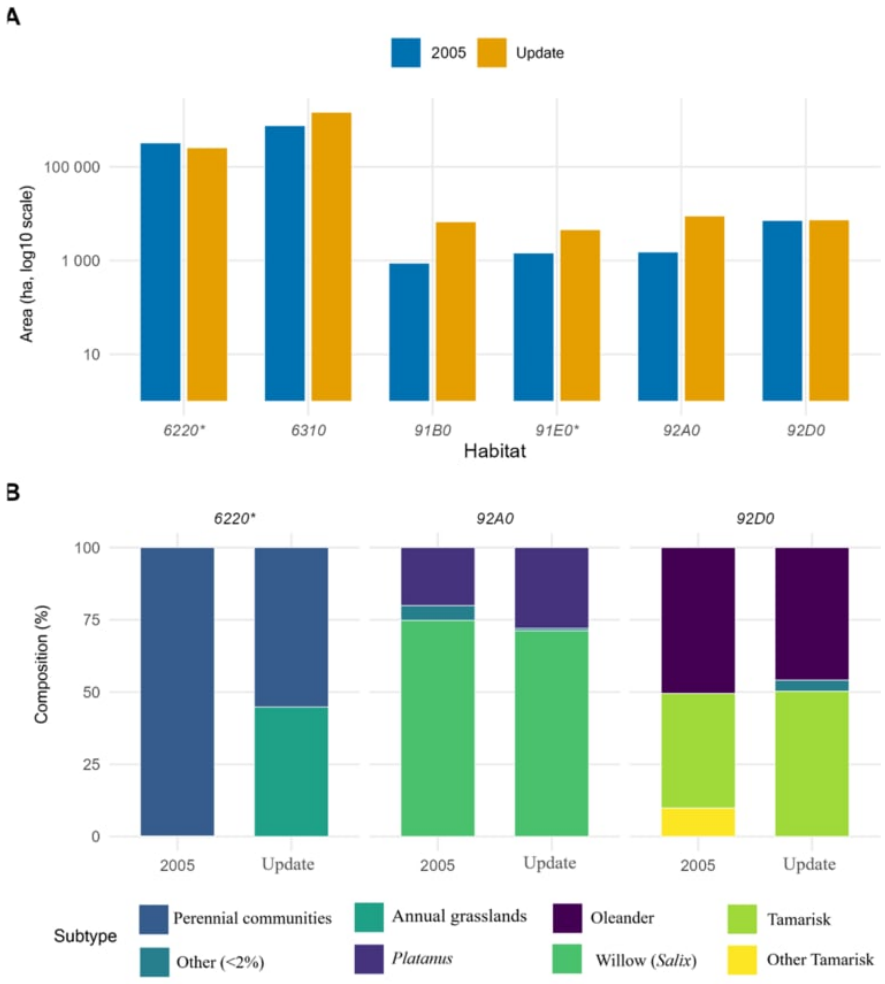


Figure 1. Habitat area and typological composition: 2005 baseline vs. cartographic update. (**A**): Total area (ha, log10 scale) by habitat and survey period. (**B**): Subtype composition (%), by survey period, for habitats 6220*, 92A0 and 92D0.

3.2. Spatial Correspondence and Nomenclature Transition of Subtypes

For 6220*, the perennial communities subtype, showed an active loss rate (4.39% yr^{-1} above the uniform rate of 2.59% yr^{-1}) combined with a dormant gain rate (0.27% yr^{-1}), consistent with a net contraction of this subtype. Annual grasslands showed active rates for both loss (5.01% yr^{-1}) and gain (5.24% yr^{-1}), indicating high spatial turnover rather than a simple one-directional change. For 6310, the single harmonised subtype "Dehesas" showed a dormant loss rate of 0.96% per year (below the uniform rate of 1.83% yr^{-1}) combined with an active gain rate of 2.14% yr^{-1}, indicating that the large increase in total area reflects net expansion rather than turnover or reclassification of existing áreas (Figure 2). The update year, interval and St for each habitat are given in Table 1 (see Section 2.2).

