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Influence of natural and anthropogenic disturbance on terrestrial isopods in coastal wetlands of Mediterranean Chile

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Resumo

Os sistemas húmidos costeiros mediterrânicos constituem ecossistemas de elevada importância ecológica, não apenas pelo seu papel na conservação da biodiversidade, mas também pelas funções ecossistémicas que desempenham, como a regulação do ciclo hidrológico, a proteção costeira, o armazenamento de carbono, etc. No entanto, estes ambientes encontram-se sob crescente pressão antrópica, que ameaça a sua biodiversidade e funcionamento. Estas alterações promovem a degradação da qualidade ecológica dos habitats, modificam a estrutura da vegetação, alteram a composição físico-química do solo e afetam a fauna local. Entre os grupos de invertebrados que habita estes ambientes, os isópodes terrestres (Isopoda: Oniscidea) são de particular interesse, pois desempenham funções ecológicas fundamentais, como a decomposição da matéria orgânica e a reciclagem de nutrientes. Estudos recentes indicam que as comunidades de isópodes nestes ambientes são particularmente sensíveis a perturbações antrópicas e a fatores de stress ambiental, como a salinização e a degradação da estrutura do habitat. No entanto, a sua diversidade e padrões de distribuição nas zonas húmidas costeiras mediterrânicos do Chile permanecem pouco estudados.

Este estudo teve como objetivo avaliar de que forma os diferentes tipos de uso do solo circundante — agrícola, plantações e áreas urbanas — influenciam a diversidade e a estrutura das comunidades de isópodes terrestres em 18 zonas húmidas costeiras na região central do Chile mediterrânico, abrangendo uma área entre os 29º e 33º de latitude sul. A amostragem decorreu na primavera de 2019 e incluiu três zonas por cada zona húmida, distribuídas ao longo de um gradiente de distância da costa (próxima, intermédia e distante da costa). Em cada zona, foram utilizadas cinco armadilhas de queda (pitfall traps) para a captura de isópodes, e avaliadas várias variáveis ambientais, incluindo características do solo (pH, salinidade, teor de humidade, matéria orgânica), da vegetação (cobertura, biomassa, riqueza), e um índice de qualidade ecológica do habitat (ECELS). Para cada local foram calculadas métricas de diversidade taxonómica (abundância total, riqueza bruta e rarefeita, e índice de diversidade de Shannon) e métricas relacionadas com o tamanho corporal dos isópodes capturados (amplitude de tamanhos, diversidade e uniformidade de tamanhos e média geométrica). Adicionalmente, foi realizada uma análise de agrupamento baseada na percentagem de ocupação do solo para classificar os locais em três grupos principais: zona húmidas rodeadas por áreas agrícolas, por plantações florestais e por zonas urbanas. A estrutura ambiental associada a cada grupo foi avaliada através de uma Análise de Componentes Principais (PCA) seguida de análise de agrupamento hierárquico, resultando em três categorias principais: agrícola (A), plantações (P) e urbana (U). Foram depois avaliadas diferenças nas variáveis ambientais e nas métricas de diversidade taxonómica e de tamanho corporal com testes de Kruskal-Wallis, análises multivariadas de ordenação (NMDS), PERMANOVA e análise SIMPER.

No total, foram capturados 23.557 indivíduos pertencentes a 9 espécies de isópodes. Os resultados revelaram que as zonas húmidas rodeadas por áreas agrícolas e de plantações apresentaram abundâncias significativamente superiores de isópodes em comparação com as zonas urbanas. As espécies mais abundantes foram *Benthanoides* sp. e *Tylos* sp., particularmente nas zonas agrícolas e de plantações. Por contraste, as zonas urbanas apresentaram significativamente menos indivíduos. No entanto, não foram encontradas diferenças significativas na riqueza taxonómica ou na diversidade de tamanhos entre os diferentes tipos de uso de solo. Ainda assim, as análises multivariadas (NMDS E

PERMANOVA) mostraram diferenças claras na composição das comunidades entre as categorias de uso de solo, com destaque para a contribuição diferenciada de algumas espécies. Por exemplo, *Benthanoides* sp. foi significativamente mais abundante nas zonas agrícolas, enquanto que *Niambia capensis* foi mais frequente nas zonas urbanas. *Tylos* sp. mostrou-se particularmente abundante nas plantações. Três espécies – *Tylos* sp., *Af.Scyphoniscus* e *Halophiloscia couchii* – foram exclusivas das plantações, contribuindo para a sua distinção face aos outros grupos. As condições ambientais também variaram consideravelmente entre os locais. As zonas urbanas apresentaram valores mais baixos de pH (solos mais ácidos), valores elevados de matéria orgânica e maior cobertura vegetal total (viva e morta), riqueza e biomassa. Por outro lado, as zonas agrícolas e de plantações mostraram uma tendência para valores mais elevados de salinidade e pH mais alcalino. O índice ECELS indicou uma qualidade do habitat geralmente medíocre ou deficiente em todos os locais, com exceção de Punta Teatino (agrícola), que apresentou valores bons em todos os seus pontos de amostragem. A pior classificação foi registada em Zapallar, um local urbano, com um valor "mau" num dos seus pontos. Este estudo demonstra que a composição e abundância das comunidades de isópodes terrestres em zonas húmidas costeiras do Chile são influenciadas tanto pelo uso do solo circundante como pelas condições ambientais locais. Os resultados realçam a sensibilidade destes invertebrados às perturbações antrópicas e destacam a sua utilidade como indicadores ecológicos em programas de monitorização e conservação de zonas húmidas costeiras mediterrânicas.

Palavras-chave: Zonas húmidas mediterrânicas, perturbação antropogénica, condições ambientais, isópodes terrestres, estrutura de tamanho

Abstract

Mediterranean coastal wetlands are ecologically valuable ecosystems under increasing anthropogenic pressures that threatens their biodiversity and ecosystem functioning. Among the invertebrate fauna inhabiting these environments, terrestrial isopods play crucial roles in decomposition processes and nutrient cycling, yet their diversity and distribution patterns in Chilean Mediterranean wetlands remain understudied. Recent research suggests that isopod communities in these wetlands are particularly sensitive to anthropogenic disturbance and environmental stressors, such as salinization and degradation of habitat. This study examined how different land uses — agriculture, plantations and urban areas surrounding the wetlands — and local environmental conditions influence the diversity and structure of terrestrial isopod communities. We sampled 18 coastal wetlands in central Chile and calculated both taxonomic and size-based metrics. We observed significantly higher isopod abundances in wetlands surrounded by agriculture and plantations, compared to urban sites. Size diversity did not differ significantly among the different land use categories. Environmental conditions varied across these wetlands, with urban soils tending to be more acidic and enriched in organic matter, whereas wetlands surrounded by agriculture and plantations were characterized by higher salinity and more alkaline soil conditions. These environmental gradients, along with the occurrence of native and exotic species in different land-use contexts, suggest species-specific ecological adaptations.

