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Riparian Ecological Infrastructures: Potential for Biodiversity-Related Ecosystem Services in Mediterranean Human-Dominated Landscapes

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Abstract: Riparian Ecological Infrastructures are networks of natural and semi-natural riparian areas located in human-dominated landscapes, crucial in supporting processes that directly or indirectly benefit humans or enhance social welfare. In this study, we developed a novel multimetric index, termed Habitat Ecological Infrastructure's Diversity Index (HEIDI), to quantify the potential of Riparian Ecological Infrastructures in supporting biodiversity, and related ecosystem services, in three managed landscapes: Intensive Agriculture, Extensive Agriculture, and Forest Production. Metrics describing the structure, composition, and management of riparian vegetation and associated habitats were used to derive the potential of Riparian Ecological Infrastructures in supporting three distinct biological dispersal groups: short-range dispersers (ants), medium-range dispersers (pollinators), and long-range dispersers (birds, bats, and non-flying small mammals). The composition of floristic resources, assessed by identifying trees and shrubs at the species and genus level, and herbaceous plants at the family level, was used as a proxy to evaluate the potential of Riparian Ecological Infrastructures in promoting seed dispersal and pollination ecosystem services provided by the three biological communities. Our research evidenced that Riparian Ecological Infrastructures located in the Forest Production and Intensive Agriculture landscapes exhibited the highest and lowest potential for biodiversity-related ecosystem services, respectively. The Forest Production landscape revealed higher suitability of forage resources for short- and medium-range dispersers and a higher landscape coverage by Riparian Ecological Infrastructures, resulting in more potential to create ecological corridors and to provide ecosystem services. The Riparian Ecological Infrastructures located in the Extensive Agriculture landscape seemed to be particularly relevant for supporting long-ranges dispersers, despite providing less habitat for the biological communities. Land-use systems in the proximity of Riparian Ecological Infrastructures should be sustainably managed to promote riparian vegetation composition and structural quality, as well as the riparian width, safeguarding biodiversity, and the sustainable provision of biodiversity-related ecosystem services.

Keywords: bank vegetation; biological dispersal; ecosystem; HEIDI; land-use/land-cover; REIs

1. Introduction

Human beings have exploited natural landscapes causing biodiversity losses and depleting crucial Ecosystem Services (ES), essential for current and future generations. Floodplains and their associated riparian habitats are amongst the most biologically diverse on Earth [1–3] and have inestimable ecological, economic, and cultural values [4,5]. They are ecotones, located in the interface between aquatic and terrestrial ecosystems, encompassing the stream channel and that portion of the terrestrial landscape where vegetation may be influenced by fluctuations in the water table, flooding, and by the ability of the

soils to hold water [6]. For this reason, floodplains and their associated riparian habitats are among the most human-exploited ecosystems in Mediterranean regions, due to the high productivity resulting from the frequent floods and the subsequent highly dynamic geomorphologic processes [7,8]. A growing literature has been highlighting the relevance and challenges of biodiversity conservation in riparian habitats surrounded by human-dominated landscapes [9–11]. New research is recently emerging, reflecting concerns about the need to promote integrative and sustainable management of landscapes that reconciles human land-use with the improvement of biodiversity and biodiversity-related ES in riparian habitats [12–16]. Several national and European policies, such as the Green Infrastructure and the Biodiversity Strategy agenda, promote the preservation of natural and semi-natural landscape elements, important for biodiversity conservation and the provision of biodiversity-related ES—the commonly denominated Ecological Infrastructures (EIs) [17,18]. As highly diverse and relevant ES providers in human-dominated landscapes, riparian habitats may adopt the concept of EIs, and therefore be termed as Riparian Ecological Infrastructures (REIs).

REIs located in Mediterranean regions have been described as high-value and resilient ecosystems, although subject to a long history of human pressure [19,20]. Among the multiple impacts affecting these REIs, the conversion of floodplains to intensive agriculture is one of the most severe [21–23]. In flat agricultural landscapes of Mediterranean lowland riverine zones, REIs have either been extensively fragmented or highly modified due to competing interests by stakeholders [7,9]. Further, in agrosilvopastoral systems, characterized by large areas with low impact livestock raising, cork oak extraction, and crop production with long rotations and closed nutrient cycles [24], human management has been causing the pervasive reduction of the riparian width [7,25]. Forest production systems, on the other hand, are often allocated to middle and upstream riverine zones, where unproductive areas with steep slopes are located. Such forested ecosystems in Portugal are dominated by monospecific stands, such as blue gum eucalyptus (*Eucalyptus globulus* Labill.) and maritime pine (*Pinus pinaster* Aiton) plantations. REIs in such areas usually show alterations in floristic composition, with an increase in the number and abundance of non-native species, and reductions in the riparian width [26,27].

In Mediterranean regions, REIs are composed of complex ecosystems characterized by a high diversity and abundance of plant species [19,28]. They support higher faunistic species richness than adjacent drylands, especially when surrounded by monocultures [29]. Complex floristic-biological interactions in riparian habitats have been related to the high variability and seasonal availability of water, shelter, nesting, and forage resources, such as seeds, pollen, nectar, and fleshy fruits [2,30–32]. In addition, several vegetation attributes of riparian habitats, such as the strata complexity [33,34], the connectivity (i.e., the degree to which the landscape facilitates or impedes movement among riparian vegetation patches [1,35,36]), and the presence of microhabitats (e.g., tree hollows, deadwood trunks, leaf litter) [37,38] are also considered critical for distinct biological communities.

The protection of well-preserved REIs and the need for restoration of highly altered ones has become an essential priority to long-term environmental and human well-being sustainability [4,13,14]. The potential of REIs in supporting distinct functional biological groups has been pointed out as a valuable indicator for the evaluation of ES, such as the provision of dispersal pathways [5,17,39]. These should be considered at different spatial scales and considering different biological dispersal capabilities. Short-range dispersers, such as ants, are involved in regulating and supporting services related to soil movement, decomposition, nutrient cycling, animal community regulation, and seed dispersal [40–42]. Medium-range dispersers, such as pollinators, are responsible for pollination services, contributing to the yield, quality, and stability of important crops while also safeguarding the conservation of wild plant populations [43,44]. Long-range dispersers, such as birds and other vertebrates, including bats and non-flying small mammals, may contribute to seed dispersal [45–47] and pest suppression services [48,49].

Riparian habitats, as ES providers, have been extensively studied under a functional approach [5,15,50–52]. Nevertheless, a tool to estimate the potential of REIs as providers of biodiversity-related ES in Mediterranean human-dominated landscapes is still lacking. To the best of our knowledge, no assessment tool directly addresses the potential of REI's traits in supporting biodiversity and promoting biodiversity-related ES in Mediterranean human-dominated landscapes. Additionally, the adoption of a proxy-based solution supported by bibliographic knowledge, which uses the structure and composition of riparian vegetation as a surrogate for the abundance and diversity of animal species, may provide additional detail, or function as an alternative when field data are unavailable [53]. In this context, our study aims to:

1. Characterize the structural attributes of existing Riparian Ecological Infrastructures (REIs) in three distinct Mediterranean human-dominated landscapes: Intensive Agriculture (IA), Extensive Agriculture (EA), and Forest Production (FP);
2. Develop a new suitability metric, based on the floristic composition of riparian vegetation, and use it as a proxy to evaluate the potential of REIs in supporting seed dispersal and pollination Ecosystem Services (ES) provided by three biological dispersal groups: short-range dispersers (represented by ants), medium-range dispersers (represented by pollinators), and long-range dispersers (represented by birds and non-flying small mammals);
3. Derive a novel multimetric index, termed Habitat Ecological Infrastructure's Diversity Index (HEIDI), by integrating metrics related to the structure and management of riparian vegetation with the new suitability metric, and use it to estimate the potential of REIs in supporting biodiversity and promoting the ES provided by the three biological dispersal groups in each landscape.