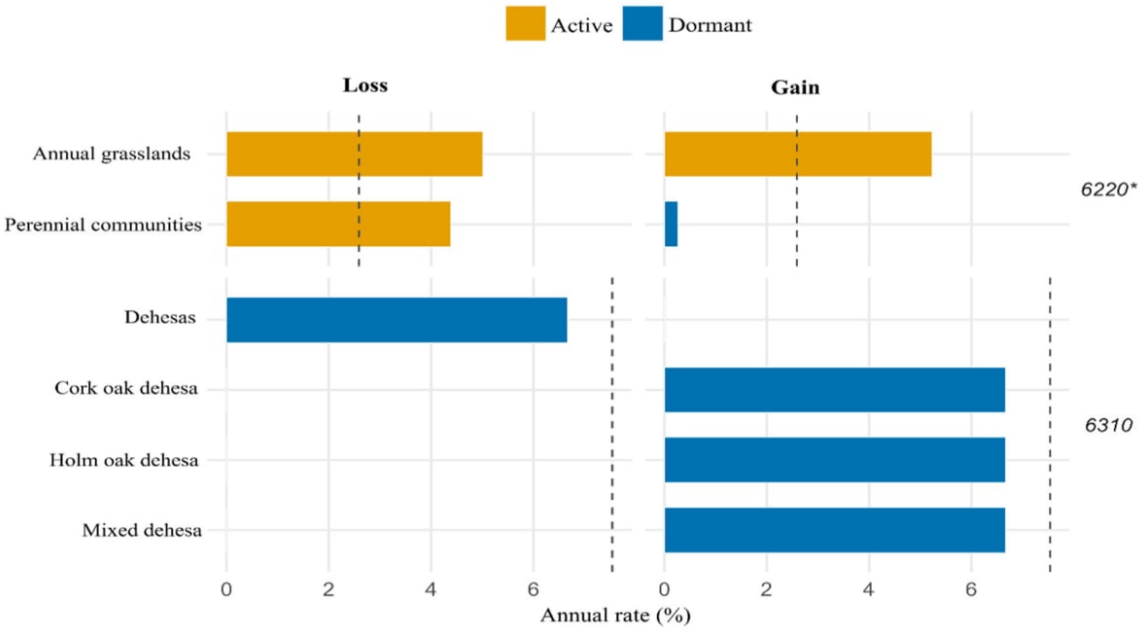


Figure 2. Category level intensity analysis [28] for habitats 6220* and 6310. Annual loss intensity (%) by subtype category and annual gain intensity (%): by subtype category. Dashed line: uniform annual intensity of change (St). Bars exceeding St are classified as active and those below as dormant. Update year, interval from 2005 (Y_t) and St: 6220*, 2024, 19 years, St= 2.59% yr^{-1}; 6310. 2020, 15 years, St = 1.83% yr^{-1} (all habitats in Table 1). Intensities were calculated from geometric polygon areas (Section 2.3).

For 6310, the transition matrix (Table 3) showed that the two 2005 subtypes, distinguished by dominant tree species ("Holm oak" = *Quercus rotundifolia* dominated; "Cork oak" = *Quercus suber* dominated), did not map one-to-one onto the three updated subtypes. Of the updated subtype "Holm oak dehesa", 67.0% corresponded spatially to the 2005 "Holm oak" subtype, while 28.1% had no 2005 precedent. Of the "Mixed dehesa" subtype, 69.7% corresponded to a 2005 subtype, the majority of which (88.3%) originated from the 2005 "Holm oak" subtype rather than the "Cork oak" subtype. The remaining 30.3% had no 2005 precedent. Specifically for the 2005 "Cork oak" subtype, 86.2% of the area with any spatial match in the update transitioned to "Mixed dehesa", while only 4.1% transitioned to "Cork oak dehesa" (Figure 2).

Table 3. Spatial transition matrix between the 2005 subtype nomenclature (NOM_COMUM field) and the updated subtype nomenclature (TipoDehesa field), restricted to transitions of more than 100 ha, for habitat 6310. Percentages refer to the 2005 total in geometric polygon area (Section 2.3).