Keywords: Mediterranean wetlands, anthropogenic disturbance, environmental conditions, terrestrial isopods, size structure

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List of Abbreviation

A – Agriculture

CONAF – National Forest Corporation

ECELS – Index of Conservation Status of Shallow Lentic Ecosystems

HUM – Soil Humidity

H1 – Hypothesis 1

H2 – Hypothesis 2

Lcov – Total Living cover

MO – Organic Matter

NMDS – Non-Metric Multidimensional scaling

P – Plantation

PCA – Principal Component Analysis

PERMANOVA – Permutational Multivariate Analysis of Variance

Rich – Species Richness

Sal – Salinity

SIMPER – Multivariate Homogeneity of Group Dispersion

Tcov – Total Vegetation cover

U – Urban

USDA NRCS – United States Department of Agriculture, Natural Resources Conservation Service

1. Introduction

Coastal wetlands are dynamic systems at the interface between land and sea with high ecological and economic value. They act as climate (temperature and humidity) stabilizers (Novoa et al., 2020), support great biodiversity and provide multiple ecosystem services that contribute to human wellbeing, e.g. water supply, hydrological regulation, tourism and natural resources, natural hazard protection, wildlife habitats, etc. (Kennish et al., 2014; Alam et al., 2017). Nonetheless, coastal wetlands suffer multiple impacts from human activities, especially in zones where population density is high. This is because human activities, such as land use transformation, drainage, shorelines modifications, non-native species introduction, among others (Maynard & Wilcox, 1997), caused substantial impacts on these ecosystems, mainly as a consequence of habitat fragmentation, contamination and overexploitation. As a result, almost 70% of coastal wetlands worldwide have been lost or are severely degraded by human activities (Neumann et al., 2015). These alterations are decreasing biodiversity, with drastic effects on the ecosystem functioning, exacerbating ecological vulnerability, and sometimes leading to a complete ecosystem collapse (Novoa et al., 2020).

Wetland invertebrates represent most of the wetland biodiversity, playing important roles such as nutrient cycling, and represent food sources for numerous species of fishes and aquatic birds (Maynard & Wilcox, 1997). Thus, they are an important link between the primary producers and decomposers to the higher trophic levels (Kenneth, 1992). Invertebrates respond to physical and chemical habitat modifications primarily through changes in community structure and composition (Saether, 1979; Krieger, 1984; Warwick, 1990; Della Bella & Mancini, 2009). Population fluctuations of many species may provide indication of subtle changes in environmental conditions but may also result from the interaction of local factors with more general environmental changes (Gerlach et al., 2013). In particular, terrestrial isopods are often used as ecological bioindicators as their small size makes them sensitive to local conditions, while their mobility enables them to move in response to changing conditions. They are also considered reliable biological indicators of environmental stress both at the community and species level (Dallinger et al., 1992; Jones & Hopkin, 1997; Paoletti & Hassall, 1999).

Body size is a fundamental biological trait influencing various aspects of animal physiology, ecology, evolution and life history (West et al., 1997). Body size can be affected by several factors including air humidity and soil moisture (Wolters & Ekschmitt, 1997), minerals, temperature, pH, soil salinity and vegetation characteristics (Coccia & Fariña, 2022). Jones and Hopkin (1997) found that in areas contaminated by heavy metals, isopods showed smallest maximum sizes when compared with those of least polluted areas. Mineral availability, particularly calcium and magnesium, plays a crucial role in isopod exoskeleton, making these organisms highly dependent on adequate mineral resources (Hopkins & Read, 1992; Warburg, 1993). Soil acidification can reduce the availability and assimilation of essential minerals, such as calcium, due to altered ion exchange dynamics (Craul & Klein, 1980; Dallinger et al., 1992). Consequently, large-bodied isopods may struggle to meet their calcium requirements in low pH environments, which could lead to a decrease in body size under such conditions. Additionally, smaller body sizes are often linked to the danger of desiccation in drier environments (Entling et al., 2010). According to Coccia et al. (2022), high water salinity or soil water stress in coastal wetlands can favor small size generalist species that are more responsive to disturbances, attenuating their effects.

Community size spectra (the relationship between body size and abundance or biomass), can be strongly altered by ecosystem transformations such as land-use change (Cardillo et al., 2005; Mulder et al., 2008; Brose et al., 2017), affecting energy fluxes and thus ecosystem functioning. Size spectra have been used to assess community processes, ecosystem health and resilience (Petchey & Belgrano, 2010; Atkinson et al., 2021). Potapov et al. (2019) showed that land use conversion from rainforest to plantation was associated with a strong decline in density of the majority of soil fauna functional groups and a prominent shift in the size spectrum, as the biomass of large soil decomposers (earthworms) increased almost tenfold, while abundance of smaller animals strongly decreased. The responses to land conversion may differ according to the species (Magura et al., 2004), e.g., forest specialists may suffer, while species associated with urban habitats may benefit from the disturbance and habitat alteration.

Recently, size-related metrics like size diversity (Shannon index; Pielou, 1969) have been used as alternatives to size spectra for studying community size distributions. This measure, reflecting the abundance of different size classes, has been largely applied to aquatic communities to assess the effects of environmental and human disturbance (Benejam et al., 2018; Sgarzi et al., 2020; Jiménez-Prado & Arranz, 2021). Nonetheless effect on size metrics linked to soil fertilization or salinity have been also observed in terrestrial invertebrates (Coccia & Farina, 2022; Coccia et al., 2022), but these effects are less studied in such systems.

Terrestrial isopods (Arthropoda: Malacostraca: Peracarida) play key roles in ecosystem processes, e.g. nutrient cycling (Khemaissia et al., 2013, 2017), decomposition of organic matter and regulation of microbial food webs (Bouchon et al., 2016) and constitute also a food source for many invertebrate and vertebrate predators (Szlavecz et al., 2018). Understanding how land use transformation surrounding coastal wetlands affects their size and taxonomic structure is relevant to understand responses to disturbance and the overall effects on ecosystem functioning.

The Mediterranean-climate area of central Chile is considered a threatened biodiversity hotspot (Mittermeier et al., 1988). Here coastal wetlands are highly vulnerable, and modifications of natural flood and drought regimes due to human activities (e.g. forestry, drainage) can lead to the loss of a large part of their plant diversity (Correa-Araneda et al., 2012) and consequently affect the existing fauna.

In this study we used taxonomic and size based approaches to investigate how land use change surrounding wetlands and local characteristics affect isopod communities of coastal wetlands in central Chile. We focused on 18 coastal wetlands with different land use (i.e. agriculture, plantation, forest and urban).

Our main hypothesis were:

H1: Isopod size diversity and abundance would be higher in agriculture and plantation areas than in urbanized areas, since urbanization significantly reduces the size diversity and taxonomic structure of isopod communities (Magura et al., 2008);

H2: Isopods inhabiting wetlands with lower soil pH and higher salinity are expected to exhibit smaller body sizes, due to physiological stress and environmental constraints associated with these conditions (Jones & Hopkin, 1997).

