applied sciences

Article

Developing a Model to Select Indicator Species Based on Individual Species' Contributions to Biodiversity

Ta-Jen Chu [1,2,3], Yi-Jia Shih [1,2,*], Chun-Han Shih [3,4], Jia-Qiao Wang [1,2], Liang-Min Huang [1,2] and Shu-Chen Tsai [5]

1 Fisheries College, Jimei University, Xiamen 361021, China; chutajen@gmail.com (T.-J.C.); skyofstar1@jmu.edu.cn (J.-Q.W.); lmhuang@jmu.edu.cn (L.-M.H.)
2 Fujian Provincial Key Laboratory of Marine Fishery Resources and Eco-Environment, Xiamen 361021, China
3 Taiwan Wetland Society, Hsinchu 300, Taiwan; d05b45003@ntu.edu.tw
4 Department of Leisure & Tourism Management, Shu-Te University, Kaohsiung 824, Taiwan
5 College of Arts and Design, Jimei University, Xiamen 361021, China; tsaishuchen@jmu.edu.cn
* Correspondence: eja0313@gmail.com

Featured Application: A simple model was developed to select indicator species based on individual species' contributions to biodiversity. Targeting biodiversity can be used for environmental monitoring and to inform restoration designs.

Abstract: In both ecological research and engineering, the selection of indicator species is crucial. Biodiversity has always been an important policy objective for ecologists and environmental managers. Based on this target requirement, we developed a method that reveals the individual contributions of species to biodiversity to quantitatively identify indicator species for selection during environmental monitoring. The Siangshan Wetland in Hsinchu, Taiwan, was selected as an application case. The spread of mangroves not only changed the original habitat composition and function of benthic organisms in wetlands, but also led to problems such as estuary filling, flooding, and black mosquito breeding. Therefore, a large-scale mangrove removal project was undertaken by the Hsinchu City Government from October 2015 to March 2016. In this study, the biological effects of mangrove removal on benthic organisms and adjacent habitats were investigated from October 2015 to September 2016. According to biodiversity contribution algorithms, we identified five indicator species, namely, *Mictyris brevidactylus*, *Macrophthalmus banzai*, *Uca arcuata*, *U. lacteal*, and *U. borealis*. These indicator species had the most prominent biodiversity contribution, and they provided direct evidence of the beneficial effect of mangrove removal for wetland restoration. After mangrove deforestation, tidal flat species returned to their original habitats, and their related densities increased significantly in mangrove removal areas. Improving our understanding of the relationships between biodiversity and indicator species is crucial for the development of coastal management processes. Mangrove removal can be confirmed as an appropriate habitat rehabilitation strategy for benthic organisms. Consequently, these indicator species and the results obtained can provide valuable ecological information for those involved in coastal management or other officials seeking to control the spread of mangroves.

Keywords: environmental monitoring; restoration; Siangshan Wetland; mangrove removal; benthic organisms

Citation: Chu, T.-J.; Shih, Y.-J.; Shih, C.-H.; Wang, J.-Q.; Huang, L.-M.; Tsai, S.-C. Developing a Model to Select Indicator Species Based on Individual Species' Contributions to Biodiversity. *Appl. Sci.* **2022**, *12*, 6748. https://doi.org/10.3390/app12136748

Academic Editors: Wei-Ling Hsu, Yan-Chyuan Shiau and Hsin-Lung Liu

Received: 27 May 2022
Accepted: 1 July 2022
Published: 3 July 2022

1. Introduction

Biodiversity is often defined by biologists as the sum of genes, species, and ecosystems in an area [1]. The traditional biodiversity types identified in the past include taxonomic diversity [2], ecological diversity [2], morphological diversity (genetic diversity and molecular diversity) [3], and functional diversity [4]. Typically, "biodiversity" is used in place of defining terms such as species diversity and species richness [5]. It is also a measure of variation at the genetic, species, and ecosystem levels [6]. Biodiversity, the variety of life, is distributed heterogeneously across the Earth [7]. Ecological crises are understood to occur from changes in biodiversity due to habitat destruction and rapid species loss. Ecosystems are subject to a variety of anthropogenic threats, including disease, pollution, climate change, habitat alteration, overexploitation, and invasive species [8]. In 1993, the shared consciousness of this problem led to the Convention on Biological Diversity being held. The Biodiversity Convention of the UNCED Conference in Rio mentioned the importance of biodiversity, with particular reference to wetland ecosystems. The values of wetlands and wetland biodiversity were discussed, and a possible strategy for their conservation and wise use was suggested [9]. Since 1971, the Ramsar Convention has promoted the conservation and rational use of wetlands. However, wetlands remain one of the most threatened ecosystems because they are directly damaged and suffer from all human activities [10].

Indicator species (ISs) are target living organisms that reflect or predict the status of the environment [11–15]. Thus, ISs are frequently used to monitor environmental changes, assess the efficacy of environmental management, and provide warning signals for impending ecological shifts [15]. In other words, ISs can signal a change in the biological conditions of a particular ecosystem and, thus, may be used to diagnose the health of an ecosystem [16–19]. ISs are mostly applied to monitor the health and integrity of an ecosystem or environment and assess habitat restoration and the effects of pollution and contamination [15]. In addition, the use of ISs is often incorporated into policies and regulations for monitoring ecological integrity [20], including in watersheds [21], lakes [22,23], semi-natural pastures [24,25], rangelands [26], and forests [27]. Almost 70% of invertebrates are particularly common indicators of aquatic and wetland health [15], with polychaetes (Annelida) used as indicators for marine pollution [28], macrobenthos used as indicators in coastal lagoons [29], and brachyuran crabs used as a biomonitoring tool for chemical pollution assessment [30].

Ecologists and engineers have frequently used ISs to monitor environmental changes and provide detectable signals for ecological shifts. For example, the life history differences and effects of physicochemical changes on coastal lagoon macrobenthos were investigated in the sewage-contaminated, moderately enriched Sykes Creek and the less polluted Banana River in Florida, USA [29]. Spatial and temporal changes in macrobenthic communities were studied at the Tyne sludge dump. Out of a total pool of 123 taxa, 81 taxa responded in only one study [31]. ISs are selected based on their sensitivity to a particular environmental attribute. Moreover, ISs can be selected based on some characteristics, including abundance, sensitivity, attractiveness, endangered, invasive species, or any combination of these [15]. While ISs have rarely proven to be an effective tool for monitoring ecosystems and informing management decisions [31,32], the use of quantitative methods to identify ISs is rapidly evolving. Consequently, the conceptual model of ISs, including direct relationships between ISs, drivers of ecosystem change, and underlying processes and variables, was extended [32]. Vegetation changes were monitored for good management practices for the sustainable use of natural rangeland resources. Several quantitative models have been developed to select IS, including principal component analysis [33], context-sensitive joint species distribution model (JSDM) [34], probability of species occurrence combined with environmental factors [35], maximum conditional co-occurrence [36], and threshold indicator taxa analysis (TITAN) [37]. However, few studies have quantitatively selected ISs from completely assembled species data, especially for benthic organisms.

Many of the well-known wetland values are related to their biodiversity [9]. The number of species is determined by the birth, death, immigration, and emigration rates of species in an area [7]. Biodiversity is one of the primary criteria used in the protection of wetlands [9]. Wetlands as ecotones are unique environments exhibiting considerable potential for biodiversity [38]. Measuring species-level biodiversity involves determining a diversity index based on individual species abundances. The elements that affect such a biodiversity index are the number of unique species (species richness) and the evenness in the distribution of species abundances. Numerous indices are applied in ecological research to measure species-level biodiversity [39]. Species abundance could be measured either in terms of biomass or the number of individual species [40].

Biodiversity is a quantitative measure that reflects how many different species exist in a community and simultaneously considers how evenly the basic entities are distributed among those living organisms. Therefore, we explored the value of individual species, which is based on the contribution to biodiversity, to identify ISs. Contribution to biodiversity from each species was employed because we sought to calculate the probabilities of biodiversity from each individual species when survey data were available. Therefore, the desired individual probabilities and expectations were treated as conditions of contribution. Thus, we developed a novel, quantitative method that was useful for determining the specific contribution of species to biodiversity. In this case study, the mangrove removal project in the Siangshan Wetland Conservation Area, Hsinchu, Taiwan, was used and analyzed.

2. Materials and Methods

2.1. Model Building

Numerous methods are available for measuring biodiversity within an ecosystem, with dominance indices and information statistic indices being the most prominent (Pielou, 1966; 1975) [41,42]. The Shannon–Wiener biodiversity index (H') [43,44] and Simpson index (D) [45] are two commonly used measures. The respective equations for these two indices are as follows:

$$\text{Shannon index } (H') = -\sum p_i \ln p_i \tag{1}$$

$$\text{Simpson index } (D) = \sum n\,(n-1)/N\,(N-1) = 1/\sum (p_i)^2 \tag{2}$$

In the Shannon and Simpson indices, p_i is the proportion (n_i/N) of the number of individuals of species found (n_i) divided by the total individuals found (N), and i is the number of species. H'_{co} and D_{co} represent the individual contributions of species in an ecosystem's diversity. Equations (3) and (4) for these indices are as follows:

$$H'_{CO}(Si) = \frac{P_i * \ln\ P_i}{\sum_{i=1}^{n} P_i * \ln P_i} \tag{3}$$

$$D_{CO}(Si) = \frac{\frac{1}{(P_i)2}}{1/\sum_{i=1}^{n}(P_i)2} \tag{4}$$

To identify ISs, we use the concept of contribution to biodiversity. Let H'_{co} or D_{co} be the separate contribution of each species i to biodiversity such that H'_{co} or D_{co} is a calculation of the individual biodiversity value of each species i divided by the sum of the total biodiversity of all species.

Therefore, when species i is present and sampled at the kth site, the contribution to the biodiversity of each species i aggregated from different sites k is calculated using (5) and (6).

$$H'_{co-site}(S_i) = \left(\prod_{k=1}^{l} H'_{CO}\ (Si)k\right)^{\frac{1}{l}} \tag{5}$$

$$D_{co-site}(S_i) = \left(\prod_{k=1}^{l} D_{CO}\ (Si)k\right)^{\frac{1}{l}} \tag{6}$$

where k = 1, 2, . . . , l site for the ith species. When some species do not exist at the kth site, a small value is set, such as 0.001, and this value is substituted into the formula.

The contribution to the biodiversity of each species i integrated from each month t is calculated using (7) and (8).

$$H'_{co-site-month}(S_i) = \left(\prod_{t=1}^{m} H'_{co-site}(S_i)_t\right)^{\frac{1}{m}} \quad (7)$$

$$D_{co-site-month}(S_i) = \left(\prod_{t=1}^{m} D_{co-site}(S_i)_t\right)^{\frac{1}{m}} \quad (8)$$

where t = 1, 2, ... , m site for the ith species.