2005 Category	Update Category	Area (ha)	% of 2005 Total
Holm oak	Holm oak dehesa	654,374.0	56.9
No 2005 precedent	Holm oak dehesa	322,944.0	28.1
Holm oak	Mixed dehesa	282,074.0	24.5
Holm oak	No update match	155,200.0	13.5
No 2005 precedent	Mixed dehesa	138,862.0	12.1
Cork oak	Mixed dehesa	37,126.0	3.2
Cork oak	No update match	9894.0	0.9
Holm oak	Cork oak dehesa	5410.0	0.5
No 2005 precedent	Cork oak dehesa	4757.0	0.4
Cork oak	Holm oak dehesa	4169.0	0.4
Cork oak	Cork oak dehesa	1787.0	0.2

The proportion of the 2005 area with no spatial match in the update ("not re-mapped") varied greatly between habitats. It was low for 6310 (14.4% of the 2005 area), but consistently high for the remaining five habitats (71.4% for 6220*; 98.5–99.0% for the four riparian habitats 91B0, 91E0*, 92A0, and 92D0). This indicates that, outside of 6310, only a minority of the mapped extent in 2005 coincides spatially with the current delineation (Table S2).

3.3. Conservation Status and Naturalness

The 2005 baseline recorded naturalness on a three-level ordinal scale (1 = lowest, 3 = highest) for all six habitats. In five of the six habitats, naturalness class 3 accounted for the largest share of the 2005 area (60.5% for 6220*, 59.1% for 6310, and 48.5–61.4% for 91B0, 91E0*, and 92D0). Habitat 92A0 was the exception, with class 3 accounting for only 4.6% of its 2005 area, while class 2 predominated (75.6%) (see Table S3a).

For the four riparian habitats (91B0, 91E0*, 92A0 and 92D0), no conservation status or naturalness field was present in the updated source data. Consequently, only the 2005 baseline distribution is reported for these habitats (Table S3a).

A directly comparable assessment for the updated period, using the same or equivalent protocol, was available only for habitat 6310. For this habitat, the updated data reported conservation status on the three-level scale of the EU Habitats Directive (Favourable, Unfavourable-Inadequate, Unfavourable-Bad). For habitat 6310, the updated distribution was Unfavourable-Bad (70.4%, 1,023,242.0 ha), Favourable (20.6%, 298,880.0 ha), and Unfavourable-Inadequate (9.0%, 130,106.0 ha). This contrasts with the 2005 baseline, in which Naturalness Class 3 predominated (59.1%) (Figure 3). Because the two scales rely on different indicators and are not equivalent, this contrast is descriptive and is not evidence of change over time.

For habitat 6220*, the updated conservation field used a five-level scale (Excellent, Good, Medium-Regular, Unfavourable, Unknown) that does not correspond to the official scale of the EU Habitats Directive protocol. The two scales were therefore not statistically compared. Using its own scale, the updated 6220* area was categorised as follows: Excellent 44.7% (113,296.0 ha), Good 32.0% (81,127.0 ha), Moderate 12.8% (32,413.0 ha), Unknown 9.2% (23,198.0 ha), Unfavourable 1.3% (3192.0 ha), with residual 0.1% (182.0 ha) recorded under the official Favourable category (Table S3b).