2. Material & methods

2.1 Study area

We sampled a total of 18 coastal wetlands in central Chile between 29º and 33º South (Fig. 2.1.1). These sites were selected according to their land use cover and accessibility. Sites were representative of different human land uses, including agriculture, forestry and urbanization. All the selected wetlands were located at the mouth of rivers and with variable connection to the sea.

The study area is characterized by a Mediterranean-type climate (Figueroa et al., 2009) with rainfall concentrated during austral winter (from May to July), with dry summers. Mean annual precipitation ranges from 26.3 to 435 mm (https://explorador.cr2.cl/). However, these areas are influenced by El Niño and La Niña phenomena, which can expand or contract the area influenced by the med-climate (e.g., Di Castri & Hajek, 1976; Luebert & Pliscoff, 2004, 2006). Wetlands vegetation included different lifeforms i.e., herbaceous, grass and shrub, which dominance varied between wetlands. The dominant plant species for the northernmost sites of the arid region (Punta Teatinos, Culebrón, Adelaida/Lagunillas, Pachingo, Litre and Teniente) are *Distichlis spicata* and *Sarcocornia neei*, while towards the south (semi-arid; El Yali and Membrillo) there is a reduction in the cover of *D. spicata* associated with an increase of *Sporobolus densiflora*. Between these two extremes, some marshes showed a decrease in dominance of *S. neei* and *D. spicata* associated with the presence of some ruderal species (*Bacharis linearis*, *Tessaria absinthioides*, *Rumex crispus* and *Eragrotis virescen*es in Salinas Chicas, Chigualoco, Quilimarí, Laguna Verde and Cartagena), prairie plants (*Galega officinalis*, *Agrostis capilaris* in Zapallar, Tunquén and Mantagua), and sand dune plant species (*Carpobrotus* sp. and *Ambrosia chamissonis* in Pichicuy and San Jerónimo).

Sampling was conducted during spring 2019, above the highest level of tidal inundation.

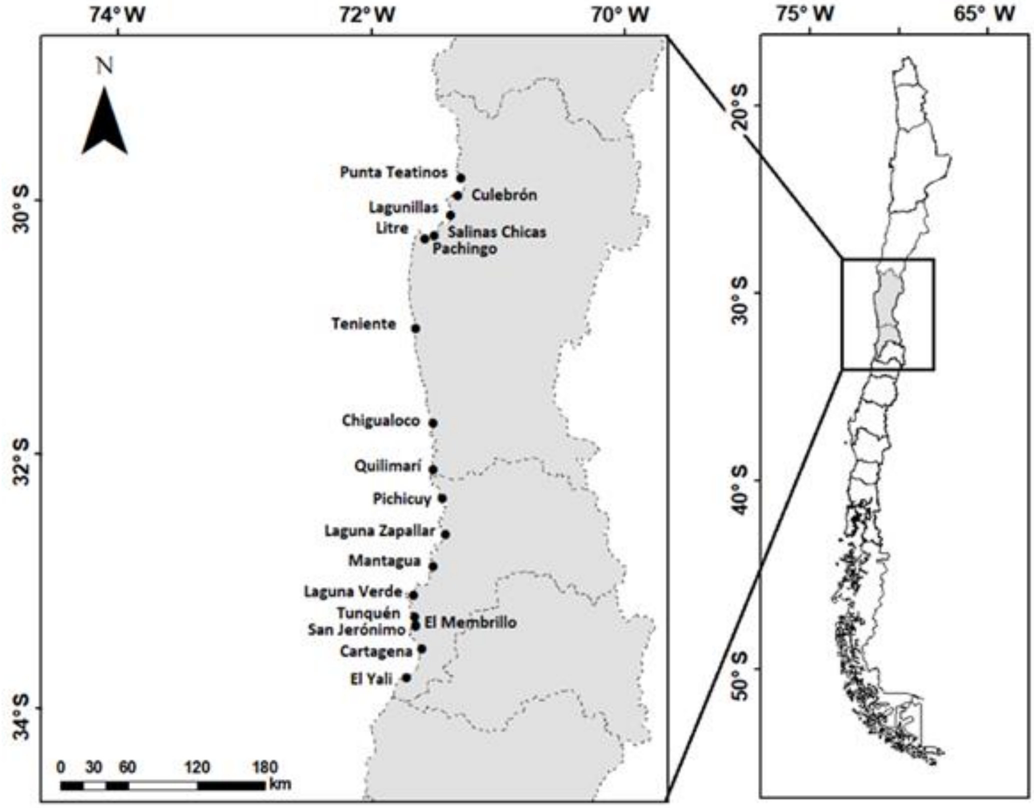


Fig. 2.1.1 Location of the studied wetlands sites along the Chilean Pacific coast. Sites are identified by their names.

2.2 Land use and local conditions

The proportion of each land use cover was calculated within a 1000 m radius buffer around each wetland using the program ArcGis (ESRI) and using thematic digital land-cover maps developed by the CONAF (https://sit.conaf.cl/) for the years 2013 and 2014.

The image classification was done using a set of thematic digital land-cover maps developed by one of the most comprehensive cartographic studies of natural vegetation conducted in Chile known as Catastro (CONAF et al., 1999). Detailed land cover classes considered were 4 of the most general classes: agriculture, plantation, forest and urban.

To evaluate the habitat condition of each sampled zone, we applied the Index of Conservation Status of Shallow Lentic Ecosystems (ECELS), which is based on 5 components: littoral morphology, human activity, water characteristics, emergent vegetation and hydrophytic vegetation (Sala et al., 2004).

The ECELS categories range from bad (0–30 out of 100), deficient (30–50 out of 100), mediocre (50–70 out of 100), good (70–90 out of 100) and very good (90–100 out of 100). Although this index was conceived for wetland within the Mediterranean Region in Europe, it has already been used for Mediterranean wetlands in Chile (Figueroa et al., 2009; Coccia et al., 2022).

2.3 Isopoda sampling and processing

Isopoda were sampled in three sites within each wetland. These sites were located at different distances from the sea (close, middle and far) to depict eventual gradients, if present. Samples were collected using 5 pitfall traps within each zone. These were arranged at the edge of a square of ca. 6 x 6 m, with one more pitfall in the center.

Pitfalls consisted of two plastic cups (each 9 cm diameter and 12.5 cm deep) inserted one inside the other and partially filled with a solution of 50% water and 50% ethylene glycol (preservative). The traps were verified and emptied after 72 hours.

In the laboratory, samples were washed, sieved (250 µm mesh) and preserved in 70% ethanol. Isopods were identified under a stereomicroscope to the lowest practical taxonomic level (genus/species/morfospecies) using the available keys (Peréz-Shultheiss, 2010; Peréz-Shultheiss & Urra, 2020). Then, individuals were photographed under an Olympus SZ61 stereomicroscope (with magnification from 0.67X to 4.5X) connected to a Jenoptik ProgResC5 digital camera and measured for length and width to the nearest 0.1 mm using the digital image analysis software Image J (ver. 1.5). Measurements were taken on all collected individuals on a sampled site except for abundant (>25 individuals) taxa, which were subsampled by measuring 25 randomly selected individuals.