We generally aim to test if the estimated potential of REIs in supporting biodiversity and related ES varies across the three landscapes and between woody and non-woody REI classes. We also aim to identify the main causes that may lead to its variability in each landscape.

2. Materials and Methods

2.1. Study Area

This study was conducted in riparian and floodplain zones of the Sorraia and Tagus rivers (Portugal) (Figure 1). The studied area is embedded in three distinct human-dominated landscapes: (i) Intensive Agriculture (IA), composed of two separated areas occupying 139.29 km² and placed in alluvial zones dominated by rice paddies and irrigated maize crops (Figure 1a,b); (ii) Extensive Agriculture (EA), covering 44.27 km² and consisting in a “montado”, i.e., an agrosilvopastoral system composed by sparse cork oak stands (*Quercus suber* L.), livestock in low densities and long rotation cereal crops (Figure 1c); and (iii) Forest Production (FP), covering 42.04 km² and composed of blue gum eucalyptus plantations intertwined with occasional maritime pine stands and near-natural cork oak forest remnants (Figure 1d).

The study area is characterized by mild winters and hot dry summers (type Csa—hot-summer Mediterranean), with frequent interannual fluctuations of precipitation [54]. Flood peaks usually occur in early winter, followed by a slow decline of flow and consequent drying during late spring and summer. The mean annual rainfall for the three landscapes is 702.1 mm and the mean annual temperature is 16.5 °C [55].

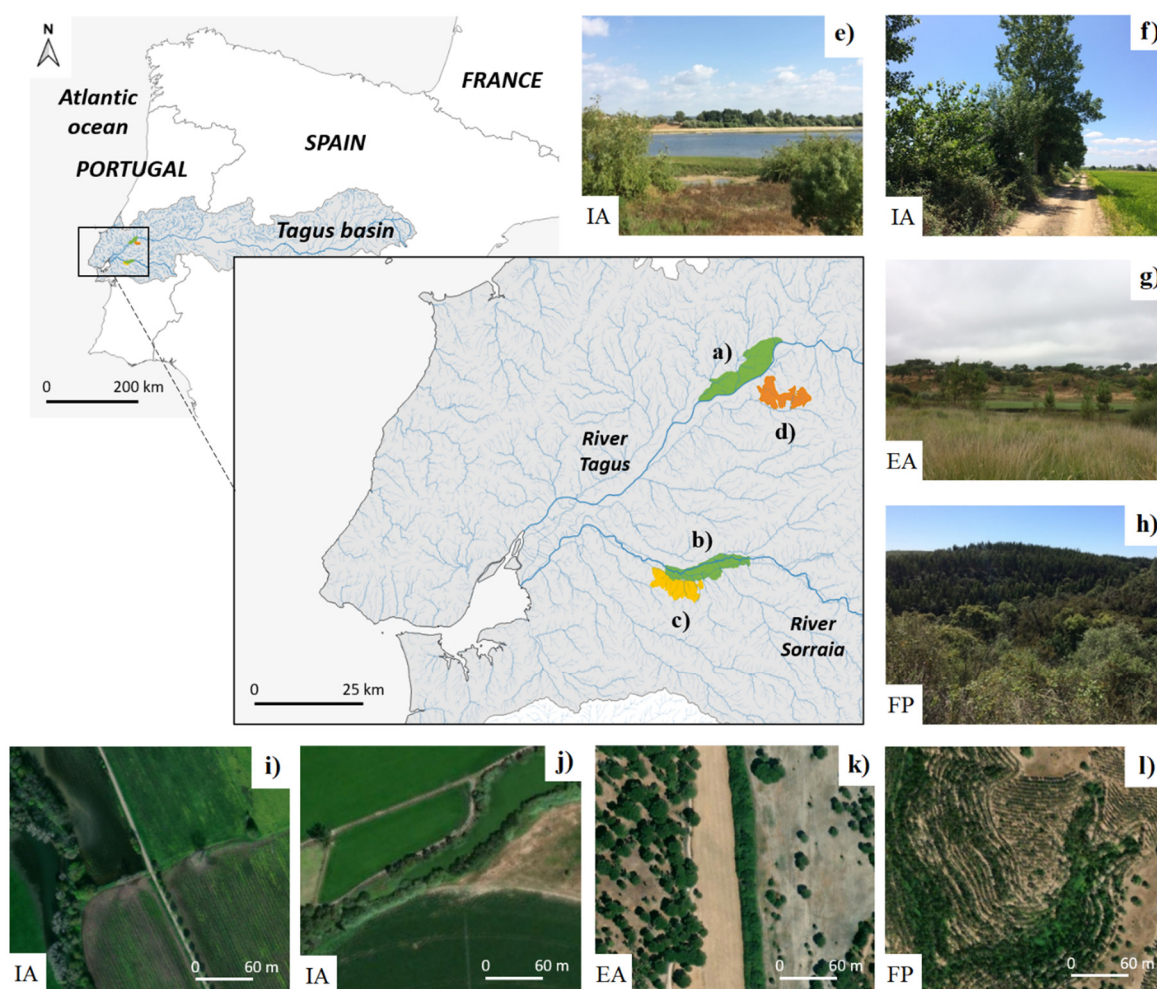


Figure 1. Geographic location of the study area (upper left panel) and the three human-dominated landscapes (central panel and photographs): (a,e,i) Intensive Agriculture (IA)—River Tagus; (b,f,j) Intensive Agriculture (IA)—River Sorraia; (c,g,k) Extensive Agriculture (EA); and (d,h,l) Forest production (FP).

2.2. Riparian Ecological Infrastructures (REIs)

The Riparian Ecological Infrastructures (REIs) are composed of remnant woody and non-woody vegetation patches, located in the surroundings of the river reaches. Woody REI patches are characterized by trees and tall shrubs, from the edge of the stream bank to the external limit of the canopy, where an abrupt change in vegetation type, height, and amount occurs [56]. Trees were considered single-stemmed woody species, with lateral branches, and including mostly micro- (2–8 m), meso- (8–30 m), macro- (30–50 m), and megaphanerophytes (>50 m) according to the Raunkiaer classification [57]. Shrubs were considered woody species branched from near the basis, usually up to 8 m, and including mostly nano- (<2 m) and microphanerophytes (2–8 m). Non-woody REI patches include open areas and are mostly dominated by low bushes and herbaceous communities.

The river reaches were initially identified using a layer of Portuguese rivers based on a 25 m resolution Digital Elevation Model. We improved the extent and detail of the river reaches using an image-based approach supported by a Geographic Information System (QGIS Version 3.4, QGIS Association, <http://www.qgis.org>, accessed on 20 September 2021) and digitized small tributaries and headwater streams over the high-resolution ESRI World Imagery layer (ArcGIS Online data, Copyright © Esri Inc., West Redlands, CA, USA), obtained in 2018, with a spatial resolution of 0.6 m (Supplementary Materials Figures S1–S4).

REI data were gathered by manually digitizing homogeneous riparian vegetation patches, at a 1:1000 scale over the high-resolution Esri World Imagery layer, using the

QGIS platform. These patches were then visually classified into woody and non-woody classes, by on-screen photo interpretation based on differences of shape, color, and texture in relation to their surroundings [56]. We selected a Minimum Mapping Unit (MMU) of 200 m², with a minimum width of 5 m, and a Minimum Gap (MG) distance among REI patches of 10 m [58]. For mapping purposes, the MMU and the MG thresholds were established to represent the minimum patch size and minimum distance between patches that are considered ecologically meaningful for the biological groups under analysis [42]. Afterward, the geographic location and the classification of the digitized REI patches were validated and reclassified, if necessary, with field surveys conducted from late Spring to early Summer of 2019.