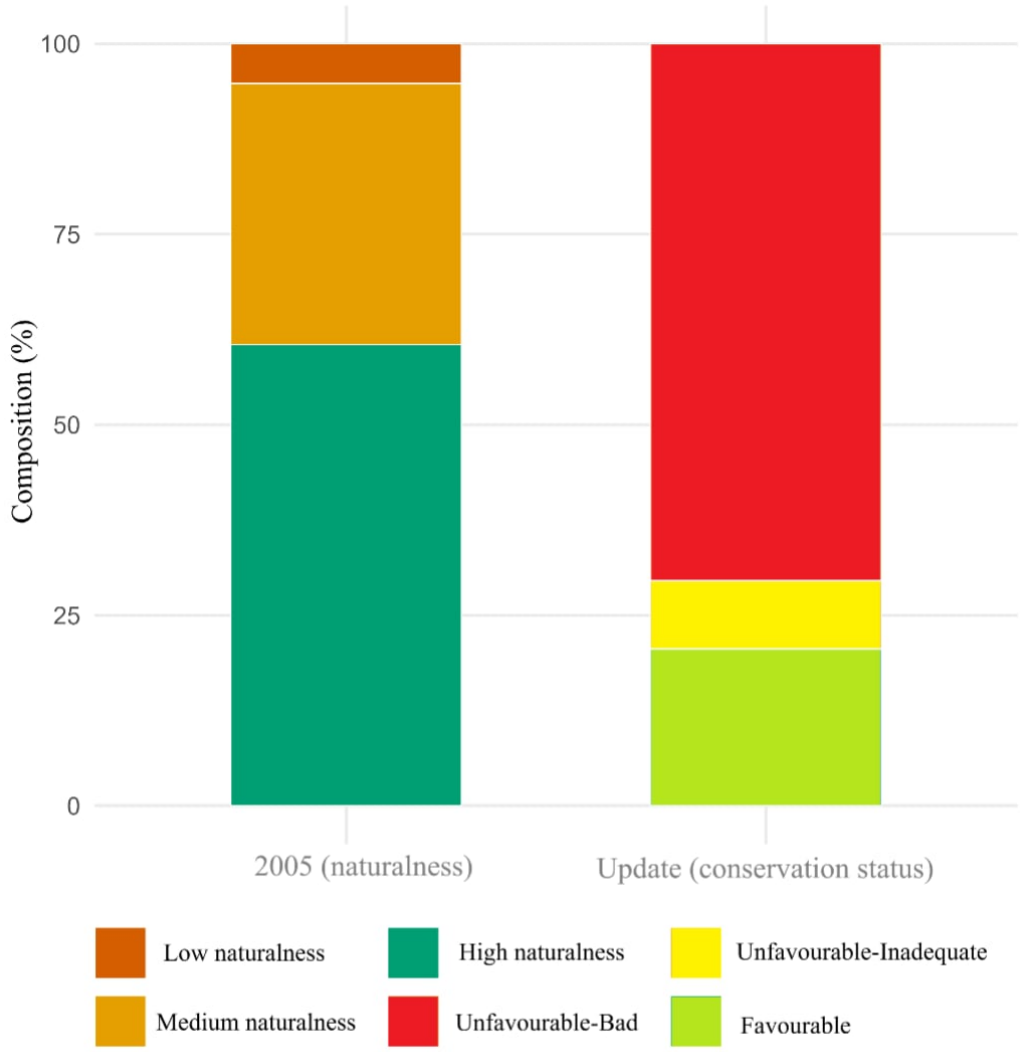


Figure 3. Conservation status and naturalness of habitat 6310: 2005 baseline versus cartographic update. The 2005 values are classified on a naturalness scale of 1–3 (low to high), while the updated values are classified on the official EU Habitats Directive scale (Favourable, Unfavourable-Inadequate, or Unfavourable-Bad). The two scales are not directly equivalent.

4. Discussion

4.1. Distinguishing Between Cartographic Refinement and Ecological Change

The clearest illustration of this distinction in our results comes from habitat 6310. The near doubling of the total mapped area was accompanied by a negligible change in subtype composition when tested using the harmonised categories (Cramer's V = 0.053). However, the fine-resolution transition matrix (Table 3) revealed that the majority of the newly delineated "Mixed dehesa" subtype corresponds spatially to areas classified in 2005 as pure "Holm oak". The available data do not establish why these areas were reclassified.

Differences in classification criteria and data source between the two products are a plausible explanation. The updated survey derived dehesa subtype and structural condition primarily from dendrometic and LiDAR-based tree population indicators rather than from a dedicated phytosociological field inventory [26], whereas the 2005 baseline consists of expert-delineated landscape mosaic polygons with an estimated percentage cover of the habitat [30]. According to phytosociological principles, a given site is usually assigned to a single climatophilous series [31,32]. A "mixed" category derived from structural or remote sensing indicators may therefore conflate genuine co-dominance with classification

uncertainty in degraded or open-canopy dehesas, where distinguishing between *Quercus rotundifolia* and *Quercus suber* climatophilus series without a field survey is known to be difficult [33–36]. However, genuine mixed stands, or real changes in tree composition since 2005, cannot be excluded without field validation. This interpretation raises broader concerns about the heterogeneity of Annex I habitat monitoring protocols across Member States and calls for science-based, field-validated vegetation classification criteria as a safeguard against this type of artefact [10–12].