2.4 Species and size diversity of Isopods

A total of four taxonomic metrics were calculated for each zone within each wetland: total abundance, Shannon diversity, raw richness and rarefied richness.

Richness and diversity (exponential Shannon Index) were calculated using the lowest possible taxonomical level (genus or species). Estimations were thus conservative. Since richness can be influenced by the sample size, which varied between zones, rarefied richness was also calculated using the “rarefy” function in R. To standardize the data, richness was adjusted according to the lowest abundance recorded on any site.

A total of four size related variables were measured to quantify the size structure of isopods for each zone within each wetland.

1) Body size range (L_{range}, mm) - the difference between the largest and smallest individuals captured;

2) Size diversity (μ);

3) Size evenness (J_e) and

4) Gmeans.

Variables 2), 3) and 4) were calculated for each zone using individual lengths following the method described by Quintana et al. (2008; 2016) and using the R script available at the <https://limnolam.org>.

Size diversity (μ) is analogous to the Shannon Weiner index but appropriated for continuous variables such as body size (e.g. length or mass). It is based on a non-parametric kernel estimation, which is applicable to any type of size distribution. Data were standardized through division by the sample geometric mean before size diversity calculation. Low values of size diversity indicate a low diversity of isopoda size range with unequal proportions of individuals across the size range. Instead, high values of size diversity indicate a high diversity of isopoda sizes with a more even distribution of individuals along the size distribution (Quintana et al., 2008; Brucet et al., 2017).

Size evenness (J_e) quantifies the uniformity of the size distribution within a community, accounting for both species richness and the relative abundance of individuals across size classes (Moore, 2013). This metric assesses how well the size distribution conforms to a log-normal pattern, indicating whether individuals are evenly spread across different size categories.

Gmeans represents the central tendency within a dataset, capturing the typical size of individuals in a community while minimizing the effect of extreme values. This metric is particularly useful for visualizing the overall structure of a community's size distribution (Roenfeldt, 2018).

2.5 Environmental characteristics

Vegetation characteristics were estimated using a 1 x 1 m quadrat around each pitfall. Within each quadrat, we visually estimated from above the total percentage cover by vegetation, including both live and dead vegetation percentage, and raw vegetation richness. We also measured the height of the tallest branch of the dominant species within each quadrat with a measuring stick.

We collected soil samples (ca. 10 cm large x 15 cm depth) in three locations randomly distributed within the sampled zone. Samples were put in plastic bags and transported to the laboratory for analysis of soil water, organic matter content, and soil salinity. Samples from each location were homogenized before analyses.

We measured water content gravimetrically after weighing, drying (48 hr at 60°C), and then reweighing each soil sample to the nearest 0.001 g using a Precisa XB 320 M balance (Precisa Gravimetrics, Switzerland). Water content was then expressed as the percent difference between initial and final weight of the soil sample. Organic matter was determined for the previously dried samples, which were combusted in a muffle at 500°C for 12 hr. Organic matter was then expressed as the percent difference in weight before and after combustion. Soil pore water salinity was measured by rehydrating dried soil samples in a known volume of distilled water, mixing constantly for 2 minutes, leaving decant for about 30 minutes and measuring the salinity and pH of the resulting supernatant with a Thermo Scientific Orion 4 Star multi parameter probe.

2.6 Statistical analysis

All analyses were conducted in RStudio Version 1.4.1106 (R Development Core Team, 2009), including functions in Vegan, Adespatial, Devtools, Dplyr, Factoextra and FactoMineR, ggplot2 and ggpubr, PairwiseAdonis and Rstatix packages.

To identify groups of wetlands with similar land use characteristics, we first used a PCA on the percentage land use cover of the 4 main land use categories. Then, we used a hierarchical cluster analysis to identify groups of wetlands with similar land use in their surroundings, based on the scores obtained from the PCA.

A second PCA based on the clusters of wetlands with similar land use was performed to assess which local variables were associated with each land use category.

We investigated differences in taxonomic and size-based- metrics between wetland clusters using Kruskal-Wallis tests with Bonferroni adjustment. To visualize differences in community structure between wetland clusters we used a non-metric multidimensional scaling (NMDS) based on the Hellinger-transformed Euclidean distance matrix of invertebrate abundance (i.e. Hellinger distances, Legendre & Gallagher, 2001). Rare species that occurred only once were eliminated to avoid potential bias. We then used a Permutational Multivariate Analysis of Variance with Hellinger distances (PERMANOVA; "ADONIS" in R, Oksanen et al., 2012) to test for differences between wetland clusters. When ADONIS revealed significant differences, we performed a Multivariate Homogeneity of Group Dispersion (SIMPER) analysis to identify which taxa drove those differences.

3. Results

3.1 Wetland land uses

The first principal component of the PCA explained 50.13% of the variance, with the second component explaining 33.14% (Fig. 3.1.1). PC1 was positively correlated with plantation and forest cover, while being negatively correlated with urban land use. PC2 showed a positive correlation with agricultural and forest cover, and a negative correlation with plantation and urban land use.

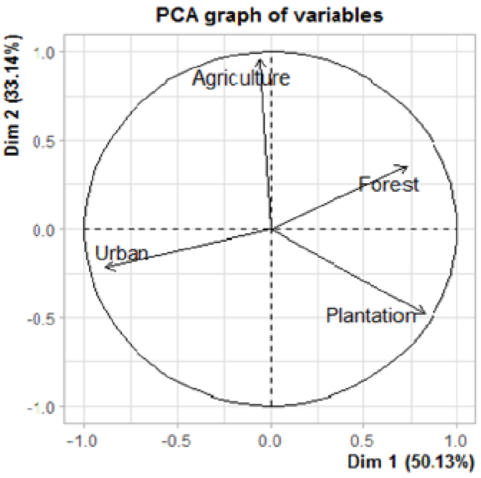


Fig. 3.1.1 PCA plot (the first two axes explain 83.27% of the total variance).

Cluster analysis grouped the wetlands into three distinct categories based on their surrounding land use: agriculture (A), plantation (P), and urban (U) (Fig. 3.1.2). The majority of the wetlands were associated with urban areas (8 sites), followed by plantation (5 sites) and agriculture (3 sites). A small number of sites associated with forested areas were excluded from the analysis due to their limited representation (Table A.1).