2.3. REI's Structural Attributes

Landscape metrics were calculated to characterize the structure of REI patches [56] using the software FRAGSTATS [59] and ArcGIS Desktop 10.5 (Copyright © 1999–2016 Esri Inc.). Three metrics were selected to represent the area and the density of REI patches: (i) Number of Patches (NP), indicating the number of woody and non-woody REI patches in each landscape; (ii) Class Area (CA) (ha), representing the total area occupied by the woody and non-woody REI classes in each landscape; and (iii) Mean Patch Size (MPS) (ha), referring to the mean woody and non-woody REI patch size in each landscape. The shape of the REI patches was quantified using the Mean Shape Index (MSI), where higher values correspond to more complex shapes. The mean nearest neighbor distance between REI patches (MNN) (m) was calculated by considering the mean distance between each REI to the closest REI of the same class. A sixth metric, termed Class Coverage (CC), was created to determine the percentage of the total area occupied by the woody and non-woody REI classes in each landscape.

2.4. Field Sampling

During the field surveys, conducted from late Spring to early Summer of 2019, we also collected data on the habitat heterogeneity of REIs, using a field sheet comprised of specific metrics extracted and adapted from the Indice de Biodiversité Potentielle, developed by Larrieu and Gonin [60] (Supplementary Materials Table S1). These metrics were specifically selected since they represent key habitat features for the three biological dispersal groups under analysis. They include: (a) the number of native tree species; (b) invasive species cover (%); (c) the number of vertical strata; (d) the number of trees with microhabitats above 3 m, such as tree hollows; (e) the number of trees with microhabitats below 3 m, including cavities in the trunk and crevices in the bark; (f) the number of standing dead trees; (g) the number of deadwood trunks on the ground; (h) the number of large living trees; (i) leaf litter cover (%); (j) the number of distinct rocky habitat types; (k) the number of distinct aquatic habitat types; (l) understory clearing (%); and (m) tree clearing (%).

In addition, we also identified trees and shrubs at the species and genus level, and herbaceous plants at the family level. Their abundances were classified in the field as either “present” (<30% of covered area) or “dominant” (≥30%) for trees, and as either “isolated individuals”, “abundant” (<30%), or “dominant” (≥30%) for shrubs and herbaceous plants. All woody taxa were later classified according to their invasiveness (invasive or non-invasive), size (using the Raunkiær classification [57]), and riparian status (obligate, preferential, facultative, or non-riparian) following the classification system developed by Johnson et al. [61].

For the field sampling, we selected a balanced sub-set of randomly distributed woody and non-woody REI patches—Sampling Units (SUs)—across the three human-dominated landscapes. Field data were collected in an area corresponding to the previously described MMU in each SU (200 m²). A minimum distance of 500 m was selected between SUs to allow for adequate spatial coverage of the study area.

2.5. Habitat Ecological Infrastructure's Diversity Index (HEIDI)

Using the field data, we developed a novel multimetric index termed Habitat Ecological Infrastructure's Diversity Index (HEIDI). Five categories of REI features were taken into account: (1) Vegetation structure; (2) Vegetation habitats; (3) Associated habitats; (4) Vegetation management; and (5) Floristic suitability. The first four categories include the adapted metrics extracted from the Indice de Biodiversité Potentielle [60]. The Floristic suitability category was newly developed to evaluate the potential of floristic composition as a proxy of habitat quality and diversity for the three biological dispersal groups. Specific metrics of floristic suitability were developed for each of the biological dispersal groups, namely: (i) Seed production suitability, for short-range dispersers (represented by ants); (ii) Pollen production suitability, for medium-range dispersers (represented by pollinators); and (iii) Fruit production suitability, for long-range dispersers (represented by birds and non-flying small mammals). The development of these novel floristic composition-derived metrics was supported by extensive bibliographic research and represents a functional evaluation of habitat diversity by assessing the capacity of plant taxa to provide food resources for the three biological dispersal groups (Supplementary S1).

Seed production suitability was developed by identifying the occurrence of plant species with elaiosome-bearing seeds, i.e., lipid-rich seed appendages that attract ants and serve as rewards for dispersal [62], and by assessing the potential for myrmecochory, i.e., the dispersal of seeds by short-range dispersers. Pollen production suitability was developed by identifying plants adapted to entomophily, i.e., pollination by insects. Plants with both the production of pollen and nectar are likely to attract medium-range dispersers, represented by a wide range of pollinator groups such as bees, wasps, and syrphid flies [32]. Fruit production suitability was developed by considering plants adapted to endozoochory, i.e., dispersed by vertebrates internally [47]. These plants usually produce fleshy fruits likely to attract long-range dispersers, namely vertebrate species such as birds and non-flying small mammals [63].

2.6. HEIDI Calculation

2.6.1. HEIDI Scoring System

The HEIDI is a multimetric index and consists of a combination of several metrics with scores of “low”, “fair”, or “high”, representing an increasing contribution of REIs to habitat diversity and quality for short-, medium-, and long-range dispersers. The collected data for the Vegetation structure, Vegetation habitats, Associated habitats, and Vegetation management categories were scored according to their relevance for each biological dispersal group. For this, we used several criteria extracted from bibliographic research and expert judgment and followed the scoring method of Karr [64] (Table 1). It should be noted that, according to the consulted references, some metrics of the HEIDI have been considered as common for all biological dispersal groups, while others are exclusive of one or two groups. Further, since different faunistic groups may respond differently to the same habitat features, specific HEIDI metrics may assume different scores depending on the biological group under analysis. For example, on one hand, the amount of leaf litter may positively affect the distribution of short-range dispersers, as many ant species depend on this layer of organic matter for nesting and foraging [65,66]. On the other hand, the amount of leaf litter may have a negative effect on the distribution of medium-range dispersers, as it may prevent herbaceous species from sprouting, which can lead to a shortage of food resources [32].

Table 1. Habitat Ecological Infrastructure’s Diversity Index (HEIDI) categories, metrics, and scoring criteria associated with short-range dispersers (ants), medium-range dispersers (pollinators), and long-range dispersers (birds, bats, and non-flying small mammals).

HEIDI Categories and Metrics	Short-Range Dispersers’ Scores				Medium-Range Dispersers’ Scores				Long-Range Dispersers’ Scores			
	Low	Fair	High	References	Low	Fair	High	References	Low	Fair	High	References
1. Vegetation structure												
Native tree species (N°)	0	1 to 3	≥4	[67]	0	1 to 3	≥4	[32]	0	1 to 3	≥4	[10,68]
Invasive species cover (%)	≥30]0, 30[0	[67,69,70]	≥30]0, 30[0	[32,69,70]	≥30]0, 30[0	[68–70]
Vertical strata (N°)	1	2 or 3	4	[66,71,72]	1	2 or 3	4	[32]	1	2 or 3	4	[33,66,73]
2. Vegetation habitats												
Trees with microhabitats above 3 m (N°)									0	1 or 2	≥3	[34,37]
Trees with microhabitats below 3 m (N°)	0	1 or 2	≥3	[74]								
Standing dead trees (N°)					0	1 or 2	≥3	[75]	0	1 or 2	≥3	[34,38]
Dead wood trunks on the ground (N°)	0	1 or 2	≥3	[74,76]	0	1 or 2	≥3	[68]				
Large living trees (N°)									0	1 to 4	≥5	[33,34,38]
Leaf litter cover (%)	0]0, 50[≥50	[65,66]	≥50]0, 50[0	[32]				
3. Associated habitats												
Rocky habitat types (N°)	0	1	≥2	[66,74]								
Aquatic habitat types (N°)					0	1	≥2	[32]	0	1	≥2	[77,78]
4. Vegetation management												
Understory clearing (%)	≥60	[20, 60[<20	[66]	≥60	[20, 60[<20	[32]				
Tree clearing (%)									≥60	[20, 60[<20	[33,34,38]
5. Floristic suitability												
Seed production suitability (initial scores)	<3	[3, 16]	>16	*								
Pollen production suitability (initial scores)					<20	[20, 38]	>38	**				
Fruit production suitability (initial scores)									<10	[10, 23]	>23	***

* Supplementary S1 and Tables S2–S4; ** Supplementary S1 and Tables S5–S7; *** Supplementary S1 and Tables S8–S10.