To determine ISs, the maximum value of contribution to biodiversity among all species can be obtained, as shown in (9) and (10).

$$max\ H'_{c.k.m.}(S_i) \quad (9)$$

$$max\ D_{c.k.m.}(S_i) \quad (10)$$

2.2. Case Study

2.2.1. Background of Study Area

The coastline of the Siangshan Wetland is approximately 8 km long with a total area of approximately 1600 ha [46]. It is a breeding ground for shrimp, crabs, shellfish, and benthic organisms [47,48]. Since 2001, the Siangshan Wetland has been officially designated as the Hsinchu City Coastal Wildlife Sanctuary. The study area is situated north of the Sanxing stream and south of the Haishan Fishing Port in Hsinchu, Taiwan. A map of the study area is depicted in Figure 1.

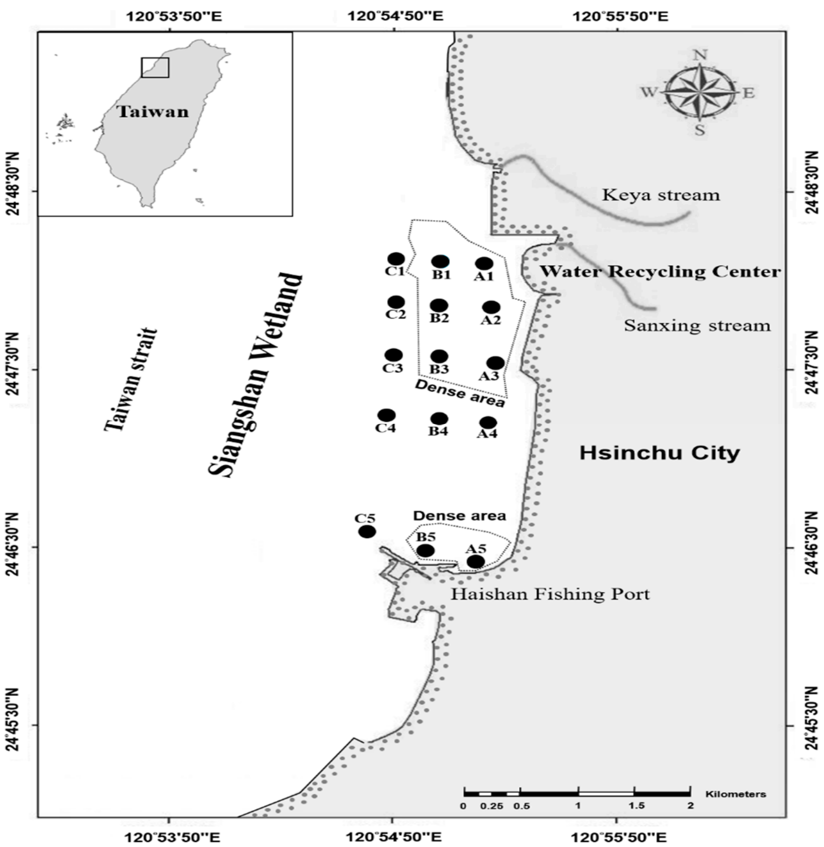

Figure 1. Sampling sites of the Siangshan Wetland, Hsinchu, Taiwan.

Mangroves have been considered good ecosystems for the past three decades [49]. Therefore, mangroves (*Kandelia candel*) are artificially planted in the Siangshan Wetland. Since 1992, a survey has been conducted, and it was revealed that the area contains approximately 5300 mangrove trees. In 1995, the area occupied by these trees was separately estimated to be 0.1 ha by the Taiwanese Government and Endemic Species Research Institute of Taiwan. Due to the original mudflat geology, hydrology, and estuary effects, the mangrove area expanded rapidly to 107 ha by 2003, as reported by the Hsinchu City Wild Animal Sanctuary Habitat Rehabilitation and Mangrove Research Program [47,50]. This expansion had negative effects, including habitat singularity, the decline of species abundance and biodiversity, flooding, and mosquito breeding. Furthermore, this initiated a debate in news/media regarding the positive and negative effects of mangroves, both in civil society and academic institutions. It was finally decided to cut down the mangroves and remove them, according to the 2011 National Important Wetland Ecological Environment Investigation and Rehabilitation Project (https://wetland-tw.tcd.gov.tw/tw/index.php (accessed on 29 June 2022)) and Assessment of Wetland Habitat Restoration in Siangshan Wetland Program [51]. The removal of mangroves between 2011 and 2015 was difficult and proved unsatisfactory. Subsequently, a large-scale removal project was planned by the Hsinchu Municipal Government in October 2015. In this study, in addition to the entire Siangshan Wetland area, special attention was paid to ecological changes in the mangrove removal area [47,48].

2.2.2. Biological Survey and the Ecological Community Structure

The Hsinchu City Government ran a large-scale mangrove removal project from October 2015 to March 2016 to restore the wetland. From October 2015 to September 2016, the biological effects of mangrove removal on benthic crabs and their adjacent habitats were investigated. To enable a comparison of the area before and after mangrove removal, sampling was performed. Sampling for benthos was conducted from October 2015 to September 2016. Three survey transects—A, B, and C—were set from north to south, and 5 sampling sites were positioned along each transect, forming a total of 15 sampling sites (Figure 1). Eight sites (A1, A2, A3, A5, B1, B2, B3, and B5) were designated to the dense mangrove regions for comparison with non-mangrove regions.

The semidiurnal and diurnal tides with a tidal range of 2.7 m were analyzed along the Siangshan Wetland. Samples were collected on the tidal flats, and each sample was taken by excavating a frame (surface area: 1 m^2) at a depth of 30 cm. Ten random frames were collected at each site for species identification, individual abundance, and comparison before and after mangrove removal. The samples were sieved with a 1 mm mesh and preserved in an 8% formaldehyde–seawater solution. In the laboratory, crabs were sorted, and species were identified, counted, and preserved in a 70% alcohol solution.

3. Results

3.1. Species Contribution to Biodiversity

The benthic density was 1–4 ind./m^2, and there were 0–3 species types at sampling sites A1, A2, A3, and A5 before mangrove removal. The crab species included *U. arcuata*, *U. lacteal*, and *Helice formosensis*. No bivalves were found. After mangrove removal, the abundance of benthic organisms was 2–25 ind./m^2, and 2–7 species types were observed. The newly discovered crab species were *U. formosensis*, *M. banzai*, *M. brevidactylus,* and two bivalvia species: *Tellina jedonensis* and *Mactra veneriformis*. The benthic density and the number of species increased after mangrove removal. The lowest benthic density was observed in January 2016, and the highest benthic density was noted in September 2016. The lowest number of species was noted in February 2016, and the highest number of species was observed in August 2016. In the sampling sites of the non-mangrove region, the benthic density was 20–60 ind./m^2, and the number of species was 5–25 before mangrove removal. On the tidal flats of the non-mangrove region, crab species included *U. arcuata*, *U. lacteal*, *M. brevidactylus*, and *M. banzai*, and the bivalvia species included

Laternula anatine, *Meretrix lusoria*, *Cycladicama oblonga*, *T. jedonensis*, and *M. veneriformis*. After mangrove removal, the benthic density was 25–130 ind./m^2, and the number of species in the indicated sampling sites was 10–19. The benthic density was the lowest in January 2016, and the highest benthic density was observed in August 2016. The highest number of species was found in October 2015 and the lowest number was observed in January 2016.

Based on the calculations of Equations (9) and (10), the biodiversity contribution values of individual species showed significant monthly changes (Tables A1 and A4). Similarly, we also compiled and presented the rankings of species according to the probability values $H'_{c.k.m.}$ and $D_{c.k.m.}$ of each month (Tables A2 and A5). The two models are ranked slightly differently. The main species contributing to biodiversity are *M. brevidactylus*, *M. banzai*, *U. arcuata*, *U. lacteal*, and *U. borealis*, which contribute to biodiversity ($H'_{c.k.m.}$ and $D_{c.k.m.}$) having the highest contribution probability (Table 1). *M. brevidactylus*, *M. banzai*, and *U. arcuata* were the major contributing species throughout the entire year. In the summer and autumn, some target species also had a low ranking, including *U. lacteal*, *H. formosensis*, and *U. formosensi*.

Table 1. Probabilities of contribution to biodiversity ($H'_{c.k.m.}$ and $D_{c.k.m.}$) and each rank by crab species (S_i).

Species	$H'_{c.k.m.}$ (S_i)	Rank	$D_{c.k.m.}$ (S_i)	Rank
Helicana doerjesi	1.2×10^{-1}	8	1.4×10^{-3}	9
Helice formosensis	1.2×10^{-1}	6	1.1×10^{-3}	6
Mictyris brevidactylus	4.5×10^{-1}	1	1.8×10^{-1}	1
Macrophthalmus banzai	3.8×10^{-1}	2	5.5×10^{-2}	3
Macrophthalmusab reviatus	1.5×10^{-1}	11	2.7×10^{-3}	11
Ocypode ceratophthalmus	1.5×10^{-1}	8	2.8×10^{-3}	8
Ocypode stimpsoni	1.1×10^{-1}	7	4.6×10^{-4}	7
Ocypode sinensis	1.3×10^{-2}	14	2.5×10^{-6}	14
Uca arcuata	3.5×10^{-1}	3	4.1×10^{-2}	2
Uca formosensis	1.1×10^{-1}	10	6.2×10^{-4}	9
Uca lactea	2.9×10^{-1}	4	2.4×10^{-2}	4
Uca perplexa	1.0×10^{-2}	18	1.0×10^{-6}	18
Uca borealis	1.7×10^{-1}	5	3.7×10^{-3}	5
Scopimera longidactyla	6.7×10^{-3}	20	3.1×10^{-7}	20
Scopimera globosa	5.4×10^{-3}	15	1.5×10^{-7}	15
Scopimera bitympana	5.7×10^{-2}	13	2.7×10^{-4}	13
Scylla serrata	6.9×10^{-3}	15	3.3×10^{-7}	15
Pagurus dubius	6.5×10^{-3}	20	3.2×10^{-7}	20
Diogenes penicillatus	5.3×10^{-3}	18	1.4×10^{-7}	18
Parapagurus diogenes	1.2×10^{-2}	17	1.0×10^{-6}	17
Parapagurus obtusifrons	2.2×10^{-2}	12	4.4×10^{-5}	12

The probabilities of species contributions to biodiversity ($H'_{c.k.m.}$ and $D_{c.k.m.}$) at different sites are shown in Tables A3 and A6, respectively. In Table A3, two species, *M. brevidactylus* and *M. banzai*, appeared in most areas, except the mangrove area. *U. arcuate*, *U. lacteal*, *H. formosensis*, and *U. borealis* were observed in the A1, A2, and A3 locations within the mangrove region. *M. brevidactylus*, *M. banzai*, and *H. formosensis* were observed in locations B1, B2, and B3, which were outside the mangrove region. A comparison between habitat types revealed that *H. formosensis* and *M. banzai* were observed more in sandy than muddy areas.