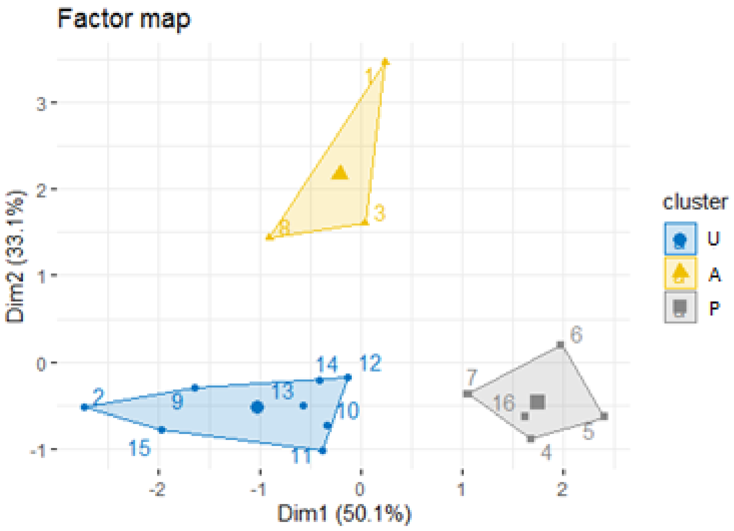


Fig. 3.1.2 Factor-map illustrating the position of the clusters of the three categories – Urban (U), Agriculture (A) and Plantation (P). Numbers 1, 3 and 8 are sites associated with Agriculture wetlands; sites 4, 5, 6, 7 and 16 are associated with plantation wetlands; and sites 2, 9, 10, 11, 12, 13, 14, 15 are associated with urban wetlands. The geometric figures (triangle, circle and square) represent the centroid of each land-use category.

3.2 Local environmental conditions

According to the PCA analyses (Fig. 3.2.1), the first principal component explains 31.7% of the total variation of the data, being mainly associated with total vegetation cover (Tcov), live plant biomass (Lcov) and species richness (Rich). The second component explains 19.9% of the variability and is related to pH, salinity (Sal), water organic content (Hum) and organic matter in the soil (MO). ECELS index is on the origin of the PCA; therefore it is neither related with PC1 nor with PC2.

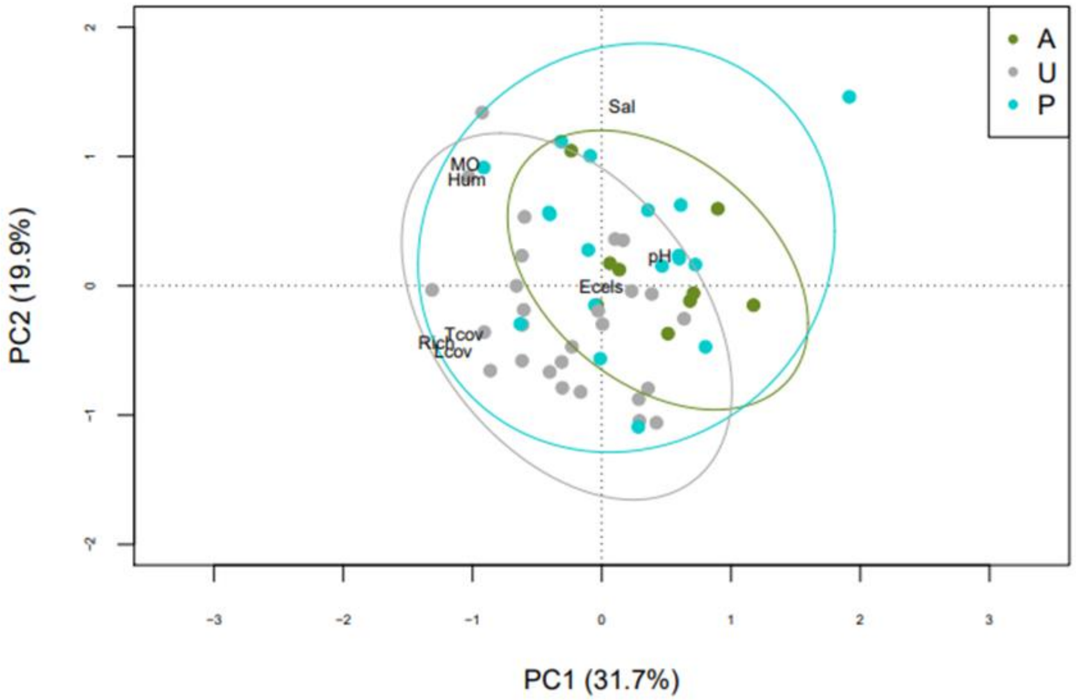


Fig. 3.2.1 PCA plot showing the distribution of three soil categories (A = Agriculture, U = Urban, P = Plantation) based on environmental variables. Tcov= total vegetation cover; Lcov= total living biomass; Rich= total species richness; pH; Sal= salinity; Hum= soil humidity; MO= soil organic matter; Ecels= Index of Conservation Status of Shallow Lentic Ecosystems.

The environmental and ecological conditions varied considerably across the study areas, as shown in Tables A.2 and A.3 (Appendix). Urban sites were characterized by more acidic soils, with the lowest recorded pH value of 5.73, and lower salinity levels (although the highest value was recorded in an urban area, 26.05 mS/µS). On the other hand, there is not a clear distinction in the abiotic conditions between plantation and agricultures sites, even though some site-specific differences exists between them, especially a tendency toward higher salinity in plantation sites.

Regarding vegetation, urban sites were associated with higher total vegetation coverage (both dry and alive), richness and biomass (Lcov) compared to both plantation and agriculture sites (Fig. 3.2.1, Table A.3).

All the three categories had deficient and mediocre values according to the ECELS index. Punta Teatinos, with agriculture land use, was the only wetland with all three sites with good ECELS values. Zapallar, one of the most urbanized wetlands from our study, was the only area with a bad ECELS value in one site (Table A.2).

3.3 Isopoda community and size structure

A total of 23,557 isopod individuals were collected, distributed among 9 species (Table A.4). The most abundant taxa were *Benthanoides* sp., found in every wetland, with 17607 individuals and *Tylos* sp., found in 5 wetlands, with 5018 individuals. They were more abundant in the agriculture (A) and plantation (P) zones, and less numerous in the urbanized ones (U). The less abundant taxa were *Scyphoniscus* sp., *Halophiloscia couchii* and *Porcellionides* sp. with 12, 11 and 15 individuals respectively. Overall, we found significant differences in total isopoda abundance between land use categories (Kruskal-Wallis test, P = 0.0047); both agriculture and plantation categories had more isopoda individuals than the urban category (Fig.3.3.1). Although there were no significant differences in both raw richness and rarefied richness between land use categories (Kruskal-Wallis test, P > 0.05) (Fig.3.3.1), the wetlands with the highest number of species were Salinas Chica (P), Litre (P), San Jerónimo (U) and Cartagena (U), all with 5 species (Table A.4).