For the new Floristic suitability category, we developed an initial scoring system (values ranging from 1 to 10) to represent the increasing potential of plant taxa as forage resource providers for each biological dispersal group. This initial scoring system allowed us to classify each plant taxa as having “very low” (0 or 1), “low” (2 or 3), “moderate” (4, 5 or 6), “high” (7 or 8), or “very high value” (9 or 10) for the biological communities under analysis (Supplementary S1 and Tables S2–S10). Then, we calculated the sum of the initial scores of all plant taxa identified within each SU, for each dispersal group, and considered that all summed scores below the first quartile, in between the first and third quartile, and higher than the third quartile would get the final HEIDI score of “low”, “fair” and “high”, respectively (Table 1).

2.6.2. HEIDI Estimation

The HEIDI value concerning the potential of each SU to support biodiversity and promote related ES was estimated by adapting an index for ordinal data with unequally weighted classes, developed by Perakis et al. [79]. This index uses the proportion of SUs with scores of “low”, “fair”, and “high” in each of the five HEIDI categories. It takes values from zero to infinity, and measures, for each SU, the degree of concentration on the scores of “high”. Since no SUs were scored with “high” on all five HEIDI categories, and since such a score would only be feasible for riparian habitats under undisturbed circumstances [80], the estimated HEIDI value of infinite should not apply to REIs.

Given the above assumptions, the HEIDI value of each SU, and for each biological dispersal group, can be estimated using the following equation, adapted from Perakis et al. [79]:

$$\text{HEIDI} = \frac{\sum_{i=1}^{\lfloor k/2 \rfloor + 1} w_i \bar{p}_i}{\sum_{i=\lfloor k/2 \rfloor + 1}^k w_{k-i+1} \bar{p}_i}$$

where k is the total number of HEIDI scoring classes ($k = 3$; “low”, “fair”, and “high”), \bar{p} is the arithmetic average of the observed proportion of HEIDI categories assuming each of the k classes (with \bar{p}_3 referring to the average proportion of HEIDI categories assuming the score “low”, \bar{p}_2 the score “fair”, and \bar{p}_1 the score “high”), and w is the weight attributed to the corresponding k class. For HEIDI categories with more than one metric, such as the Vegetation structure and Vegetation habitats, p is represented by the arithmetic average of the observed proportion of metrics assuming each of the k classes.

The weight w was calculated for each k class using the following equation, also adapted from Perakis et al. [79]:

$$w_j = 2 \left(\frac{\lfloor k/2 \rfloor - j + 2}{\lfloor k/2 \rfloor + 2} \right), j = 1, \dots, \lfloor k/2 \rfloor + 1$$

Since $k = 3$, $w_1 = 4/3$ for the HEIDI scores of “low” and “high”, and $w_2 = 2/3$ for the HEIDI score of “fair”.

A global HEIDI value, to evaluate the potential of each SU for the overall biodiversity and related ecosystem services, hereafter termed *global HEIDI*, was calculated for each landscape using the arithmetic average of the estimated HEIDI values of each biological dispersal group.

2.7. Statistical Analysis

To understand how REIs vary across the three human-dominated landscapes, both structurally (using the landscape metrics) and qualitatively (using the estimated HEIDI values), a non-parametric Kruskal–Wallis test, with 2 degrees of freedom, was applied. Dunn’s post hoc comparisons were run whenever a significant statistical difference between landscapes was found ($p < 0.05$). To understand how estimated HEIDI values vary between woody and non-woody REIs, we applied a Mann–Whitney U test. All statistical analyses were performed using the JASP software Version 0.14.1 [81].

3. Results

3.1. REI’s Structural Attributes

A total of 538, 592, and 820 REI patches were identified in the Intensive Agriculture (IA), Extensive Agriculture (EA), and Forest Production (FP) landscapes, covering a total of 441.55, 142.80, and 462.45 ha, respectively (Supplementary Materials Figures S5–S8). Nevertheless, those REI areas only represent 5.2% of the three landscapes. While the FP was the smallest study area, REI patches were more numerous and covered a larger portion of the landscape when compared to the IA and EA landscapes. Furthermore, REI woody patches were generally larger and more numerous than non-woody patches in all three landscapes (Table 2).

Irrespective of the landscape metrics, REI patches displayed a wide variety of spatial configurations, with overall significant statistical differences between the three landscapes regarding patch size, nearest neighbor and shape index (Supplementary Materials Table S11). Although REI patches form natural elongated shapes, following the trajectory of the riparian corridors, some differences are apparent in their size and spatial distribution. On one hand, woody REI patches in the IA landscape had a higher mean patch size, with a higher number of REIs with more than 10 ha (higher MPS standard deviation). On the other hand, non-woody REI patches in the IA landscape were significantly smaller and more fragmented (lower MPS and MNN) (Figure 2a). In the EA landscape, woody REI patches featured simpler spatial configurations (lower MSI) and were significantly

smaller and more fragmented (lower MPS and higher MNN) (Figure 2b), whereas in the FP landscape they were more numerous and covered more of the total landscape (higher NP and CC). As for non-woody REI patches located in the FP landscape, they were larger and in higher numbers, with overall lower fragmentation levels (higher MPS and lower MNN) and higher landscape coverage (higher CC) (Figure 2c).

Table 2. Structural attributes of woody and non-woody Riparian Ecological Infrastructures (REIs) located in the Intensive Agriculture (IA), Extensive Agriculture (EA), and Forest Production (FP) landscapes, including the Number of Patches (NP), Mean Patch Size (MPS) (\pm standard deviation), Class Area (CA), Class Coverage (CC), Mean Nearest Neighbor (MNN) (\pm standard deviation), and Mean Shape Index (MSI) (\pm standard deviation).

Landscape	River	Landscape Area (ha)	REI Class	NP	CA (ha)	MPS (ha)	CC (%)	MNN (m)	MSI
IA	Sorraia	5455.10	Woody	173	162.85	0.94 (\pm 3.45)	2.98	34.15 (\pm 58.40)	3.07 (\pm 2.38)
			Non-woody	165	30.70	0.19 (\pm 0.33)	0.56	93.54 (\pm 194.27)	2.79 (\pm 1.20)
	Tagus	8473.59	Woody	109	237.74	2.18 (\pm 8.27)	2.81	49.53 (\pm 73.03)	3.42 (\pm 2.63)
			Non-woody	91	10.26	0.11 (\pm 0.18)	0.12	243.40 (\pm 405.34)	2.30 (\pm 0.76)
EA	Sorraia	4427.40	Woody	296	78.98	0.27 (\pm 0.60)	1.78	47.37 (\pm 62.85)	2.49 (\pm 1.28)
			Non-woody	295	63.57	0.22 (\pm 0.43)	1.44	52.14 (\pm 115.69)	3.02 (\pm 1.32)
FP	Tagus	4204.00	Woody	379	283.83	0.75 (\pm 4.12)	6.75	39.57 (\pm 52.54)	3.32 (\pm 2.51)
			Non-woody	441	178.62	0.41 (\pm 0.80)	4.25	40.17 (\pm 47.60)	3.78 (\pm 2.37)