A stronger compositional redistribution was observed for habitat 6220*, where the compositional shift between perennial communities and annual grasslands was statistically significant and of moderate to strong effect size ($V = 0.556$). Unlike in habitat 6310, this shift is less likely to reflect a change in classification granularity, since both subtypes were already distinguished using consistent vernacular terminology in both periods. It may reflect genuine, management-driven grassland dynamics and the rapid transformation of grazed dehesa grassland formations in Mediterranean agrossilvo pastoral systems [37–39]. However, in the absence of field verification, a contribution from interpreter-dependent classification at the perennial and annual communities boundary cannot be excluded [14], and the fact that 71.4% of the 2005 extent has no spatial match in the update limits what can be inferred.

The four riparian habitats (91B0, 91E0*, 92A0 and 92D0) present a markedly different picture. Between 98.5% and 99.0% of the mapped area in 2005 was not spatially aligned with any polygon in the updated layer, compared to 71.4% for 6220* and 14,4% for 6310. This proportion is far higher than that observed for 6310 and is unlikely to be explained solely by ecological turnover. Riparian corridors are narrow, linear features that are highly sensitive to the spatial precision of the mapping unit. The sensitivity to mapping precision is particularly concerning for riparian galleries, which function ecologically as linear corridors that support landscape-scale connectivity and plant species richness beyond their immediate footprint [8]. The 2005 baseline was mapped using coarse landscape mosaic polygons, which only provided stated percentage coverage of the target habitat [40]. This method is poorly suited to capturing such narrow, linear features. In contrast, the updated layers were delineated directly on the habitat itself [40]. Coarse baseline mapping may therefore contribute to the low spatial correspondence. This is consistent with evidence that changing from expert-delineated polygon mapping using coarse landscape mosaic to remote sensing and GIS-supported delineation can make the mapped habitat extent appear substantially different, even if there has been no change on the ground [3,15–17,41]. A natural extension of this work would be to evaluate whether the observed changes in riparian habitat extent translate into measurable changes in landscape connectivity using established connectivity indices [42,43].

Unlike 6310 and 6220*, no technical documentation of the updated survey methodology was available for riparian habitats. This limits our ability to determine how much of the discontinuity is attributable to improved spatial precision, as opposed to differences in survey extent or classification criteria between the two SERMA-derived source layers used. Consequently, the effects of mapping precision, survey coverage, classification criteria and ecological change cannot be separated for these habitats, and the coarse baseline explanation above remains a hypothesis. The diagnostic result for these four habitats is therefore that attribution is not possible without documentation of the updated survey. This result is nonetheless informative. The scale of discontinuity also underscores the value of standardised, repeatable riparian assessment protocols, such as those developed for river habitats more broadly [44] or the RFV index applied to Iberian riparian forests [45], as a basis for future comparable monitoring of these habitats in Extremadura.

4.2. Conservation Status Assessment Across Different Protocols

The conservation status results illustrate a further related problem. While the 2005 baseline recorded the same habitat as predominantly of high naturalness, the update recorded it as predominantly in an unfavourable-bad conservation status. While this could suggest a severe decline at first glance, because the two assessments use different indicators, criteria and thresholds, the comparison cannot be interpreted as a direct measurement of change over time. This is not merely a caveat regarding the quality of the data specific to this study, as the comparability of conservation status assessments across different national and regional protocols is a recognised structural weakness of Article 17 reporting under the Habitats Directive [9,21,22]. This further reinforces the need to adopt consistent Favourable Reference Values (FRVs) and assessment criteria before successive assessments can be compared as evidence of a trend. For 6220*, the finding that the updated conservation field mixed two classification scales, only one of which corresponded to the documented official protocol, further illustrates how this kind of comparison can be easily compromised by undocumented data provenance.

4.3. Implications for the Monitoring of Natura 2000 Habitats

Taken together, these results support the broader point that apparent changes between successive maps of the same territory should not be interpreted as evidence of ecological change as habitat cartography for Natura 2000 sites is progressively updated using newer technologies and protocols. Rather, it is necessary to evaluate whether these changes are, at least in part, an artefact of differing mapping unit precision, classification vocabulary, or assessment protocol. As remote sensing, machine learning and higher-resolution environmental layers become more widely used to update Annex I habitats cartography, this concern is likely to grow rather than diminish [18–20]. These methods can substantially increase mapping precision and detection capability relative to older, expert-delineated inventories, precisely the mechanism identified here for riparian habitats.