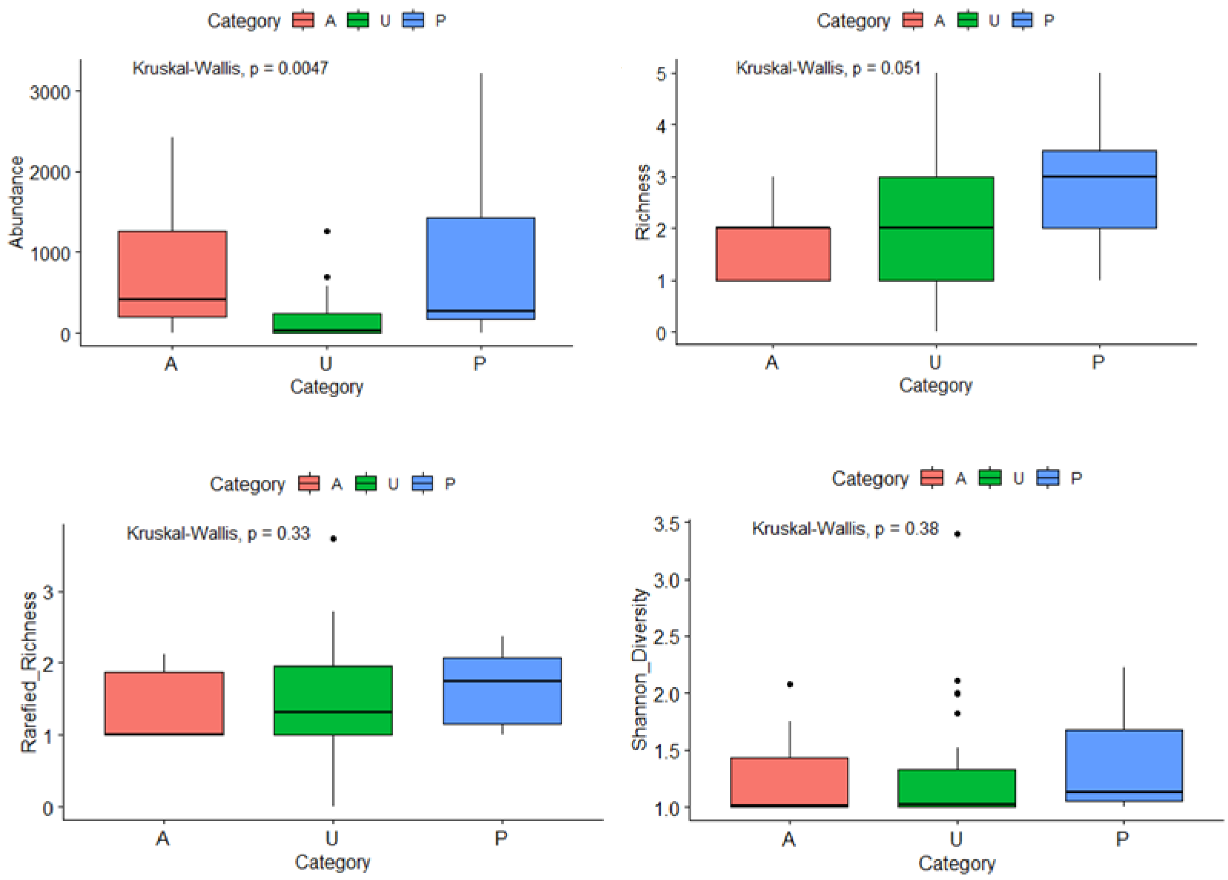


Fig. 3.3.1 Boxplot showing isopoda abundance, richness, rarefied richness and shannon diversity in the three different land use categories: Agriculture (A), Urban (U) and Plantation (P). Each box represents the interquartile range of the taxonomic metrics and the horizontal line indicates the median.

The NMDS analysis showed clearly differences in the isopoda community composition between wetland categories (A, P and U), with a 0.085 stress (<0.1 – great representation in reduced dimensions) (Fig. 3.3.2).

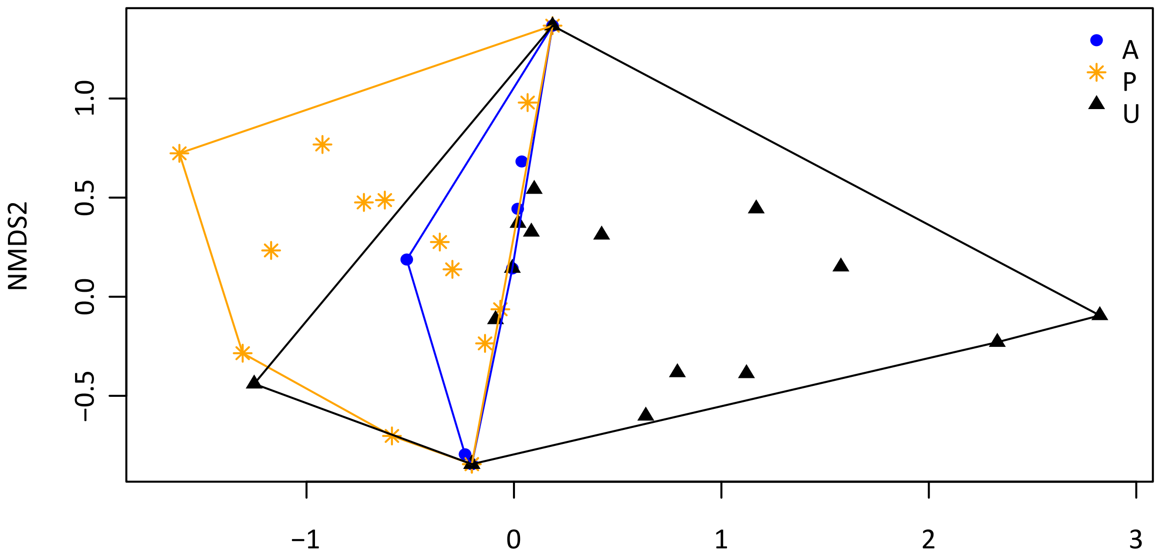
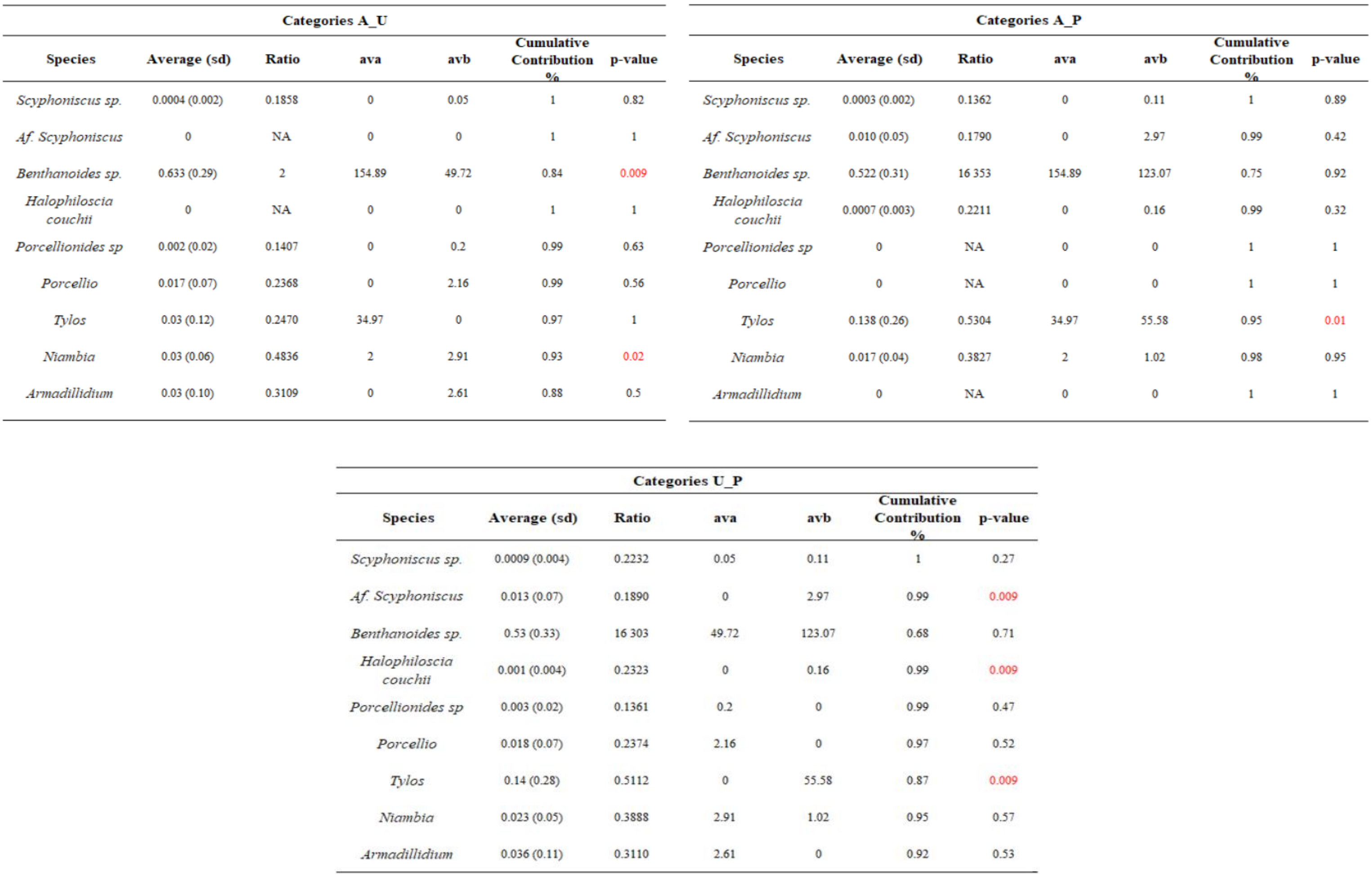