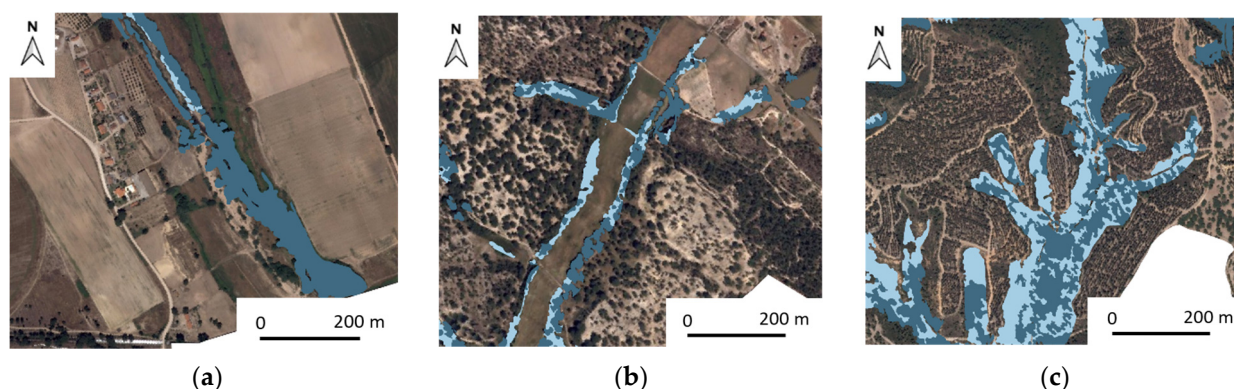


Figure 2. Illustration of the mapping of Riparian Ecological Infrastructures (REIs), with woody (dark blue polygons) and non-woody patches (light blue polygons) in (a) Intensive Agriculture, (b) Extensive Agriculture, and (c) Forest Production landscapes.

3.2. Global HEIDI Results

A total of 39, 28, and 24 SUs were surveyed in woody REIs, and 22, 18, and 20 SUs in non-woody REIs located in the IA, EA, and FP landscapes, respectively (Supplementary Materials Figures S9–S12). The highest global HEIDI value, i.e., considering all biological dispersal groups, was achieved in the FP landscape, followed by the EA and IA landscapes (Table 3). Differences between the IA and EA, and between the IA and FP landscapes were statistically significant ($p = 0.022$, $p = 0.002$, respectively). Nevertheless, when considering the biological dispersal groups separately, only short- and medium-range dispersers had significantly different HEIDI values between landscapes. Dunn's post hoc tests revealed that the habitat diversity within REI patches, for short- and medium-range dispersers, was significantly different between the IA and FP landscapes, and also between the IA and EA landscapes for short-range dispersers (Supplementary Materials Table S12).

Regarding the structural classification of REI patches, for all biological dispersal groups, higher estimated HEIDI values were associated with woody REIs. Differences between woody and non-woody REIs were statistically significant for global HEIDI values ($p < 0.001$), except for medium-range dispersers ($p = 0.167$) (Table 4).

Table 3. Habitat Ecological Infrastructure's Diversity Index (HEIDI) estimated values (mean (\pm standard deviation)) for short-, medium-, and long-range dispersers, and global HEIDI values in Sampling Units located in the Intensive Agriculture (IA), Extensive Agriculture (EA), and Forest Production (FP) landscapes, including Kruskal–Wallis test for differentiation, with 2 degrees of freedom ($H_{(2)}$).

HEIDI Estimated Values	IA	EA	FP	$H_{(2)}$	p
Short-range dispersers	0.95 (\pm 0.43)	1.26 (\pm 0.66)	1.43 (\pm 0.71)	13.89	<0.001
Medium-range dispersers	1.28 (\pm 0.60)	1.44 (\pm 0.78)	1.63 (\pm 0.75)	6.98	0.031
Long-range dispersers	1.25 (\pm 0.81)	1.61 (\pm 1.33)	1.54 (\pm 1.02)	5.51	0.064
Global	1.16 (\pm 0.45)	1.43 (\pm 0.79)	1.53 (\pm 0.69)	9.41	0.009

Table 4. Habitat Ecological Infrastructure's Diversity Index (HEIDI) estimated values (average (\pm standard deviation)) for short-, medium-, and long-range dispersers, and global HEIDI values in Sampling Units located in woody Riparian Ecological Infrastructures (REIs) and non-woody REIs, including results for the Mann–Whitney U test (W).

HEIDI Estimated Values	Woody REIs	Non-Woody REIs	W	p
Short-range dispersers	1.40 (\pm 0.60)	0.86 (\pm 0.52)	1097.50	<0.001
Medium-range dispersers	1.46 (\pm 0.67)	1.37 (\pm 0.77)	2368.00	0.167
Long-range dispersers	1.72 (\pm 1.25)	1.03 (\pm 0.40)	1266.00	<0.001
Global	1.53 (\pm 0.68)	1.09 (\pm 0.51)	1372.00	<0.001

3.3. HEIDI Results by Category

3.3.1. Vegetation Structure, Vegetation Habitats, Associated Habitats, and Vegetation Management

The proportion of SUs with HEIDI scores of “low”, “fair”, and “high” was distinct for all HEIDI categories, across all biological dispersal groups, and all landscapes (Figure 3). In the Vegetation structure category, SUs in the EA landscape showed a higher proportion of “high” scores. These SUs showed a reduced cover of invasive species and a higher number of native tree species when compared to the IA and FP landscapes (Table 5). Ash (*Fraxinus angustifolia* Vahl), grey-willow (*Salix atrocinerea* Brot.), and cork oak were the most frequent native tree species found in SUs located in the IA, EA, and FP landscapes, respectively (Supplementary Materials Table S13). Invasive species were mostly represented by giant reed (*Arundo donax* L.) in the IA and EA landscapes and bushy needlewood (*Hakea sericea* Schrad. and J.C. Wendl.) in the FP landscape. The proportion of SUs with a higher number of vertical strata was greater in the IA, followed by the FP and EA landscapes (Table 5).

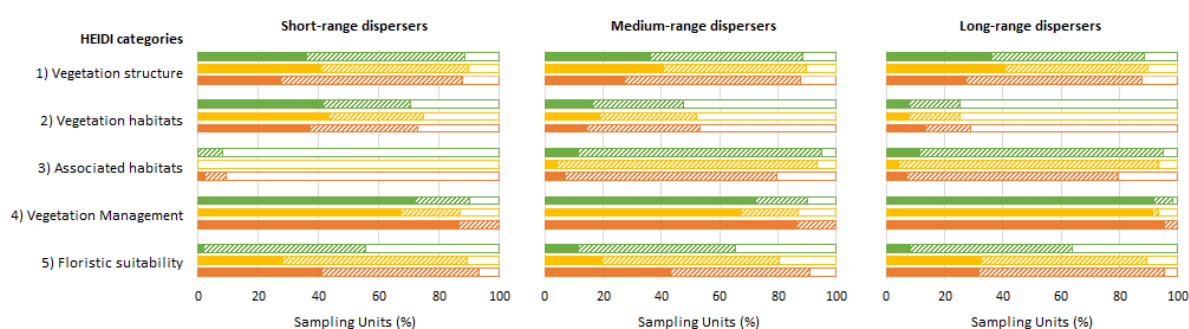


Figure 3. Average proportion of Sampling Units (%), by HEIDI category, with scores of “high” (fill), “fair” (line fill), and “low” (no fill) for each biological dispersal group (Short-, Medium-, and Long-range dispersers) in the Intensive Agriculture (green), Extensive Agriculture (golden) and Forest Production (orange) landscapes.

Table 5. Proportion of Sampling Units (%) with scores of “low”, “fair”, and “high”, per HEIDI metric, in the Intensive Agriculture (IA), Extensive Agriculture (EA), and Forest Production (FP) landscapes.