3.2. Using ISs to Monitor Mangrove Removal

Three crab species, namely, *U. arcuata*, *U. borealis*, and *M. banzai*, were observed at the sampling sites within the mangrove region before mangrove removal (Figure 2). After mangrove removal, *U. arcuata*, *U. borealis*, and *U. lacteal* were also observed. This indicates that *U. lacteal* returned to this area after mangrove removal. *U. arcuate*, *U. lacteal*, and *U. borealis* were generally found in mangroves, salt marshes, and sandy or muddy areas in the Siangshan Wetland.

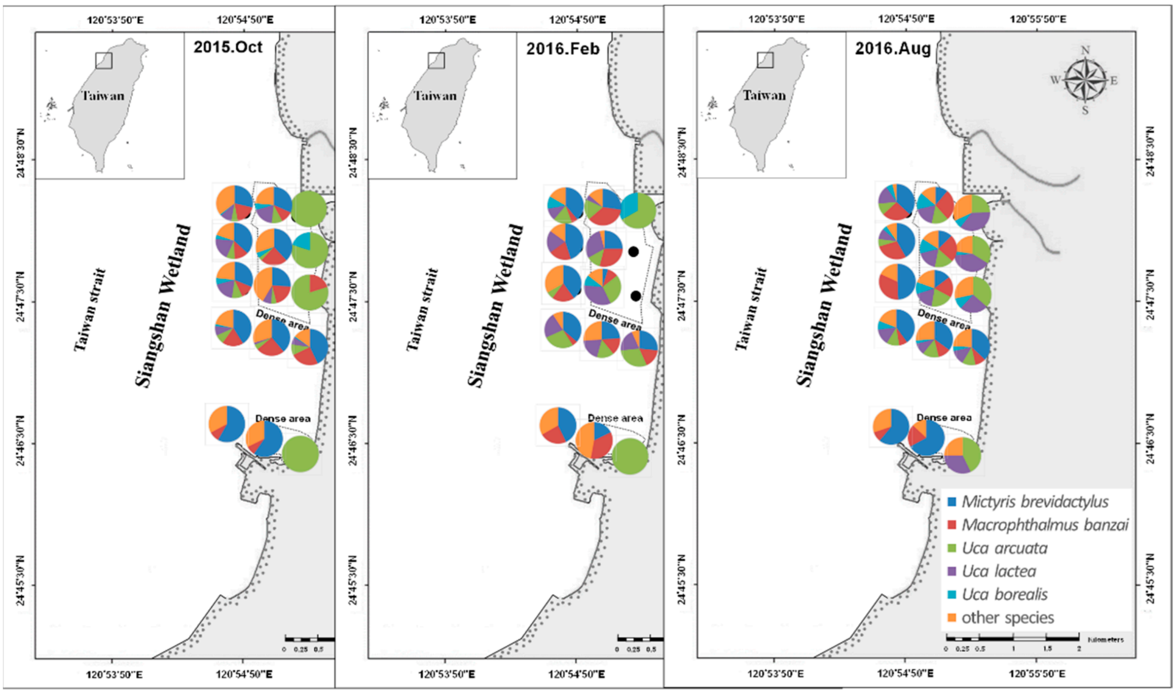

Figure 2. Spatiotemporal variations of target species with its quantity.

The densities of five ISs and untargeted species in the two regions (mangrove: A1, A5; non-mangrove: B1, B5) were compared (Figure 3). The density of crab species varied from month to month, but the densities measured in the mangrove region were significantly lower than those in the non-mangrove region. Comparing ISs with untargeted species, the density change of ISs can be seen more clearly from Figure 3. These results indicate that these benthic organisms were forced to migrate from their original habitat to nearby areas because of the spread of mangroves. After mangrove removal, the density of these species gradually recovered to the original level. Specifically, three crab species, namely, *U. arcuate*, *U. lacteal*, and *U. borealis*, returned to the original habitat shortly after mangrove removal.

Because fiddler crabs are omnivorous scavengers and eat items such as fish flakes, pellets, dried bloodworms, and brine shrimp in captivity, their feeding habits also played a vital role in the preservation of wetland environments. By sifting through sands, they aerate the substrate and control anaerobic conditions from occurring (Levinton et al., 1995) [52]. Regarding habitat, fiddler crabs prefer a wet environment. This means they are more likely to live along waterways and tidal pools in the wetlands after the tide's ebb. The expansion of mangroves often blocked the waterways, resulting in sediment gradually drying on land. These species then moved to areas outside the mangroves. When the mangroves were removed, clear waterways appeared, and these species naturally returned to their original habitat.

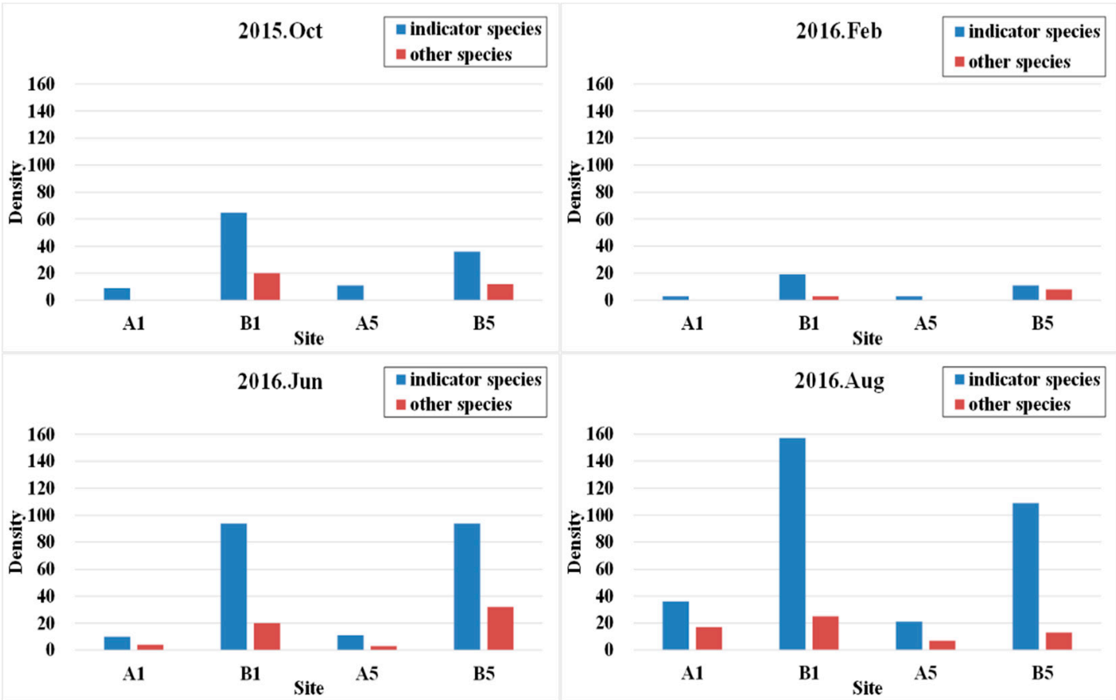

Figure 3. Densities of indicator species and other species at four sites, A1, B1, A5, and B5, in October 2015 and February, June, and August 2016.

4. Discussion

4.1. Comparison with Previous Research

On the other hand, a previous study [36] used the same material to select ISs by using the conditional probability of species co-occurrence. When a species exists in a habitat with other species, the status of these species is most likely to represent other species living in the same community [36]. Finally, the indicator species were selected, including *M. brevidactylus*, *M. banzai*, *U. arcuata*, *U. lacteal*, *U. borealis*, and *H. formosensis*. Comparing the ideas of the two models, one is conditional species co-occurrence and the other is biodiversity. Both models screen out the same species.

Due to differences in seasons and habitats, ISs vary at different months and sites. That said, each IS has its own suitable habitat and season. Through the accumulation of months and sites, species belonging to a wide area and a long time were screened out. It also means that the accumulation or product in the formula will select some species with a wide range of characteristics. They may be high-abundance, ubiquitous, low-sensitivity, and high-tolerance species. Conversely, low-abundance, scarce, highly sensitive, and low-tolerant species will not be screened. This also illustrates the difference between maximization and minimization in Equation (9). Differences in the use of maximization and minimization in mathematical notation seem to determine the dominant or sensitive species. This is also related to the size of the residuals in the statistics. If an analysis of environmental factors is added, the representative significance of index species is clearer. Another concept arises from the combination and selection of time and space. When the experimental design adopted more sites and a longer time, the selection of widely and narrowly distributed species also seemed to be different. Thus, the amount of time and space also seems to determine the different distributions of species.

4.2. Different Purposes and Practices

4.2.1. The Anping Beach Nourishment Project

In recent years, the performances of coastal protection activities aimed at protecting the coastal environment, such as levees, flood gates, sea walls, and stormwater retention areas, have been assessed [35]. Coastal structures, such as concrete blocks or dykes, are major coastal protection engineering facilities, which may be ecologically beneficial if properly designed. Conversely, inappropriate structural measures can lead to severe ecological deterioration. Since engineering design cannot take all species into account, the key to this engineering design is to consider ISs. In order to increase the biodiversity of artificial structures, the selection of ISs helps to realize the design of coastal structures based on biological considerations [35]. A practical case is to assess environmental influences on the artificial beach nourishment at the Anping coast of Taiwan by using benthic invertebrate communities. The Anping Beach Nourishment Project (22°59′ N and 120° 9′ E) is located between the northern dike off the Anping Harbor and the southern dike off the Anping Fishing Port [53]. They are constructed in the form of semi-enclosed gravel piles to protect the beach. These artificial headlands are designed to help stabilize the beach from the dredged sand stockpiled at the southern corner of the division in 2003 and 2005. The results indicate that both biotic and abiotic parameters showed significant differences before and after beach engineering construction. Biological conditions became worse in the beginning stages of the engineering but improved after the restoration work completion [53]. Compared to the method of Kuo et al., this study appears to provide an innovative and clear approach to coastal management and biodiversity applications.

4.2.2. Mangrove Removal Project

In Taiwan, due to topographic factors, sediments from the upper watershed were deposited by rivers in the estuaries of the west coast [46]. Thus, this deposition forms a series of shoals, and the swamps are suitable for mangrove growth and expansion. Therefore, mangroves have become one of the main habitat types of coastal wetlands in Western Taiwan [47]. Since 2006, ecotourism and conservation education have been carried out in wetlands. The delineation of "National Important Wetlands" has been completed by the Construction and Planning Department of the Ministry of the Interior. In 2011, there were 100 important wetlands, including 51 "international wetlands" and "national wetlands", 40 "local wetlands", and 7 undetermined wetlands [48,54]. Moreover, the Wetland Conservation Act was passed in July of 2013, and the implementation of the Act began on 2 February 2015. Therefore, Taiwan has become an area of wetland protection and specific legislation [54].