Fig. 3.3.2 Non-metric multidimensional scaling (NMDS) ordination plot representing isopods community composition in different types of land-use wetlands, used to visualize similarities or differences in community composition among samples based on species abundance data. Points closer together indicate more similar communities than points farther apart. Polygons contain the sites for each land use. Blue circles = Agriculture; Yellow stars = Plantation; Black triangles = Urban.

ADONIS results revealed significant differences in community composition between all wetland categories (ADONIS, $P < 0.05$; Pairwise ADONIS, A vs U: $R^2 = 0.06$, $P = 0.002$; A vs P: $R^2 = 0.04$, $P = 0.01$; U vs P: $R^2 = 0.08$, $P = 0.001$). Simper analysis revealed that *Benthanoides* sp. and *Tylos* sp. were the most important taxa driving the differences among land use categories, presenting both the highest average contribution values and also high cumulative contribution (Fig. 3.3.3). Specifically, among the wetland categories, the species that contributed most to the dissimilarity between agriculture and urban wetland sites were *Benthanoides* sp. with higher abundance in agriculture wetlands, and *Niambia capensis* with higher abundance in urban wetlands ($P = 0.009$; $P = 0.029$ respectively). Differences between agriculture and plantation wetlands were driven by *Tylos* sp. showing higher abundance in plantation wetlands ($P = 0.019$). *Tylos* sp., *Af.Scyphoniscus* and *Halophiloscia couchii* contributed more to differences between urban and plantation wetlands, all of them found only in plantation wetlands ($P = 0.009$ for all the three species).



Categories A_U

Species	Average (sd)	Ratio	ava	avb	Cumulative Contribution %	p-value
Scyphoniscus sp.	0.0004 (0.002)	0.1858	0	0.05	1	0.82
Af. Scyphoniscus	0	NA	0	0	1	1
Benthanoides sp.	0.633 (0.29)	2	154.89	49.72	0.84	0.009
Halophiloscia couchii	0	NA	0	0	1	1
Porcellionides sp	0.002 (0.02)	0.1407	0	0.2	0.99	0.63
Porcellio	0.017 (0.07)	0.2368	0	2.16	0.99	0.56
Tylos	0.03 (0.12)	0.2470	34.97	0	0.97	1
Niambia	0.03 (0.06)	0.4836	2	2.91	0.93	0.02
Armadillidium	0.03 (0.10)	0.3109	0	2.61	0.88	0.5

Categories A_P

Species	Average (sd)	Ratio	ava	avb	Cumulative Contribution %	p-value
Scyphoniscus sp.	0.0003 (0.002)	0.1362	0	0.11	1	0.89
Af. Scyphoniscus	0.010 (0.05)	0.1790	0	2.97	0.99	0.42
Benthanoides sp.	0.522 (0.31)	16 353	154.89	123.07	0.75	0.92
Halophiloscia couchii	0.0007 (0.003)	0.2211	0	0.16	0.99	0.32
Porcellionides sp	0	NA	0	0	1	1
Porcellio	0	NA	0	0	1	1
Tylos	0.138 (0.26)	0.5304	34.97	55.58	0.95	0.01
Niambia	0.017 (0.04)	0.3827	2	1.02	0.98	0.95
Armadillidium	0	NA	0	0	1	1