HEIDI Categories and Metrics	IA			EA			FP		
	Low	Fair	High	Low	Fair	High	Low	Fair	High
1. Vegetation structure									
Native tree species	16.39	77.05	6.56	17.39	69.57	13.04	13.64	86.36	0.00
Invasive species cover	9.84	22.95	67.21	2.17	8.70	89.13	13.64	27.27	59.09
Vertical strata	8.20	57.38	34.43	10.87	69.57	19.57	9.09	68.18	22.73
2. Vegetation habitats									
Trees with microhabitats above 3 m	77.05	13.11	9.84	67.39	13.04	19.57	56.82	18.18	25.00
Trees with microhabitats below 3 m	34.43	4.92	60.66	32.61	10.87	56.52	40.91	11.36	47.73
Standing dead trees	86.89	11.48	1.64	78.26	17.39	4.35	77.27	20.45	2.27
Dead wood trunks on the ground	44.26	18.03	37.70	34.78	21.74	43.48	34.09	31.82	34.09
Large living trees	60.66	27.87	11.48	78.26	21.74	0.00	79.55	6.82	13.64
Leaf litter cover (short-range)	9.84	63.93	26.23	8.70	60.87	30.43	6.82	63.64	29.55
Leaf litter cover (medium-range)	26.23	63.93	9.84	30.43	60.87	8.70	29.55	63.64	6.82
3. Associated habitats									
Rocky habitat types	91.80	8.20	0.00	100.00	0.00	0.00	90.91	6.82	2.27
Aquatic habitat types	4.92	83.61	11.48	6.52	89.13	4.35	20.45	72.73	6.82
4. Vegetation management									
Understory clearing	9.84	18.03	72.13	13.04	19.57	67.39	0.00	13.64	86.36
Tree clearing	1.64	6.56	91.80	6.52	2.17	91.30	0.00	4.55	95.45
5. Floristic suitability									
Seed production suitability	44.26	54.10	1.64	10.87	60.87	28.26	6.82	52.27	40.91
Pollen production suitability	34.43	54.10	11.48	19.57	60.87	19.57	9.09	47.73	43.18
Fruit production suitability	36.07	55.74	8.20	10.87	56.52	32.61	4.55	63.64	31.82

Regarding the Vegetation habitats category, Sus in the EA landscape displayed an overall higher proportion of Sus with HEIDI scores of “high”, except for long-range dispersers, where the FP landscape was favored (Figure 3). On one hand, the number of trees with microhabitats above 3 m was higher in Sus located in the EA and FP landscapes. On the other hand, the IA landscape showed a higher number of trees with microhabitats below 3 m. As for the number of deadwood trunks on the ground and standing dead trees, Sus located in the EA landscape showed a higher proportion of “high” scores. Concerning the number of large living trees, a higher proportion of Sus with five or more trees was observed in the IA landscape (Table 5).

For the Associated habitats category, the diversity of habitat types was higher in Sus located in the IA landscape for medium- and long-range dispersers, and in the FP landscape for short-range dispersers (Figure 3). Nonetheless, rocky habitats were rare in all landscapes (Table 5). For the Vegetation management category, we observed that Sus located in the EA landscape showed a slightly higher management activity regarding tree and understory clearing for all biological dispersal groups (Figure 3).

3.3.2. Floristic Suitability

Overall, we have identified a total of 28 tree and 27 shrub taxa from 23 families, and 44 herbaceous families (Supplementary Materials Tables S13–S15). The highest potential of floristic composition to support the biological communities was achieved in the FP landscape (Figure 3). However, Sus in the IA landscape presented higher overall plant richness ($n = 66$), especially in tree taxa ($n = 21$, of which 12 are native) and herbaceous taxa ($n = 31$); while Sus in the FP landscape had a higher richness in shrubs ($n = 16$, with 15 natives).

For the “Seed production suitability” metric, the SU with the highest potential, given by the sum of the initial scores of the Floristic suitability category, was found in the FP landscape. This SU was located in a woody REI and was composed of blue gum eucalyptus and maritime pine in the higher strata, with dominant *Cistus* sp. And *Ulex* sp. In the lower strata, and narrow-leaved mock privet (*Phillyrea angustifolia* L.) together with mastic tree (*Pistacia lentiscus* L.) in abundance. The lowest potential for seed dispersal by short-range dispersers was achieved in Sus of the IA landscape, where the sum of the initial scores for the Floristic

suitability category was equal to 0. These Sus were mostly composed of ashes and willows (*Salix* sp.) and dominated in the understory by *Asteraceae*, *Poaceae* and *Typhaceae*.

As for the “Pollen production suitability” metric, the SU with the highest potential for pollination was equally found in the FP landscape. This SU was located in a non-woody REI dominated by *Ulex* sp., with *Cistus* sp., *Erica* sp., Mediterranean buckthorn (*Rhamnus alaternus* L.), myrtle (*Myrtus communis* L.), *Rubus* sp., and *Erica* sp. In abundance, together with *Poaceae* and *Xanthorrhoeaceae* herbaceous families. The lowest potential for pollination was found in the IA landscape, in an atypically woody REI dominated by ash and by planted Mediterranean hackberry (*Celtis australis* L.) and river oak (*Casuarina cunninghamiana* Miq.), with several herbaceous *Poaceae* in the understory.

For the “Fruit production suitability” metric, although the EA landscape showed an overall higher proportion of Sus with scores of “high”, the SU with the highest potential to attract birds and non-flying small mammals was found in the IA landscape. This SU was located in a non-woody REI, featuring a total of 12 plant taxa, dominated by blackthorn (*Prunus spinosa* L.) and *Rubus* sp., with a large area occupied by elderberry (*Sambucus nigra* L.) and hawthorn (*Crataegus monogyna* Jacq.) alongside several *Apiaceae* and *Poaceae* herbaceous taxa. The lowest potential for seed dispersal by birds and non-flying small mammals was mostly found in Sus of the IA landscape.

4. Discussion

4.1. Relevance of REI's Structural Attributes for Biodiversity

The heterogeneity of habitats found within REIs and the spatial configuration of REI patches are essential features in supporting the dispersal of the considered biological groups. Several studies suggest that animal species richness and diversity are positively correlated with the structure and composition of riparian vegetation patches [2,29,82,83].

Woody REIs are generally more numerous and larger than non-woody REIs, covering a higher proportion of the study area in all landscapes while being less fragmented. Additionally, global HEIDI values tend to be significantly higher in woody REIs. This is especially evident in the Intensive Agriculture landscape. The larger mean patch size of woody REIs in the IA landscape may represent more available habitats for the biological communities [84], but the combination of elongated shapes with a higher fragmentation increases the likelihood of contact with the surrounding landscape [58]. In highly modified landscapes, such as irrigated croplands, REI patches are few and elongated, only exceptionally occupying large areas, meaning edge effects will be elevated, which is known to have a detrimental effect on biodiversity [7]. Thus, the lower estimated HEIDI values for Sus located in the IA landscape may suggest a higher influence of the croplands and their management practices on the overall quality of REIs, corroborating similar findings [85].

In the Extensive Agriculture landscape, even though woody REIs are more numerous than in the IA landscape, they are also significantly smaller and cover less of the total landscape. Despite this, the EA landscape had Sus with higher HEIDI values. This may be due to the less impactful nature of the surrounding “montado”, as agroforestry systems tend to be less detrimental for biodiversity when compared to agricultural areas [85,86].

Regarding the REIs located in the Forest Production landscape, even though they appear to have similar structural attributes to those of the IA landscape, they are less fragmented and occupy a much larger portion of the study area. This is a consequence of a higher number of patches, especially non-woody REI patches, which also tend to be larger than those of the IA landscape. Considering the higher coverage and the common contiguous character of REIs associated with riparian corridors, this may translate into wider REIs, where the detrimental edge effects on biodiversity are not as prevailing [69,87]. As a consequence, Sus located in the FP landscape may tend to have higher estimated HEIDI values. Nonetheless, woody and non-woody REIs in all three landscapes are important sources of biodiversity, as riparian areas tend to show higher species richness and diversity when compared to their human-dominated surroundings [3,19].