The Siangshan Wetland is located in Hsinchu City. Mangroves have been planted since 1969, causing a change in the original habitat and posing a serious threat to the endemic Taiwanese fiddler crab species (such as *U. formosensis*). Unfortunately, mangroves were defined as invasive species, which altered the structure and function of benthic habitats and caused sediments to accumulate with estuarine flooding during heavy rains, and allowed the invasion of the small black mosquito (*Forcipomyia taiwana*) [47,48]. The expansion of mangrove lands changed the state of the habitat substrate, and *U. formosensis*, for example, was forced to move outwards or even disappeared. When mangroves are removed, the habitat gradually returns to its original state [36]. At this time, species that disappeared in the past gradually return and increase in abundance. Five indicator species, *M. brevidactylus*, *M. banzai*, *U. arcuata*, *U. lacteal*, and *U. borealis*, were identified as having high co-occurrence probabilities, which provide direct evidence of the benefits of mangrove removal on wetland restoration.

Compared with our results, both studies selected similar indicator species. These ISs also have a high probability of co-occurrence and a high contribution to biodiversity. Under the premise of biodiversity, these species can also be set as target species for policy implementation.

4.3. Developing Restoration or Mitigation Mechanisms by Using ISs

Integrating the ideas of previous studies [35,36], more complex conditions will be presented to select ISs. Another approach, trait-based metrics, can also be used to select ISs, where assemblages respond to a gradient of environmental degradation. The functional trait method can also screen out good candidates and ISs that are of ecological significance for biomonitoring and are broadly applicable over biogeographic boundaries [55,56]. In this context, the complex factor with the selection of ISs can be expressed in terms of its concept as follows:

$$\text{Selection of indicator species} = \text{maximum or minimum}\,\textit{integrated function (species trait, species abundance, time, place, and environment factor)} \quad (11)$$

The habitat evaluation procedure (HEP) for formulating a restoration or mitigation project was developed and is shown in Figure 4. Firstly, this procedure combines the quantitative method of the indicator species from the environmental monitoring plan to select ISs. Second, it analyzes the relationship between the species and the environment after the selected ISs. The biological coefficient can be obtained with the environment or biological parameters with engineering. Finally, this information can be used to develop rehabilitation or recovery plans.

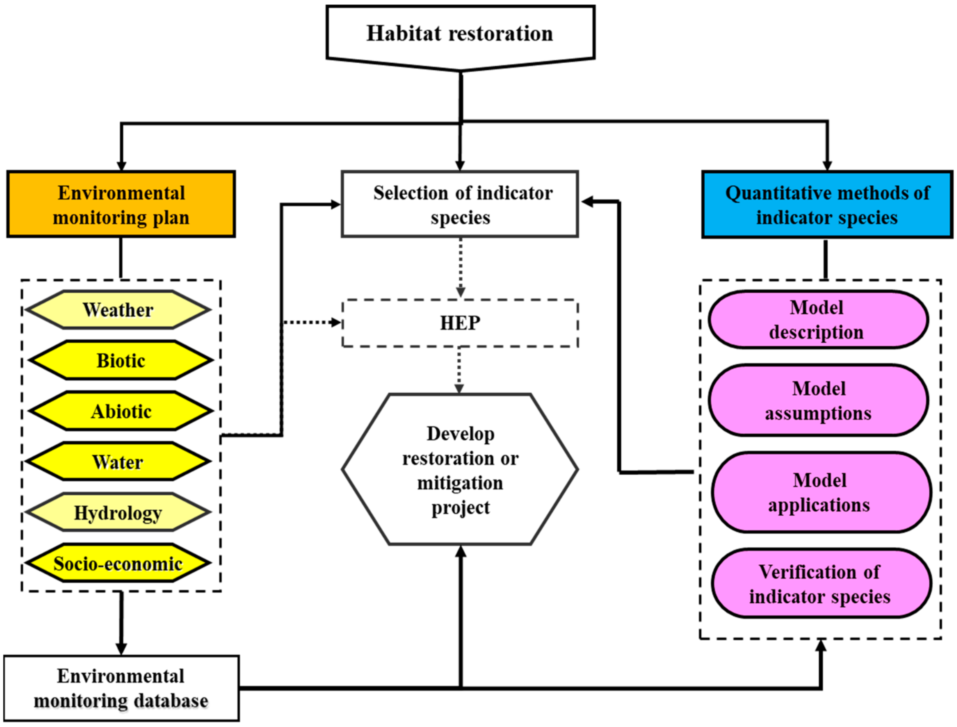

Figure 4. A mechanism and procedure for formulating a rehabilitation of mitigation project by using indicator species.

When government managers formulate restoration or rehabilitation projects, collecting biological information and understanding the meaning of environmental habitat is an important first step. In particular, in the engineering design stage, the environmental or engineering parameters must be obtained from the biological coefficient, as shown in Figure 4. Therefore, engineering designed in this way is also called ecological engineering and considers biological and ecological significance. It is crucial that clear biological coefficients can be obtained after determining ISs. Engineering parameters include porosity, the surface roughness of structure, and the particle size of the bottom substrate. From these, it can be determined whether the sediment is mud or hard soil, which originates from the consideration that the habitat choice is a mudflat or mangrove.

Overall, almost 80% of studies on ISs have used multiple species within the same taxa or a cross-taxa index as indicators [15]. Almost all pertinent studies on ISs have been descriptive field investigations and have selected numerical dominants in areas that differ widely in terms of environmental gradients [29,57]. In this study, we developed a new calculation method that incorporates a higher probability of contributing to biodiversity across taxonomic groups based on IS analyses. Specifically, the concept of ISs is based on the hypothesis that environmental shifts are reflected by changes in the diversity, abundance, or growth rate of species [11–14]. Therefore, ISs are usually employed for monitoring in the fields of wildlife conservation, habitat management, and ecological restoration [58–61]. Our results demonstrate that some taxonomic groups contribute significantly more to biodiversity than others. The evaluation of conservation or restoration requires a holistic approach that integrates the consideration of several endpoints on different scales (target species, indicator groups, and an ecosystem scale) [62]. In this paper, ISs were selected to assess environmental changes after mangrove removal. The combination of various approaches and their integration into holistic perspectives is required so that an understanding of ISs and the biodiversity of marine ecology can be further developed.

We further confirmed that the relationships between biodiversity and ISs are crucial for coastal management. This study can assist policymakers and planners involved in coastal management in providing superior ecological assessment methods and enable them to assess ISs and their contributions to biodiversity more effectively. Collectively, the proposed ISs are those that, for ecological reasons, are believed to be valuable for the understanding, management, and conservation of the natural environment. Currently, numerous marine scientists agree that issues related to the maintenance of original wetland ecosystems are more complex than ever. This consideration will assist in constructing an evaluation model with greater precision and allow the work of conservationists to continue.

Author Contributions: Conceptualization, T.-J.C., Y.-J.S. and C.-H.S.; methodology, T.-J.C., Y.-J.S. and C.-H.S.; software, T.-J.C., Y.-J.S. and C.-H.S.; validation, T.-J.C., Y.-J.S., C.-H.S., J.-Q.W. and L.-M.H.; formal analysis, T.-J.C. and C.-H.S.; investigation, T.-J.C., C.-H.S. and Y.-J.S.; resources, T.-J.C. and C.-H.S.; data curation, T.-J.C. and C.-H.S.; writing—original draft preparation, T.-J.C. and C.-H.S.; writing—review and editing, T.-J.C., C.-H.S., Y.-J.S., J.-Q.W., S.-C.T. and L.-M.H.; visualization, T.-J.C., C.-H.S., Y.-J.S., J.-Q.W., S.-C.T. and L.-M.H.; supervision, T.-J.C. and C.-H.S.; project administration, T.-J.C. and C.-H.S.; funding acquisition, T.-J.C. and C.-H.S. All authors have read and agreed to the published version of the manuscript.

Funding: This work was supported by the Jimei University grant number C619061 and Taiwan Wetland Society grant number TWS 107-2221-A-002. The funders had no role in the study design, data collection and analysis, the decision to publish, or the preparation of the manuscript.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Not applicable.

Acknowledgments: We thank Kuo Yi-Yu for his contributions regarding the suggested revisions to the manuscript. Useful suggestions from the anonymous reviewers were incorporated into the manuscript.

Conflicts of Interest: The authors declare that there is no conflict of interest.

Appendix A

Table A1. Probabilities of contribution to biodiversity ($H'_{c.k.m.}$) by crab species (S_i) by month.