A null diagnosis is informative in three ways. First, it identifies which component of the comparison is missing. For the four riparian habitats, it is the documentation of the update that is missing, not the data themselves. The layers therefore exist and are spatially explicit, and can be overlaid with the baseline. The gap can therefore be closed at lower cost than a new survey, by retrieving documentation from the producer of the layers. Second, it presents a specific misuse. Differences in mapped area of +668.2% (91B0), +488.0% (92A0) and +215.1% (91E0*) (Table 2), together with 98.5–99.0% of the 2005 extent lacking a spatial match, should not be reported as trends in extent or as evidence of change in Article 17 assessments until attribution is possible. Third, it orders data collection priorities by the sequence in which the questions can be answered: (i) retrieve the documentation for the riparian layers and then repeat diagnosis; (ii) if the documentation shows that the update delineates the habitat itself, the change in mapping unit alone would limit comparison with the 2005 baseline, and the update would be more usefully treated as a new reference for future monitoring than as a second time point; and (iii) direct field validation to habitats where a testable hypothesis already exists, namely “Mixed dehesa” in 6310 and the perennial-annual subtype boundary in 6220*. For these two habitats where documentation was available, the same framework produced specific hypotheses that define where field effort should be placed. The framework thus acts as a triage step that separates habitats for which field validation can be designed immediately from habitats for which documentation must be obtained first.

The area composition transition framework used in this study combines a statistical test of composition change, a spatially explicit nomenclature transition matrix and category level intensity analysis [28]. It offers a replicable diagnostic for identifying where

differences between successive maps can be interpreted as ecological change and where they cannot, but it does not by itself quantify the relative contribution of cartographic differences and ecological change. It is consistent with recent calls for more rigorous science-informed and field-validated vegetation approaches to Annex I habitat classification and monitoring [11,46–48]. As only a minority of EU habitats and species are currently assessed as being in favourable conservation status [4] and the practical impact of the Habitats Directive continues to depend on the reliability of the data underpinning its implementation [49], improving the comparability of successive habitat cartographies is essential for sound conservation decision-making.

4.4. Recommendations for Future Cartographic Updates and Article 17 Reporting

The riparian case shows that the value of an updated map for monitoring depends on the metadata that accompany it. Each item below responds to a specific gap encountered in this study and is justified by it. To make successive cartographies comparable, and consistent with calls for harmonised Article 17 data [9,21], we suggest that each Annex I habitat delivered for reporting be accompanied by the metadata in Table 4 at a minimum:

Table 4. Metadata recommendations and their basis in this study.

#	Gap (Case)	Evidence	Recommendation
R1	Survey extent (riparian)	Two layers, different extents; 98.5–99.0% non-match (Sections 2.2 and 3.2)	Report survey extent and areas not re-surveyed
R2	Typology without crosswalk (6310; 91B0, 91E0*)	88.3% of “Mixed dehesa” from 2005 “Holm oak” (Table 3); no composition test possible for 91B0/91E0* (Table 2)	Report criteria and typology, with crosswalk
R3	Mapping unit and datum (riparian vs. 6310)	Mosaic vs. direct delineation; 98.5–99.0% vs. 14.4% non-match; ED50 to ETRS89 uncertainty (Sections 2.2 and 4.5)	Report mapping unit, minimum mapping unit, datum
R4	Area measures (all)	Cover-weighted vs. full area; overlaps not dissolved (Sections 2.3 and 3.1)	Report area treatment and overlap handling
R5	No field validation (all)	“Mixed dehesa” and 6220* boundary remain hypotheses (Sections 4.1 and 4.5)	Report extent and result of validation
R6	Version unrecorded (riparian; 6220*)	“Provisional 2023”, two sources (Table 1)	Provide version ID, date, production status
R7	Non-equivalent status scales (6220*, 6310, riparian)	Five-level scale vs. official; no status field in riparian (Sections 3.3 and 4.2)	Report scale, indicators, reference values; keep official values in a separate field