4.2. Floristic Suitability to Support Biodiversity-Related ES

4.2.1. Seed Dispersal by Short-Range Dispersers

Seed-harvester ants can play an important role as seed dispersers in Mediterranean grassland and scrublands [88]. These species are largely influenced by plant propagules, namely elaiosome-bearing seeds, which some plant species adapted to myrmecochory possess [62]. In this work, a wider diversity of myrmecochoric plant seeds was found in Sus located in the FP landscape. The low impactful management of REIs in the FP landscape allowed for the establishment of important native understory species for myrmecochory, such as myrtle and mastic tree. The preservation of a shrubby cover in riparian ecosystems is known to hold many species, thus contributes significantly to key ecosystems functions [66]. Nevertheless, *Cistus* sp., another important shrub taxa for myrmecochory, achieved the highest coverage in Sus of the EA landscape, which is consistent with the typical “montado” ecosystem. This type of understory cover is highly appreciated by seed-harvester ants, especially for ant species specialized in *Cistaceae* seeds [89]. These results are in agreement with the estimated HEIDI values, which proved to be significantly higher in the FP and EA landscapes when compared to the IA landscape.

4.2.2. Pollination by Medium-Range Dispersers

Mediterranean landscapes comprise a complex mosaic of different habitats that vary in the diversity of their floristic communities, pollinator communities, and pollination services [32,43]. Sus located in the FP landscape exhibited a higher proportion of Sus with a HEIDI score of “high” for the Floristic suitability category, reflecting a higher potential to attract pollinators, especially when comparing to the IA landscape. This is mainly due to a higher abundance of tree and shrub taxa with a higher value as forage resources for pollinators. Herbaceous taxa, by turn, were more relevant in Sus located in the IA and EA landscapes, where the structure of riparian vegetation was less dense. Other studies suggest high values for plant-pollinator communities in Mediterranean mixed oak woodlands [43]. Near-natural land-uses, such as riparian scrublands, riparian forests, and broadleaved forests also showed a higher capacity to support pollination services when compared to agricultural areas or forest production systems [32].

4.2.3. Seed Dispersal by Long-Range Dispersers

The vast majority of Mediterranean fleshy-fruited plants are dispersed either by birds alone or by some combination of birds and mammals [63]. Overall, the EA landscape showed more potential to support endozoochoric seed dispersal, given the higher proportion of Sus with “high” scores. Sus located in the EA landscape were commonly dominated by maritime pine and especially by a well-established *Cistus* sp. And *Rubus* sp. Shrub layers. Costa et al. [47] recognized the importance of birds as seed dispersers and refer *Rubus* sp., a widely distributed shrub in Sus of the EA landscape, as the most dispersed plant in a study across Portugal. *Caprifoliaceae* herbaceous taxa were also more commonly found in the EA landscape, most likely as a consequence of the frequent occurrence of open areas and less densely vegetated REIs. In undisturbed conditions, breeding bird species richness describes a linear increase from the pioneer herbaceous communities of headwaters to the canopy forest of lowland large streams [7], but this was not reflected in the proportion of Sus with HEIDI scores of “high” for the Seed production suitability metric. Some studies suggest that this phenomenon may be a consequence of an upstream-downstream anthropogenic gradient, where Mediterranean headwaters contain forested areas with less anthropogenic impact and flat lowlands are highly impacted by agricultural activities [90–92]. Additionally, increasing levels of birds and non-flying mammals’ biodiversity have been previously reported for riparian habitats embedded in agroforestry landscapes when compared to agricultural areas and forest production systems [73,93].

4.3. REI's Potential for Biological Dispersal in the Three Human-Dominated Mediterranean Landscapes

In this study, REIs located in the FP landscape showed the highest global potential for biodiversity-related ES, followed by the EA and the IA landscapes (although no significant differences were observed between HEIDI global values of the FP and the EA landscape).

The estimated global HEIDI values observed in the Sus of the FP landscape seem to be the result of a combination of factors. The reduced accessibility of riparian areas located in the FP landscape, with steep slopes and rocky formations, discourages human intervention in REIs and allows their expansions outwards from the active channel. Other studies suggest that habitat quality in Mediterranean riparian areas tends to increase with decreasing human pressure and increasing forest cover in the river surroundings [90,91]. The majority of dominant native trees found within SUs were either obligate or preferential riparian species, except for the FP landscape where riparian areas were dominated by cork oak trees. In the FP landscape, blue gum eucalyptus and maritime pine plantations are replacing old “montados”, forming a landscape mosaic composed of mixed stands for timber harvesting and cork extraction. The presence of a contiguous natural forest within the river watershed would increase the resilience of REIs to biodiversity degradation [94]. Nonetheless, these non-riparian woody plant communities currently play a role in providing physical habitat and food resources for many animal species that rely on riparian areas [24,45].

Concerning the REIs located in the EA landscape, we were expecting to find a higher potential for biodiversity support, given that agrosilvopastoral systems tend to increase ES provision and biodiversity, especially when compared to forest production and agricultural systems [73,86]. The current land management practices observed in the studied EA landscape, with the elimination of understory vegetation and grazing both within and surrounding REIs, are probably responsible for the decline of biodiversity in the studied riparian areas [73,93,95] and the decrease of riparian vegetation cover [94]. One reason for higher vegetation management in REIs of the EA landscape can be attributed to wildfire prevention. Riparian vegetation under a seasonally water-stressed “montado” system can function as dangerous corridors for wildfire propagation, especially when *Cistus* sp. are present in abundance [96], which was the case in our study area. Thus, understory clearing in such REIs should be kept at the minimum required to prevent wildfire propagation, while simultaneously complying with the essential requirements for biodiversity preservation. The conservation of shrub patches around tree trunks in the surrounding landscape, for instance, protects the superficial root system of cork oaks and may contribute greatly to improve the ecological quality of REIs [73].

As for the IA landscape, HEIDI results and REI landscape metrics showed that the structure and composition of riparian vegetation can be dramatically altered in riparian areas dominated by agriculture. This can be explained by water and space overdemand, crop production, and regulation of the water supply in agricultural areas [28]. For this reason, we were expecting to find a much higher pressure in the management of REIs located in the IA landscape. However, irrigation or drainage activities also create canals resembling riparian habitats. According to Carlson et al. [97], irrigation canals can be important landscape elements for biodiversity conservation in human-dominated landscapes, albeit functioning at a much lower level than natural aquatic and riparian ecosystems. These channels are usually low-grade mimics of the river, unstructured, narrow, and with low-quality vegetation structure, but still, they are potentially useful as hosts for REIs. Such canals in the IA landscape would greatly benefit from REIs with an increased width, as they often form protection buffer strips [5,11,97], essential to prevent sand extraction from riverbanks and sidebars, provide nutrient and sediment retention services, and discourage clear-cuts of riparian vegetation [5,32].

4.4. HEIDI Synthesis and Applications

Riparian Ecological Infrastructures can be managed to promote the sustainable development of agricultural land and forest production systems. The assessment of the main factors that drive animal species abundance and richness, such as the heterogene-

ity and quality of available habitats, is an important first step in developing sustainable management guidelines and policies for biodiversity conservation in human-dominated landscapes [10,11,98,99]. Riparian Ecological Infrastructures are located in heavily degraded riparian areas, due to the human-dominated nature of the landscape in which they are embedded. Zaimes and Iakovoglou [91] identified that for Mediterranean regions, the ecological status of riparian areas should be monitored using small-scale tools (e.g., protocols and bioindicators). The HEIDI may contribute to such assessment, by incorporating habitat quality features that have a direct known relationship with the diversity of distinct faunistic groups. Nonetheless, higher estimated HEIDI values may not be synonymous with a positive contribution to an overall good ecological status of REIs. As an example, although the exotic cultivated blue gum eucalyptus may be considered highly suitable for myrmecochory and pollination services, it is not beneficial to riparian ecosystems [27,88]. Concomitantly, *Rubus* sp. has been classified with a very high value for medium- and long-range dispersers. For this reason, REIs composed of dense strips of *Rubus* sp. provide valuable resources, rarely found in the surrounding landscape, and may constitute important ecological corridors for these faunistic groups. Nonetheless, the overdominance of this native shrub may be an indicator of some level of habitat degradation [73].