Species	**Time**												$H'_{c.k.m.}(S_i)$
	2015			**2016**									
	October	**November**	**December**	**January**	**February**	**March**	**April**	**May**	**June**	**July**	**August**	**September**	
Helicana doerjesi	5.2×10^{-2}	1.4×10^{-1}	4.6×10^{-2}	6.1×10^{-2}	1.1×10^{-1}	7.9×10^{-2}	4.1×10^{-2}	9.5×10^{-2}	7.4×10^{-2}	7.7×10^{-2}	5.3×10^{-2}	1.0×10^{-1}	1.2×10^{-1}
Helice formosensis	7.3×10^{-3}	9.8×10^{-2}	2.6×10^{-2}	7.2×10^{-2}	5.0×10^{-2}	8.3×10^{-2}	9.5×10^{-2}	1.0×10^{-1}	9.1×10^{-2}	1.3×10^{-1}	1.7×10^{-1}	1.8×10^{-1}	1.2×10^{-1}
Mictyris brevidactylus	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	3.5×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	4.5×10^{-1}
Macrophthalmus banzai	2.9×10^{-1}	3.2×10^{-1}	3.1×10^{-1}	3.2×10^{-1}	3.1×10^{-1}	3.0×10^{-1}	2.7×10^{-1}	2.8×10^{-1}	2.8×10^{-1}	2.9×10^{-1}	2.8×10^{-1}	2.8×10^{-1}	3.8×10^{-1}
Macrophthalmusab reviatus	1.0×10^{-1}	1.3×10^{-1}	1.2×10^{-1}	1.8×10^{-1}	6.3×10^{-2}	1.0×10^{-1}	7.1×10^{-2}	1.0×10^{-1}	8.3×10^{-2}	7.7×10^{-2}	8.1×10^{-2}	6.1×10^{-2}	1.5×10^{-1}
Ocypode ceratophthalmus	1.2×10^{-1}	1.5×10^{-1}	1.2×10^{-1}	3.5×10^{-2}	1.0×10^{-1}	1.0×10^{-1}	1.0×10^{-1}	1.2×10^{-1}	1.2×10^{-1}	6.2×10^{-2}	5.3×10^{-2}	9.2×10^{-2}	1.5×10^{-1}
Ocypode stimpsoni	0	7.8×10^{-2}	7.1×10^{-2}	1.4×10^{-1}	1.4×10^{-1}	6.9×10^{-2}	1.7×10^{-1}	8.5×10^{-2}	9.3×10^{-2}	5.2×10^{-2}	6.2×10^{-2}	8.3×10^{-2}	1.1×10^{-1}
Ocypode sinensis	7.3×10^{-2}	5.6×10^{-2}	1.5×10^{-2}	0	7.5×10^{-2}	0	0	1.4×10^{-2}	0	0	0	0	1.3×10^{-2}
Uca arcuata	2.3×10^{-1}	2.5×10^{-1}	3.0×10^{-1}	3.0×10^{-1}	3.0×10^{-1}	2.6×10^{-1}	2.7×10^{-1}	2.6×10^{-1}	2.5×10^{-1}	2.5×10^{-1}	2.7×10^{-1}	2.7×10^{-1}	3.5×10^{-1}
Uca formosensis	0	2.5×10^{-2}	2.6×10^{-2}	7.2×10^{-2}	5.0×10^{-2}	1.3×10^{-1}	1.1×10^{-1}	1.6×10^{-1}	1.6×10^{-1}	1.8×10^{-1}	1.5×10^{-1}	1.6×10^{-1}	1.1×10^{-1}
Uca lactea	2.1×10^{-1}	1.1×10^{-1}	1.7×10^{-1}	2.3×10^{-1}	2.8×10^{-1}	1.8×10^{-1}	2.1×10^{-1}	1.9×10^{-1}	2.6×10^{-1}	2.3×10^{-1}	2.8×10^{-1}	2.6×10^{-1}	2.9×10^{-1}
Uca perplexa	4.5×10^{-2}	3.9×10^{-2}	2.6×10^{-2}	0	2.1×10^{-2}	0	0	0	0	0	0	0	1.0×10^{-2}
Uca borealis	8.5×10^{-2}	1.0×10^{-1}	8.6×10^{-2}	4.8×10^{-2}	7.5×10^{-2}	1.1×10^{-1}	1.4×10^{-1}	9.8×10^{-2}	1.5×10^{-1}	1.1×10^{-1}	1.7×10^{-1}	1.4×10^{-1}	1.7×10^{-1}
Scopimera longidactyla	5.6×10^{-2}	4.4×10^{-2}	0	0	0	0	0	0	0	0	0	0	6.7×10^{-3}
Scopimera globosa	1.0×10^{-1}	0	0	0	0	0	0	0	0	0	0	0	5.4×10^{-3}
Scopimera bitympana	1.2×10^{-1}	1.4×10^{-2}	0	0	5.0×10^{-2}	8.7×10^{-2}	7.1×10^{-2}	6.0×10^{-2}	8.5×10^{-2}	4.1×10^{-2}	3.2×10^{-2}	6.3×10^{-2}	5.7×10^{-2}
Scylla serrata	7.0×10^{-2}	6.0×10^{-2}	0	0	0	0	0	0	0	0	0	0	6.9×10^{-3}
Pagurus dubius	6.3×10^{-2}	2.5×10^{-2}	0	0	0	0	0	0	0	0	0	0	6.5×10^{-3}
Diogenes penicillatus	7.0×10^{-2}	0	0	0	0	0	0	0	0	0	0	0	5.3×10^{-3}
Parapagurus diogenes	0	1.4×10^{-2}	0	0	0	4.9×10^{-2}	0	0	7.1×10^{-2}	0	2.1×10^{-2}	1.3×10^{-2}	1.2×10^{-2}
Parapagurus obtusifrons	6.3×10^{-2}	6.0×10^{-2}	1.2×10^{-1}	0	0	0	0	1.4×10^{-2}	1.2×10^{-2}	0	3.9×10^{-2}	3.5×10^{-2}	2.2×10^{-2}
Total species (S)	18	19	14	11	14	13	12	14	14	12	14	14	
Species richness index (D)	0.19	0.19	0.23	0.20	0.17	0.22	0.20	0.21	0.17	0.22	0.17	0.16	
Pielu's evenness index (J')	2.16	2.12	1.83	1.86	2.02	1.95	1.96	1.97	2.11	1.89	2.06	2.13	
Shannon index (H')	0.75	0.72	0.69	0.77	0.77	0.76	0.79	0.75	0.80	0.76	0.78	0.81	
Simpson index (SI)	2.49	2.67	2.17	1.78	2.33	1.86	1.73	1.93	1.88	1.53	1.77	1.74	

Table A2. Monthly rankings in terms of probability of contributing to biodiversity ($H'_{c.k.m.}$).

	Time											
Species	**2015**			**2016**								**Rank**
	October	**November**	**December**	**January**	**February**	**March**	**April**	**May**	**June**	**July**	**August**	**September**	
Helicana doerjesi	13	8	11	11	6	7	12	10	7	10	10	9	8
Helice formosensis	18	11	13	9	11	6	7	5	6	5	5	5	6
Mictyris brevidactylus	2	1	1	1	1	1	1	1	3	1	3	1	1
Macrophthalmus banzai	1	2	2	2	2	2	2	3	4	2	4	3	2
Macrophthalmusab reviatus	15	7	8	6	9	10	10	11	12	9	11	12	11
Ocypode ceratophthalmus	11	5	9	10	8	11	9	8	9	11	13	10	8
Ocypode stimpsoni	19	10	7	5	5	8	6	9	11	8	9	8	7
Ocypode sinensis	9	12	14	12	13	14	13	14	15	13	15	15	14
Uca arcuata	3	3	3	3	3	3	3	4	2	3	2	2	3
Uca formosensis	19	19	10	7	12	9	8	6	8	6	7	6	10
Uca lactea	4	4	4	4	4	4	4	2	1	4	4	3	4
Uca perplexa	17	15	12	12	14	14	13	15	15	13	15	15	18
Uca borealis	7	6	6	8	7	5	5	7	5	7	6	7	5
Scopimera longidactyla	16	14	15	12	15	14	13	15	15	13	15	15	20
Scopimera globosa	5	20	15	12	15	14	13	15	15	13	15	15	15
Scopimera bitympana	6	17	15	12	10	12	11	12	13	12	14	13	13
Scylla serrata	12	13	15	12	15	14	13	15	15	13	15	15	15
Pagurus dubius	14	16	15	12	15	14	13	15	15	13	15	15	20
Diogenes penicillatus	8	20	15	12	15	14	13	15	15	13	15	15	18
Parapagurus diogenes	19	17	15	12	15	13	13	15	10	13	12	14	17
Parapagurus obtusifrons	10	9	5	12	15	14	13	13	14	13	8	11	12

Table A3. Probabilities of the contribution of crab species to biodiversity ($H'_{c.k.m.}$) at different sites.

Species	Site															$H'_{c.k.m.}(S_i)$
	A1	**A2**	**A3**	**A4**	**A5**	**B1**	**B2**	**B3**	**B4**	**B5**	**C1**	**C2**	**C3**	**C4**	**C5**	
Helicana doerjesi	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	2.7×10^{-1}	3.2×10^{-1}	2.9×10^{-1}	2.8×10^{-1}	3.0×10^{-1}	2.6×10^{-1}	1.9×10^{-1}	1.7×10^{-1}	1.5×10^{-1}	1.1×10^{-1}	2.7×10^{-1}	0	2.3×10^{-2}
Helice formosensis	2.1×10^{-1}	0	1.1×10^{-1}	1.1×10^{-1}	0	2.2×10^{-1}	2.4×10^{-1}	2.4×10^{-1}	1.1×10^{-1}	0	0	0	0	1.0×10^{-1}	0	7.3×10^{-2}
Mictyris brevidactylus	3.6×10^{-1}	3.6×10^{-1}	3.4×10^{-1}	2.2×10^{-1}	3.5×10^{-1}	2.9×10^{-1}	2.9×10^{-1}	2.9×10^{-1}	2.0×10^{-1}	1.2×10^{-1}	1.7×10^{-1}	1.6×10^{-1}	1.3×10^{-1}	2.1×10^{-1}	0	7.2×10^{-2}
Macrophthalmus banzai	0	0	0	0	0	0	7.1×10^{-3}	3.7×10^{-2}	1.4×10^{-2}	0	5.2×10^{-2}	1.2×10^{-2}	0	1.1×10^{-2}	0	8.5×10^{-2}
Macrophthalmusab reviatus	1.8×10^{-1}	1.5×10^{-1}	1.8×10^{-1}	7.2×10^{-2}	0	2.3×10^{-1}	2.1×10^{-1}	2.2×10^{-1}	6.5×10^{-2}	7.2×10^{-3}	1.0×10^{-1}	1.1×10^{-1}	5.7×10^{-2}	9.0×10^{-2}	0	5.9×10^{-3}
Ocypode ceratophthalmus	0	0	0	0	0	0	0	4.2×10^{-2}	1.9×10^{-2}	0	5.8×10^{-2}	2.1×10^{-2}	0	0	0	9.8×10^{-3}
Ocypode stimpsoni	0	0	0	0	0	1.9×10^{-2}	2.2×10^{-2}	2.1×10^{-2}	1.4×10^{-2}	0	3.0×10^{-2}	3.8×10^{-2}	4.2×10^{-2}	1.9×10^{-2}	0	7.7×10^{-3}
Ocypode sinensis	0	0	0	0	0	2.3×10^{-2}	0	3.7×10^{-2}	5.9×10^{-3}	1.4×10^{-1}	7.1×10^{-2}	5.9×10^{-2}	5.0×10^{-2}	0	1.9×10^{-1}	5.7×10^{-3}
Uca arcuata	0	0	0	0	0	1.9×10^{-2}	1.2×10^{-2}	2.7×10^{-2}	0	0	3.8×10^{-2}	5.2×10^{-2}	4.2×10^{-2}	1.9×10^{-2}	0	1.8×10^{-1}
Uca formosensis	0	0	0	0	0	0	1.2×10^{-2}	3.7×10^{-2}	1.9×10^{-2}	0	5.2×10^{-2}	0	0	0	0	1.5×10^{-2}
Uca lactea	0	0	0	0	0	1.0×10^{-2}	1.2×10^{-2}	1.5×10^{-2}	1.0×10^{-2}	0	2.1×10^{-2}	2.1×10^{-2}	2.4×10^{-2}	1.1×10^{-2}	0	1.7×10^{-1}
Uca perplexa	0	0	0	4.1×10^{-2}	0	1.5×10^{-2}	7.1×10^{-3}	0	2.2×10^{-2}	2.7×10^{-2}	0	0	1.3×10^{-2}	2.7×10^{-2}	3.1×10^{-2}	3.1×10^{-3}
Uca borealis	0	0	0	3.7×10^{-2}	0	3.0×10^{-2}	2.7×10^{-2}	1.5×10^{-2}	4.2×10^{-2}	2.7×10^{-2}	2.1×10^{-2}	4.5×10^{-2}	6.5×10^{-2}	4.7×10^{-2}	3.1×10^{-2}	5.5×10^{-2}
Scopimera longidactyla	3.6×10^{-1}	3.6×10^{-1}	3.6×10^{-1}	2.7×10^{-1}	3.2×10^{-1}	2.9×10^{-1}	2.8×10^{-1}	3.0×10^{-1}	2.6×10^{-1}	1.9×10^{-1}	1.7×10^{-1}	1.5×10^{-1}	1.1×10^{-1}	2.7×10^{-1}	0	2.5×10^{-3}
Scopimera globosa	2.1×10^{-1}	0	1.1×10^{-1}	1.1×10^{-1}	0	2.2×10^{-1}	2.4×10^{-1}	2.4×10^{-1}	1.1×10^{-1}	0	0	0	0	1.0×10^{-1}	0	5.5×10^{-3}
Scopimera bitympana	3.6×10^{-1}	3.6×10^{-1}	3.4×10^{-1}	2.2×10^{-1}	3.5×10^{-1}	2.9×10^{-1}	2.9×10^{-1}	2.9×10^{-1}	2.0×10^{-1}	1.2×10^{-1}	1.7×10^{-1}	1.6×10^{-1}	1.3×10^{-1}	2.1×10^{-1}	0	7.9×10^{-3}
Scylla serrata	0	0	0	0	0	0	7.1×10^{-3}	3.7×10^{-2}	1.4×10^{-2}	0	5.2×10^{-2}	1.2×10^{-2}	0	1.1×10^{-2}	0	4.7×10^{-3}
Pagurus dubius	1.8×10^{-1}	1.5×10^{-1}	1.8×10^{-1}	7.2×10^{-2}	0	2.3×10^{-1}	2.1×10^{-1}	2.2×10^{-1}	6.5×10^{-2}	7.2×10^{-3}	1.0×10^{-1}	1.1×10^{-1}	5.7×10^{-2}	9.0×10^{-2}	0	2.4×10^{-3}
Diogenes penicillatus	0	0	0	0	0	0	0	4.2×10^{-2}	1.9×10^{-2}	0	5.8×10^{-2}	2.1×10^{-2}	0	0	0	4.3×10^{-3}
Parapagurus diogenes	0	0	0	0	0	1.9×10^{-2}	2.2×10^{-2}	2.1×10^{-2}	1.4×10^{-2}	0	3.0×10^{-2}	3.8×10^{-2}	4.2×10^{-2}	1.9×10^{-2}	0	5.0×10^{-3}
Parapagurus obtusifrons	0	0	0	0	0	2.3×10^{-2}	0	3.7×10^{-2}	5.9×10^{-3}	1.4×10^{-1}	7.1×10^{-2}	5.9×10^{-2}	5.0×10^{-2}	0	1.9×10^{-1}	1.3×10^{-2}