4.5. Limitations

There are several limitations to this study that should be noted. Firstly, the 2005 baseline and the updated layers used different geodetic datums (ED50 and ETRS89, respectively), requiring datum transformation prior to spatial analysis. Residual positional uncertainty from this transformation cannot be fully excluded. Secondly, the 2005 baseline area was estimated using a percentage cover of the target habitat within each landscape mosaic unit. This method is inherently less precise than direct habitat boundary delineation and likely contributes to the low spatial correspondence reported for several habitats. Thirdly, a small

proportion of the update period data for the riparian habitats lacked a subtype attribute or any conservation status attribute, restricting the depth of comparison possible for this group relative to 6220* and 6310. Finally, unlike the technical reports available for 6220* (not published) and 6310, there is no documented survey methodology for the riparian update layers. This means that the relative contribution of genuine ecological change and cartographic refinement could not be evaluated with the same confidence across all six habitats studied.

This study is a diagnostic of attributability for six habitats in one region. It describes how two cartographic products differ and what can be inferred from the available documentation, it does not measure ecological change and it should not be generalised to other habitats or regions without repeating diagnosis. In addition, no independent validation (field data or ground-truth assessment) was available for any habitat, so the interpretations for 6310 and 6220* are hypotheses consistent with the data and not demonstrated outcomes. The harmonisation of subtype nomenclature is a conceptual grouping and may not be invariant across classification approaches, and the chi-squared test are used descriptively because their independence assumption is not met.

Report scale, indicators, reference values; keep official values in a separate field.

Items 1–4 and 6 address the gaps that prevented attribution in this study; items 5 and 7 improve traceability and their contribution here was smaller.

A field validation design is the next main step. For 6310, polygons of "Mixed dehesa" that overlap 2005 "Holm oak" and "Cork oak" areas could be sampled in a stratified way, recording the cover or basal area of *Quercus rotundifolia* and *Quercus suber* and phytosociological revelés, to test whether these are genuine mixed stands or classification artefacts. For 6220*, revelés along the perennial-annual community boundary would test the contribution of interpreted-dependent classification. For the riparian habitats, no valid design can be proposed until documentation of the updated layers is obtained (Section 4.3). These recommendations are proposals for future work and were not tested in this study.

5. Conclusions

This study compared the 2005 baseline and updated cartography (2020–2024) for six Natura 2000 habitats in Extremadura to assess when differences between maps can be interpreted as ecological change. Mapped areas differed where it could be tested for all habitats (6220*, 6310, 92A0 and 92D0), but these differences were not tested statistically and reflect different area measures as well as possible change. Subtype composition differed substantially for 6220*, negligibly for 6310 and to a small or moderate degree for 92A0 and 92D0, with no composition test possible for 91B0 and 91E0*. The spatial analysis indicates that part of the observed differences are consistent with differences in classification and mapping units rather than habitat turnover. However, the study cannot quantify the relative contributions of cartographic differences and ecological change, and no independent field validation was available. Where conservation status data existed for the updated period (6220* and 6310), the protocols differed in indicators and criteria, so the contrast with the 2005 baseline cannot be read as direct evidence of change. For the riparian habitats, comparable data were unavailable. For these four habitats, the lack of documentation of the updated survey means that attribution is not possible. This indicates that documentation retrieval should precede any use of these differences in Article 17 reporting, and that field validation should be directed to habitats where testable hypotheses exist. Successive habitat cartographies should therefore not be compared at face value. The framework combining area composition, transition and intensity analysis can serve as a diagnostic step to identify where such comparisons can and cannot be interpreted as ecological change, and which metadata would allow them to be interpreted.

Supplementary Materials: The following supporting information can be downloaded at https://www.mdpi.com/article/10.3390/land15122295/s1.

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Data Availability Statement: The cartography of the habitats from the first survey (2005) is available on the Extremadura Habitat 2005 website (Habitat 2005 Extremadura), and the dehesa habitats are included in the updated Habitat 6310 (2020) (Habitat 6310 (2020) updated). The 6220* habitat and the riparian habitats are currently private data belonging to the Regional Government of Extremadura and the Regional Ministry of Agriculture, Livestock and Sustainable Development. They have granted us access to these habitats for the purposes of this project and study.

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