According to Daily et al. [100], decision-making regarding landscape management should be formed in collaboration with stakeholders, since they are key players in defining adequate alternative scenarios of future land use. The HEIDI could be a valuable tool for such collaboration, especially when a rapid assessment of biodiversity-related ES is desired. The higher the estimated HEIDI value for a specific biological dispersal group, the more likely the REI will feature better habitat conditions to provide the ES promoted by that group. Additionally, and since the HEIDI is composed of five distinct but complementary categories (Vegetation structure, Vegetation habitats, Associated habitats, Vegetation management, and Floristic suitability), riparian management action plans can be implemented by prioritizing interventions based on the scores of each category. The individual categorical HEIDI scores may identify the aspects of the vegetation that need to be restored or improved (those that show a low proportion of “high” scores), or that need to be conserved (with a higher proportion of “high” scores).

The HEIDI scoring system was applied to each biological dispersal group by evaluating the attractiveness of local flora and the suitability of REI habitats, based on extensive bibliographic research and expert knowledge. Other studies have successfully developed metric-based vegetation indicators as surrogates for the richness and abundance of animal species [53,60]. Nevertheless, the HEIDI scoring system was developed based on a theoretical and bibliographic-supported approach using a broad range of animal species from different regions and under distinct land-use systems. Although most species belonging to the same group may share similar biological dispersal abilities, intraspecific and interspecific differences are expected to be found. Certain ant, pollinator, bird, bat, and non-flying small mammal species may possess unique characteristics not contemplated by the current methodology. Additionally, the individual capacity for biological dispersal may vary depending on local conditions [101]. In this context, if the HEIDI is to be used as a predictor of biodiversity in human-dominated Mediterranean landscapes, a robust validation process must take place [102]. By making use of independent data on the local biodiversity of the considered biological dispersal groups, this validation procedure would ensure the avoidance of circular reasoning when using the HEIDI to predict the provision of ES. Without it, the HEIDI should be used with caution, and restricted to assess the heterogeneity of riparian habitats, notwithstanding its capacity to provide mindful insight about their potential to host the biological communities under analysis. For its application in other climatic areas, land-use systems, and non-riparian Ecological Infrastructures, the Floristic suitability category must be revised to include the plant species that are associated with such environments. Furthermore, families of herbaceous species are very diverse, and although some species from the same family may share similar characteristics (height, stem, palatable characteristics), others contribute differently to the Floristic suitability category

and are expressed by a single initial score. A lower taxonomic level for herbaceous species would increase the HEIDI's precision, but we intended to obtain a simplified field data collection to enable landscape management in collaboration with stakeholders [16,52].

5. Conclusions

Vegetation aspects, such as floristic composition, structural attributes, spatial arrangement, type and number of associated habitats, and level of management may provide relevant and complementary information about the capacity of Riparian Ecological Infrastructures (REIs) to support biodiversity-related Ecosystems Services (ES). Our results showed that REIs located in the Forest Production landscape displayed more potential to support short-range and medium-range dispersers, especially when compared to those located in the Intensive Agriculture landscape. The potential of REIs in supporting long-range dispersers appeared to be slightly higher in the Extensive Agriculture landscape, although those differences were not statistically significant. This can be explained by the longer travel distances associated with this group, overcoming barriers that appear as insurmountable to the other communities. The HEIDI is an index that favors the simplicity of use, allowing for better collaboration with stakeholders. Given its theoretical proxy-based nature, a robust validation process must take place if the index is to be used as a biodiversity and ES predictor in the Mediterranean region. Future work on ants, pollinators, birds, bats, and non-flying small mammal species richness, diversity, and activity rates could provide empirical evidence of the HEIDI strength. Additionally, a landscape connectivity analysis could complement the capacity of REIs to function as high-quality ecological corridors for the considered biological groups. Nonetheless, the results obtained by the currently proposed method can offer valuable insight when identifying potential riparian areas in need of conservation or restoration. The HEIDI, when used with caution, could be an important complementary tool in prioritizing actions for the sustainable management of Mediterranean human-dominated landscapes.

Supplementary Materials: The following are available online at <https://www.mdpi.com/article/10.3390/su131910508/s1>, Supplementary S1: Habitat Ecological Infrastructure's Diversity Index (HEIDI) Floristic suitability initial scoring criteria and references, Figure S1: River reaches of the IA landscape (Tagus), Figure S2: River reaches of the IA landscape (Sorraia), Figure S3: River reaches of the EA landscape, Figure S4: River reaches of the FP landscape, Figure S5: Woody (dark blue polygons) and non-woody (light blue polygons) REIs in the IA landscape (Tagus), Figure S6: Woody (dark blue polygons) and non-woody (light blue polygons) REIs in the IA landscape (Sorraia), Figure S7: Woody (dark blue polygons) and non-woody (light blue polygons) REIs in the EA landscape, Figure S8: Woody (dark blue polygons) and non-woody (light blue polygons) REIs in the FP landscape, Figure S9: SUs in the IA landscape (Tagus), within woody (dark blue polygons) and non-woody (light blue polygons) REIs, Figure S10: SUs in the IA landscape (Sorraia), within woody (dark blue polygons) and non-woody (light blue polygons) REIs, Figure S11: SUs in the EA landscape, within woody (dark blue polygons) and non-woody (light blue polygons) REIs, Figure S12: SUs in the FP landscape, within woody (dark blue polygons) and non-woody (light blue polygons) REIs, Table S1: Field protocol for the assessment of riparian habitat heterogeneity, Table S2: Scoring assigned to tree taxa for their suitability to seed dispersal by short-range dispersers (ants) based on dispersion mode and plant physiognomy, with references, Table S3: Scoring assigned to shrub taxa for their suitability to seed dispersal by short-range dispersers (ants) based on dispersion mode, with references, Table S4: Scoring assigned to herbaceous taxa for their suitability to seed dispersal by short-range dispersers (ants) based on dispersion mode and plant physiognomy, with references, Table S5: Scoring assigned to tree taxa for their suitability to pollination by medium-range dispersers (pollinators) based on pollination mode, with references, Table S6: Scoring assigned to shrub taxa for their suitability to pollination by medium-range dispersers (pollinators) based on pollination mode, with references, Table S7: Scoring assigned to herbaceous families for their suitability to pollination by medium-range dispersers (pollinators) based on pollination mode, with references, Table S8: Scoring assigned to tree taxa for their suitability to seed dispersal by long-range dispersers (birds and non-flying small mammals) based on dispersion mode and plant physiognomy, with references, Table S9: Scoring assigned to tree taxa for their suitability to seed dispersal by long-range dispersers

(birds and non-flying small mammals) based on dispersion mode, with references, Table S10: Scoring assigned to herbaceous taxa for their suitability to seed dispersal by long-range dispersers (birds and non-flying small mammals) based on dispersion mode, with references, Table S11: Dunn's post hoc comparisons between landscapes for the patch size, nearest neighbor, and shape index of woody and non-woody REIs, Table S12: Dunn's post hoc comparisons between landscapes for estimated global HEIDI values and of short-range and medium-range dispersers, Table S13: Plant-growth form classification, riparian classification, nativeness, and relative occurrence of tree taxa in SUs located in the IA, EA, and FP landscapes, Table S14: Plant-growth form classification, riparian classification, nativeness, and relative occurrence of shrubby taxa (%) in SUs located in the IA, EA, and FP landscapes, Table S15: Relative occurrence of herbaceous families (%) in SUs located in the IA, EA, and FP landscapes.

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