Table A4. Probabilities of contribution to biodiversity ($D_{c.k.m.}$) by crab species (S_i) by month.

Species	Time												$D_{c.k.m.}(S_i)$
	2015			2016									
	October	November	December	January	February	March	April	May	June	July	August	September	
Helicana doerjesi	1.4×10^{-4}	1.4×10^{-4}	1.0×10^{-4}	2.0×10^{-4}	1.1×10^{-3}	4.1×10^{-4}	7.7×10^{-5}	6.8×10^{-4}	3.5×10^{-4}	3.9×10^{-4}	1.4×10^{-4}	7.8×10^{-4}	1.4×10^{-3}
Helice formosensis	1.1×10^{-6}	1.1×10^{-6}	2.5×10^{-5}	3.2×10^{-4}	1.2×10^{-4}	4.7×10^{-4}	6.9×10^{-4}	8.8×10^{-4}	6.0×10^{-4}	1.7×10^{-3}	3.8×10^{-3}	4.6×10^{-3}	1.1×10^{-3}
Mictyris brevidactylus	1.3×10^{-1}	1.3×10^{-1}	1.5×10^{-1}	1.0×10^{-1}	7.8×10^{-2}	1.6×10^{-1}	1.4×10^{-1}	1.5×10^{-1}	1.1×10^{-1}	1.5×10^{-1}	8.8×10^{-2}	8.6×10^{-2}	1.8×10^{-1}
Macrophthalmus banzai	2.7×10^{-2}	2.7×10^{-2}	3.4×10^{-2}	4.2×10^{-2}	3.4×10^{-2}	2.8×10^{-2}	1.9×10^{-2}	2.1×10^{-2}	2.0×10^{-2}	2.6×10^{-2}	2.4×10^{-2}	2.1×10^{-2}	5.5×10^{-2}
Macrophthalmusab reviatus	7.9×10^{-4}	7.9×10^{-4}	1.4×10^{-3}	5.1×10^{-3}	2.3×10^{-4}	8.7×10^{-4}	3.0×10^{-4}	8.1×10^{-4}	4.6×10^{-4}	3.9×10^{-4}	4.4×10^{-4}	2.1×10^{-4}	2.7×10^{-3}
Ocypode ceratophthalmus	1.6×10^{-3}	1.6×10^{-3}	1.4×10^{-3}	5.1×10^{-5}	9.1×10^{-4}	9.7×10^{-4}	8.9×10^{-4}	1.5×10^{-3}	1.3×10^{-3}	2.1×10^{-4}	1.4×10^{-4}	6.3×10^{-4}	2.8×10^{-3}
Ocypode stimpsoni	0	0	3.1×10^{-4}	2.1×10^{-3}	2.4×10^{-3}	2.9×10^{-4}	4.0×10^{-3}	5.1×10^{-4}	6.5×10^{-4}	1.3×10^{-4}	2.1×10^{-4}	4.7×10^{-4}	4.6×10^{-4}
Ocypode sinensis	3.3×10^{-4}	3.3×10^{-4}	6.3×10^{-6}	0	3.5×10^{-4}	0	0	5.6×10^{-6}	0	0	0	0	2.5×10^{-6}
Uca arcuata	1.1×10^{-2}	1.1×10^{-2}	3.1×10^{-2}	3.2×10^{-2}	3.0×10^{-2}	1.7×10^{-2}	2.0×10^{-2}	1.6×10^{-2}	1.4×10^{-2}	1.3×10^{-2}	1.9×10^{-2}	2.0×10^{-2}	4.1×10^{-2}
Uca formosensis	0	0	2.5×10^{-5}	3.2×10^{-4}	1.2×10^{-4}	1.6×10^{-3}	1.2×10^{-3}	3.1×10^{-3}	3.0×10^{-3}	4.4×10^{-3}	2.7×10^{-3}	3.5×10^{-3}	6.2×10^{-4}
Uca lactea	7.2×10^{-3}	7.2×10^{-3}	3.9×10^{-3}	1.0×10^{-2}	2.4×10^{-2}	4.7×10^{-3}	8.0×10^{-3}	6.0×10^{-3}	1.5×10^{-2}	1.1×10^{-2}	2.2×10^{-2}	1.6×10^{-2}	2.4×10^{-2}
Uca perplexa	9.4×10^{-5}	9.4×10^{-5}	2.5×10^{-5}	0	1.4×10^{-5}	0	0	0	0	0	0	0	1.0×10^{-6}
Uca borealis	5.1×10^{-4}	5.1×10^{-4}	5.1×10^{-4}	1.1×10^{-4}	3.5×10^{-4}	1.2×10^{-3}	2.2×10^{-3}	7.5×10^{-4}	2.8×10^{-3}	1.2×10^{-3}	4.1×10^{-3}	2.4×10^{-3}	3.7×10^{-3}
Scopimera longidactyla	1.6×10^{-4}	1.6×10^{-4}	0	0	0	0	0	0	0	0	0	0	3.1×10^{-7}
Scopimera globosa	9.8×10^{-4}	0	0	0	0	0	0	0	0	0	0	0	1.5×10^{-7}
Scopimera bitympana	1.3×10^{-3}	1.3×10^{-3}	0	0	1.2×10^{-4}	5.4×10^{-4}	3.0×10^{-4}	2.0×10^{-4}	5.1×10^{-4}	7.8×10^{-5}	4.0×10^{-5}	2.2×10^{-4}	2.7×10^{-4}
Scylla serrata	2.9×10^{-4}	2.9×10^{-4}	0	0	0	0	0	0	0	0	0	0	3.3×10^{-7}
Pagurus dubius	2.2×10^{-4}	2.2×10^{-4}	0	0	0	0	0	0	0	0	0	0	3.2×10^{-7}
Diogenes penicillatus	2.9×10^{-4}	0	0	0	0	0	0	0	0	0	0	0	1.4×10^{-7}
Parapagurus diogenes	0	0	0	0	0	1.1×10^{-4}	0	0	3.1×10^{-4}	0	1.4×10^{-5}	5.0×10^{-6}	1.0×10^{-6}
Parapagurus obtusifrons	2.2×10^{-4}	2.2×10^{-4}	1.4×10^{-3}	0	0	0	0	5.6×10^{-6}	3.8×10^{-6}	0	6.9×10^{-5}	5.3×10^{-5}	4.4×10^{-5}
Total species (S)	18	19	14	11	14	13	12	14	14	12	14	14	
Species richness index (D)	0.19	0.19	0.23	0.20	0.17	0.22	0.20	0.21	0.17	0.22	0.17	0.16	
Pielu's evenness index (J')	2.16	2.12	1.83	1.86	2.02	1.95	1.96	1.97	2.11	1.89	2.06	2.13	
Shannon index (H')	0.75	0.72	0.69	0.77	0.77	0.76	0.79	0.75	0.80	0.76	0.78	0.81	
Simpson index (SI)	5.38	5.38	4.35	5.04	5.77	4.59	4.92	4.74	5.81	4.61	6.01	6.37	

Table A5. Monthly rankings in terms of probability of contributing to biodiversity ($D_{c.k.m.}$).

Species	Time											Rank
	2015			**2016**								
	October	**November**	**December**	**January**	**February**	**March**	**April**	**May**	**June**	**July**	**August**	**September**	
Helicana doerjesi	13	8	11	11	6	7	12	10	8	10	10	9	9
Helice formosensis	18	11	13	9	11	6	7	5	6	5	5	5	6
Mictyris brevidactylus	2	2	2	2	1	1	1	1	3	1	3	1	1
Macrophthalmus banzai	3	3	3	3	3	2	2	4	4	2	4	4	3
Macrophthalmusab reviatus	15	7	9	6	9	10	10	11	12	9	11	12	11
Ocypode ceratophthalmus	11	5	8	10	8	11	9	8	9	11	13	10	8
Ocypode stimpsoni	19	10	7	5	5	8	6	9	11	8	9	8	7
Ocypode sinensis	9	12	14	12	13	14	13	14	15	13	15	15	14
Uca arcuata	1	1	1	1	2	3	3	3	2	3	2	2	2
Uca formosensis	19	19	10	7	11	9	8	6	7	6	7	6	9
Uca lactea	4	4	4	4	4	4	4	2	1	4	1	3	4
Uca perplexa	17	15	12	12	14	14	13	15	15	13	15	15	18
Uca borealis	7	6	6	8	7	5	5	7	5	7	6	7	5
Scopimera longidactyla	16	14	15	12	15	14	13	15	15	13	15	15	20
Scopimera globosa	5	20	15	12	15	14	13	15	15	13	15	15	15
Scopimera bitympana	6	17	15	12	10	12	11	12	13	12	14	13	13
Scylla serrata	12	13	15	12	15	14	13	15	15	13	15	15	15
Pagurus dubius	14	16	15	12	15	14	13	15	15	13	15	15	20
Diogenes penicillatus	8	20	15	12	15	14	13	15	15	13	15	15	18
Parapagurus diogenes	19	17	15	12	15	13	13	15	10	13	12	14	17
Parapagurus obtusifrons	10	9	5	12	15	14	13	13	14	13	8	11	12

Table A6. Probabilities of ($D_{c.k.m.}$) at different sites.

Species	Site															$D_{c.k.m.}(S_i)$
	A1	A2	A3	A4	A5	B1	B2	B3	B4	B5	C1	C2	C3	C4	C5	
Helicana doerjesi	0	0	0	1.0×10^{-3}	0	1.8×10^{-5}	1.5×10^{-4}	8.0×10^{-5}	2.0×10^{-3}	3.1×10^{-4}	3.1×10^{-4}	3.9×10^{-4}	1.2×10^{-4}	6.3×10^{-4}	1.3×10^{-3}	2.6×10^{-3}
Helice formosensis	2.2×10^{-2}	2.7×10^{-2}	3.7×10^{-2}	9.6×10^{-4}	2.5×10^{-2}	1.1×10^{-3}	2.3×10^{-3}	4.1×10^{-3}	7.0×10^{-4}	5.8×10^{-4}	1.3×10^{-4}	3.1×10^{-4}	0	9.8×10^{-4}	0	5.4×10^{-3}
Mictyris brevidactylus	0	0	0	1.8×10^{-1}	0	2.7×10^{-2}	3.1×10^{-2}	2.4×10^{-2}	1.6×10^{-1}	2.8×10^{-1}	1.5×10^{-1}	1.6×10^{-1}	1.8×10^{-1}	1.8×10^{-1}	3.3×10^{-1}	2.0×10^{-1}
Macrophthalmus banzai	0	0	1.2×10^{-4}	1.8×10^{-2}	0	5.8×10^{-2}	4.7×10^{-2}	2.7×10^{-2}	2.0×10^{-2}	1.7×10^{-2}	4.8×10^{-2}	5.1×10^{-2}	6.6×10^{-2}	1.8×10^{-2}	1.0×10^{-2}	4.1×10^{-2}
Macrophthalmusab reviatus	0	0	0	5.1×10^{-3}	0	7.3×10^{-7}	1.7×10^{-5}	0	6.3×10^{-3}	0	0	3.5×10^{-5}	5.0×10^{-6}	4.3×10^{-3}	0	1.4×10^{-2}
Ocypode ceratophthalmus	0	0	0	0	0	4.7×10^{-5}	0	2.6×10^{-5}	6.2×10^{-6}	9.5×10^{-3}	8.6×10^{-4}	5.6×10^{-4}	5.0×10^{-4}	0	2.4×10^{-2}	1.5×10^{-2}
Ocypode stimpsoni	0	0	0	7.4×10^{-7}	0	1.8×10^{-5}	0	0	0	6.3×10^{-4}	8.1×10^{-3}	8.3×10^{-3}	1.6×10^{-2}	0	1.7×10^{-3}	3.0×10^{-2}
Ocypode sinensis	0	0	0	0	0	2.9×10^{-6}	1.0×10^{-4}	1.4×10^{-5}	4.4×10^{-5}	0	3.4×10^{-5}	6.2×10^{-5}	2.0×10^{-5}	1.2×10^{-5}	0	3.4×10^{-3}
Uca arcuata	1.5×10^{-1}	1.7×10^{-1}	1.9×10^{-1}	1.9×10^{-2}	3.0×10^{-1}	2.4×10^{-2}	2.3×10^{-2}	3.1×10^{-2}	1.6×10^{-2}	5.7×10^{-3}	3.9×10^{-3}	2.8×10^{-3}	1.2×10^{-3}	1.9×10^{-2}	0	3.0×10^{-2}
Uca formosensis	8.0×10^{-3}	0	1.0×10^{-3}	1.1×10^{-3}	0	9.1×10^{-3}	1.1×10^{-2}	1.2×10^{-2}	1.2×10^{-3}	0	0	0	0	8.7×10^{-4}	0	4.7×10^{-2}
Uca lactea	8.6×10^{-2}	1.3×10^{-1}	6.1×10^{-2}	8.5×10^{-3}	8.2×10^{-2}	2.7×10^{-2}	2.5×10^{-2}	2.6×10^{-2}	6.6×10^{-3}	1.5×10^{-3}	3.6×10^{-3}	3.5×10^{-3}	1.8×10^{-3}	7.7×10^{-3}	0	1.9×10^{-2}
Uca perplexa	0	0	0	0	0	0	1.0×10^{-6}	5.9×10^{-5}	6.2×10^{-6}	0	1.3×10^{-4}	3.9×10^{-6}	0	3.0×10^{-6}	0	9.7×10^{-3}
Uca borealis	4.8×10^{-3}	2.6×10^{-3}	5.1×10^{-3}	3.2×10^{-4}	0	1.0×10^{-2}	8.4×10^{-3}	9.4×10^{-3}	2.4×10^{-4}	1.1×10^{-6}	9.8×10^{-4}	1.2×10^{-3}	1.8×10^{-4}	5.9×10^{-4}	0	2.4×10^{-3}
Scopimera longidactyla	0	0	0	0	0	0	0	8.0×10^{-5}	1.1×10^{-5}	0	1.8×10^{-4}	1.5×10^{-5}	0	0	0	6.7×10^{-2}
Scopimera globosa	0	0	0	0	0	1.1×10^{-5}	1.7×10^{-5}	1.4×10^{-5}	6.2×10^{-6}	0	3.4×10^{-5}	6.2×10^{-5}	8.0×10^{-5}	1.2×10^{-5}	0	3.2×10^{-3}
Scopimera bitympana	0	0	0	0	0	1.8×10^{-5}	0	5.9×10^{-5}	6.9×10^{-7}	2.1×10^{-3}	3.1×10^{-4}	1.9×10^{-4}	1.2×10^{-4}	0	5.2×10^{-3}	8.3×10^{-3}
Scylla serrata	0	0	0	0	0	1.1×10^{-5}	4.3×10^{-6}	2.6×10^{-5}	0	0	6.1×10^{-5}	1.4×10^{-4}	8.0×10^{-5}	1.2×10^{-5}	0	7.4×10^{-3}
Pagurus dubius	0	0	0	0	0	0	4.3×10^{-6}	5.9×10^{-5}	1.1×10^{-5}	0	1.3×10^{-4}	0	0	0	0	5.9×10^{-2}
Diogenes penicillatus	0	0	0	0	0	2.9×10^{-6}	4.3×10^{-6}	6.5×10^{-6}	2.7×10^{-6}	0	1.5×10^{-5}	1.5×10^{-5}	2.0×10^{-5}	3.0×10^{-6}	0	1.7×10^{-3}
Parapagurus diogenes	0	0	0	7.4×10^{-5}	0	6.6×10^{-6}	1.0×10^{-6}	0	1.7×10^{-5}	2.7×10^{-5}	0	0	5.0×10^{-6}	2.7×10^{-5}	3.7×10^{-5}	2.5×10^{-3}
Parapagurus obtusifrons	0	0	0	6.0×10^{-5}	0	3.6×10^{-5}	2.7×10^{-5}	6.5×10^{-6}	8.3×10^{-5}	2.7×10^{-5}	1.5×10^{-5}	9.8×10^{-5}	2.4×10^{-4}	1.0×10^{-4}	3.7×10^{-5}	6.4×10^{-4}

References

1. Larsson, T.B.; Angelstam, P.; Balent, G.; Barbati, A.; Bijlsma, R.J.; Boncina, A.; Bradshaw, R.; Bücking, W.; Ciancio, O.; Corona, P.; et al. Biodiversity evaluation tools for European forests. In *Biodiversity Evaluation Tools for European Forests*; Ecological Bulletins: Oxford, UK; Blackwell Science: Hoboken, NJ, USA, 2001; Volume 50, p. 237.
2. Sahney, S.; Benton, M.; Ferry, P.A. Links between global taxonomic diversity, ecological diversity and the expansion of vertebrates on land. *Biol. Lett.* **2010**, *6*, 544–547. [CrossRef] [PubMed]
3. Campbell, A.K. Save those molecules! Molecular biodiversity and life *. *J. Appl. Ecol.* **2003**, *40*, 193–203. [CrossRef]
4. Zhang, X.; Xue, Y.; Zhang, C.; Ren, Y.; Xu, B.; Chen, Y. Sampling intensity influences the estimation of functional diversity indices of fish communities. *Ecol. Indic.* **2021**, *121*, 107169. [CrossRef]
5. Walker, B.H. Biodiversity and Ecological Redundancy. *Conserv. Biol.* **1992**, *6*, 18–23. [CrossRef]
6. Gaston, K.J.; Spicer, J.I. *Biodiversity: An Introduction*; John Wiley & Sons: Hoboken, NJ, USA, 2013.
7. Gaston, K.J. Global patterns in biodiversity. *Nature* **2000**, *405*, 220–227. [CrossRef] [PubMed]
8. Neeman, N.; Servis, J.; Naro-Maciel, E. Conservation Issues: Oceanic Ecosystems. In *Encyclopedia of the Anthropocene*; Elsevier: Amsterdam, The Netherlands, 2017; Volume 1–5, pp. 193–201. [CrossRef]
9. Denny, P. Biodiversity and wetlands. *Wetl. Ecol. Manag.* **1994**, *3*, 55–611. [CrossRef]
10. Gopal, B.; Junk, W.J. Biodiversity in wetlands: An introduction. In *Biodiversity in Wetlands: Assessment, Function and Conservation*; Gopal, B., Junk, W.J., Davis, J.A., Eds.; Backhuys: Leiden, The Netherlands, 2000; Volume 1, pp. 1–10.
11. Landres, P.B.; Verner, J.; Thomas, J.W. Ecological Uses of Vertebrate Indicator Species: A Critique. *Conserv. Biol.* **1988**, *2*, 316–328. [CrossRef]
12. Cairns, J., Jr.; Pratt, J.R. A history of biological monitoring using benthic macroinvertebrates. In *Freshwater Biomonitoring and Benthic Macroinvertebrates*; Rosenberg, D.M., Resh, V.H., Eds.; Chapman & Hall: New York, NY, USA, 1993.
13. Bartell, S.M. Biomarkers, bioindicators, and ecological risk assessment—A brief review and evaluation. *Environ. Bioindic.* **2006**, *1*, 39–52. [CrossRef]
14. Burger, J. Bioindicators: Types, Development, and Use in Ecological Assessment and Research. *Environ. Bioindic.* **2006**, *1*, 22–39. [CrossRef]
15. Siddig, A.A.H.; Aaron, M.; Ellison, A.; Ochs, A.; Villar-Leeman, C.; Lau, M.K. How do ecologists select and use indicator species to monitor ecological change? Insights from 14 years of publication in Ecological Indicators. *Ecol. Indic.* **2016**, *60*, 223–230. [CrossRef]
16. Lee, H.Y.; Shih, S.S. Impacts of vegetation changes on the hydraulic and sediment transport characteristics in Guandu man-grove wetland. *Ecol. Eng.* **2004**, *23*, 85–94. [CrossRef]
17. Lee, T.-M.; Yeh, H.-C. Applying remote sensing techniques to monitor shifting wetland vegetation: A case study of Danshui River estuary mangrove communities, Taiwan. *Ecol. Eng.* **2009**, *35*, 487–496. [CrossRef]
18. Shih, S.-S.; Yang, S.-C.; Lee, H.-Y.; Hwang, G.-W.; Hsu, Y.-M. Development of a salinity-secondary flow-approach model to predict mangrove spreading. *Ecol. Eng.* **2011**, *37*, 1174–1183. [CrossRef]
19. Yang, S.-C.; Shih, S.-S.; Hwang, G.-W.; Adams, J.B.; Lee, H.-Y.; Chen, C.-P. The salinity gradient influences on the inundation tolerance thresholds of mangrove forests. *Ecol. Eng.* **2013**, *51*, 59–65. [CrossRef]
20. Carignan, V.; Villard, M.-A. Selecting Indicator Species to Monitor Ecological Integrity: A Review. *Environ. Monit. Assess.* **2002**, *78*, 45–61. [CrossRef] [PubMed]
21. Moyle, P.B.; Randall, P.J. Evaluating the biotic integrity of watersheds in the Sierra Nevada, California. *Conserv. Biol.* **1998**, *12*, 1318–1326. [CrossRef]
22. Karr, J.R. Assessment of Biotic Integrity Using Fish Communities. *Fisheries* **1981**, *6*, 21–27. [CrossRef]
23. Harig, A.L.; Bain, M.B. Defining and Restoring Biological Integrity in Wilderness Lakes. *Ecol. Appl.* **1998**, *8*, 71–87. [CrossRef]
24. Pärt, T.; Söderström, B. Conservation Value of Semi-Natural Pastures in Sweden: Contrasting Botanical and Avian Measures. *Conserv. Biol.* **1999**, *13*, 755–765. [CrossRef]
25. Pärt, T.; Söderström, B. The effects of management regimes and location in landscape on the conservation of farmland birds breeding in semi-natural pastures. *Biol. Conserv.* **1999**, *90*, 113–123. [CrossRef]
26. Bradford, D.F.; Franson, S.E.; Neale, A.C.; Heggem, D.T.; Miller, G.R.; Canterbury, G.E. Bird Species Assemblages as Indicators of Biological Integrity in Great Basin Rangeland. *Environ. Monit. Assess.* **1998**, *49*, 1–22. [CrossRef]
27. Brooks, R.P.; O'Connell, T.J.; Wardrop, D.H.; Jackson, L.E. Towards a Regional Index of Biological Integrity: The Example of Forested Riparian Ecosystems. *Environ. Monit. Assess.* **1998**, *51*, 131–143. [CrossRef]
28. Dean, H.K. The use of polychaetes (Annelida) as indicator species of marine pollution: A review. *Rev. Biol. Trop.* **2008**, *56*, 11–38.
29. Grizzle, R.E. Pollution indicator species of macrobenthos in a coastal lagoon. *Mar. Ecol. Prog. Ser.* **1984**, *18*, 191–200. [CrossRef]
30. Arya, S.; Trivedi, J.N.; Vachhrajani, K.D. Brachyuran crabs as a biomonitoring tool: A conceptual framework for Chemical Pollution Assessment. *Int. Res. J. Environ.* **2014**, *3*, 49–57.
31. Bustos-Baez, S.; Frid, C. Using indicator species to assess the state of macrobenthic communities. *Hydrobiologia* **2003**, *496*, 299–309. [CrossRef]
32. Fleming, J.E.; Sutherland, C.; Sterrett, S.C.; Grant, E.H.C. A latent process model approach to improve the utility of indicator species. *Oikos* **2020**, *129*, 1753–1762. [CrossRef]

33. Ünal, S.; Şahin, B.; Urla, Ö.; Efe, B. The Use of Indicator species and ecological degradation model for rangeland condition assessment in Turkey. *Tarla Bitk. Merk. Araşt. Enst. Derg.* **2017**, *26*, 190–202.
34. De Jonge, M.M.J.; Benítez-López, A.; Hennekens, S.; Santini, L.; Huijbregts, M.A.J.; Schipper, A.M. Conditional love? Co-occurrence patterns of drought-sensitive species in European grasslands are consistent with the stress-gradient hypothesis. *Glob. Ecol. Biogeogr.* **2021**, *30*, 1609–1620. [CrossRef]
35. Kuo, Y.Y.; Shih, C.H.; Lee, Y.C.; Chang, W.T.; Chu, T.J. Identifying indicator species in habitats created by coastal structures. *China Ocean Eng.* **2010**, *24*, 117–134.
36. Chu, T.-J.; Shih, C.-H.; Lu, Y.-M.; Shih, Y.-J.; Wang, J.-Q.; Huang, L.-M. Incorporating Species-Conditional Co-Occurrence When Selecting Indicator Species to Monitor Restoration after Mangrove Removal from the Siangshan Wetland, Taiwan. *J. Mar. Sci. Eng.* **2021**, *9*, 1044. [CrossRef]
37. Baker, M.E.; King, R.S. A new method for detecting and interpreting biodiversity and ecological community thresholds. *Methods Ecol. Evol.* **2010**, *1*, 25–37. [CrossRef]
38. Junk, W.J. Flood pulsing and the linkages between terrestrial, aquatic, and wetland systems. *Verh. Int. Ver. Theor. Angew. Limnol./Proc. Int. Assoc. Theor. Appl. Limnol./Trav. Assoc. Int. Limnol. Theor. Appl.* **2005**, *29*, 11–38. [CrossRef]
39. Magurran, A.E. *Measuring Biological Diversity*; Blackwell Science: Oxford, UK, 2004; p. 256.
40. Buckland, S.T.; Magurran, A.E.; Green, R.E.; Fewster, R. Monitoring change in biodiversity through composite indices. *Philos. Trans. R. Soc. B Biol. Sci.* **2005**, *360*, 243–254. [CrossRef] [PubMed]
41. Pielou, E.C. The measurement of diversity in different types of biological collections. *J. Theor. Biol.* **1966**, *13*, 131–144. [CrossRef]
42. Pielou, E.C. *Ecological Diversity*; John Wiley and Sons: New York, NY, USA, 1975; p. 165.
43. Shannon, C.E. A Mathematical Theory of Communication. *Bell Syst. Tech. J.* **1948**, *27*, 379–423 and 623–656. [CrossRef]
44. Shannon, E.; Weaver, W. *The Mathematical Theory of Communication*; University of Illinois: Urbana, IL, USA, 1963.
45. Simpson, E.H. Measurement of diversity. *Nature* **1949**, *163*, 688. [CrossRef]
46. Kuo, Y.Y.; Lee, L.C. *Construct the Coastal Landscape and Ecology Environmentp*; Chansbook: Hsinchu, Taiwan, 2006; pp. 340–345. (In Chinese)
47. Chen, Y.-C.; Chu, T.-J.; Wei, J.-D.; Shih, C.-H. Effects of mangrove removal on benthic organisms in the Siangshan Wetland in Hsinchu, Taiwan. *PeerJ* **2018**, *6*, e5670. [CrossRef]
48. Chen, Y.-C.; Shih, C.-H. Sustainable Management of Coastal Wetlands in Taiwan: A Review for Invasion, Conservation, and Removal of Mangroves. *Sustainability* **2019**, *11*, 4305. [CrossRef]
49. Sandilyan, S.; Kathiresan, K. Mangrove conservation: A global perspective. *Biodivers. Conserv.* **2012**, *21*, 3523–3542. [CrossRef]
50. Chen, Z.B.; Xie, H.L. Report on the Hsinchu City Wild Animal Sanctuary Habitat Rehabilitation and Mangrove Research Program; Hsinchu City Government, Taiwan. 2003. Available online: https://ir.sinica.edu.tw/handle/201000000A/9653 (accessed on 29 June 2022). (In Chinese)
51. Yang, S.S. *Assessment of Wetland Habitat Restoration in Siangshan Wetland*; The Academia Sinica Institutional Repository: Hsinchu, Taiwan, 2011. (In Chinese)
52. Levinton, J.S.; Judge, M.L.; Kurdziel, J.P. Functional differences between the major and minor claws of fiddler crabs (Uca, family Ocypodidae, Order Decapoda, Subphylum Crustacea): A result of selection or developmental constraint? *J. Exp. Mar. Biol. Ecol.* **1995**, *193*, 147–160. [CrossRef]
53. Shih, C.H.; Kuo, Y.Y.; Chu, T.J.; Chou, W.C.; Chang, W.T.; Lee, Y.C. Eco-environmental impact assessment of artificial beach nourishment on benthic invertebrate community at Anping coast, Taiwan. *China Ocean Eng.* **2011**, *25*, 215–236. [CrossRef]
54. Shao, K.T.; Lai, K.C. The current situation and challenges of Taiwan's marine protected areas. *Mar. Aff. Policy Rev.* **2011**, *1*, 65–90.
55. Statzner, B.; Bis, B.; Dolédec, S.; Usseglio-Polatera, P. Perspectives for biomonitoring at large spatial scales: A unified measure for the functional composition of invertebrate communities in European running waters. *Basic Appl. Ecol.* **2001**, *2*, 73–85. [CrossRef]
56. Statzner, B.; Bady, P.; Dolédec, S.; Schöll, F. Invertebrate traits for the biomonitoring of large European rivers: An initial assessment of trait patterns in least impacted river reaches. *Freshw. Biol.* **2005**, *50*, 2136–2161. [CrossRef]
57. Pearson, T.H.; Rosenberg, R. Macrobenthic succession in relation to organic enrichment and pollution of the marine envi-ronment. Oceanogr. *Mar. Biol.* **1978**, *16*, 229–311.
58. Simberloff, D. Flagships, umbrellas, and keystones: Is single-species management passé in the landscape era? *Conserv. Biol.* **1998**, *83*, 247–257. [CrossRef]
59. Morrison, M.L. *Restoring Wildlife: Ecological Concepts and Practical Applications*; Society for Ecological Restoration International: Island, WA, USA, 2009.
60. Caro, T. *Conservation by Proxy: Indicator, Umbrella, Keystone, Flagship, and Other Surrogate Species*; Island Press: Washington, WA, USA, 2010.
61. Primack, R.B. *Essentials of Conservation Biology*, 5th ed.; Sinauer Associates Inc. Press: Sunderland, MA, USA, 2010.
62. Pander, J.; Geist, J. Ecological indicators for measuring stream restoration success. *Ecol. Indic.* **2013**, *30*, 106–118. [CrossRef]