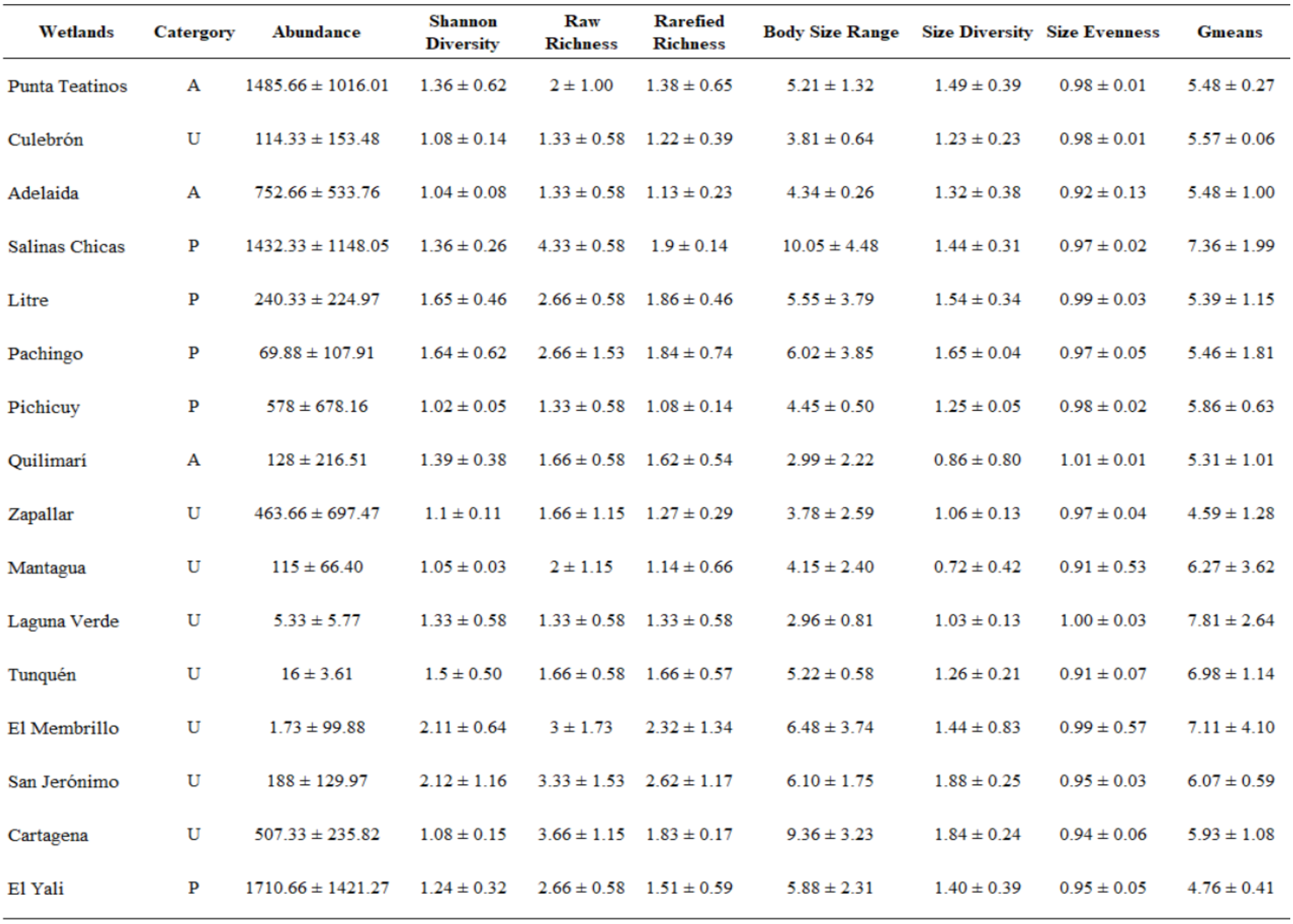
Categories U_P

Species	Average (sd)	Ratio	ava	avb	Cumulative Contribution %	p-value
Scyphoniscus sp.	0.0009 (0.004)	0.2232	0.05	0.11	1	0.27
Af. Scyphoniscus	0.013 (0.07)	0.1890	0	2.97	0.99	0.009
Benthanoides sp.	0.53 (0.33)	16 303	49.72	123.07	0.68	0.71
Halophiloscia couchii	0.001 (0.004)	0.2323	0	0.16	0.99	0.009
Porcellionides sp	0.003 (0.02)	0.1361	0.2	0	0.99	0.47
Porcellio	0.018 (0.07)	0.2374	2.16	0	0.97	0.52
Tylos	0.14 (0.28)	0.5112	0	55.58	0.87	0.009
Niambia	0.023 (0.05)	0.3888	2.91	1.02	0.95	0.57
Armadillidium	0.036 (0.11)	0.3110	2.61	0	0.92	0.53

Fig. 3.3.3 Summary of SIMPER analysis results, used to identify the isopod species that contribute most to the dissimilarity in community composition among land use categories.

We did not find any significant differences between land use categories for all size-based metrics (Kruskal-Wallis test, P > 0.05). However, isopods in urban wetlands showed lower diversity and evenness, but higher gmeans (Table 3.3.1). The sites with the smallest size range of individuals were Quilimarí (A) and Laguna Verde (U) (2.99mm and 2.96mm body size range respectively) and the sites with the biggest individuals were Salinas Chicas (P) and Cartagena (U) (10.05mm and 9.36mm body size range respectively) (Table 3.3.1).

Table 3.3.1 Mean value ± SD of size diversity and taxonomic metrics – abundance, Shannon diversity, raw richness, rarefied richness, body size range, size diversity, size evenness, gmeans – for terrestrial isopods at each studied site.



Wetlands	Catergory	Abundance	Shannon Diversity	Raw Richness	Rarefied Richness	Body Size Range	Size Diversity	Size Evenness	Gmeans
Punta Teatinos	A	1485.66 ± 1016.01	1.36 ± 0.62	2 ± 1.00	1.38 ± 0.65	5.21 ± 1.32	1.49 ± 0.39	0.98 ± 0.01	5.48 ± 0.27
Culebrón	U	114.33 ± 153.48	1.08 ± 0.14	1.33 ± 0.58	1.22 ± 0.39	3.81 ± 0.64	1.23 ± 0.23	0.98 ± 0.01	5.57 ± 0.06
Adelaida	A	752.66 ± 533.76	1.04 ± 0.08	1.33 ± 0.58	1.13 ± 0.23	4.34 ± 0.26	1.32 ± 0.38	0.92 ± 0.13	5.48 ± 1.00
Salinas Chicas	P	1432.33 ± 1148.05	1.36 ± 0.26	4.33 ± 0.58	1.9 ± 0.14	10.05 ± 4.48	1.44 ± 0.31	0.97 ± 0.02	7.36 ± 1.99
Litre	P	240.33 ± 224.97	1.65 ± 0.46	2.66 ± 0.58	1.86 ± 0.46	5.55 ± 3.79	1.54 ± 0.34	0.99 ± 0.03	5.39 ± 1.15
Pachingo	P	69.88 ± 107.91	1.64 ± 0.62	2.66 ± 1.53	1.84 ± 0.74	6.02 ± 3.85	1.65 ± 0.04	0.97 ± 0.05	5.46 ± 1.81
Pichicuy	P	578 ± 678.16	1.02 ± 0.05	1.33 ± 0.58	1.08 ± 0.14	4.45 ± 0.50	1.25 ± 0.05	0.98 ± 0.02	5.86 ± 0.63
Quilimarí	A	128 ± 216.51	1.39 ± 0.38	1.66 ± 0.58	1.62 ± 0.54	2.99 ± 2.22	0.86 ± 0.80	1.01 ± 0.01	5.31 ± 1.01
Zapallar	U	463.66 ± 697.47	1.1 ± 0.11	1.66 ± 1.15	1.27 ± 0.29	3.78 ± 2.59	1.06 ± 0.13	0.97 ± 0.04	4.59 ± 1.28
Mantagua	U	115 ± 66.40	1.05 ± 0.03	2 ± 1.15	1.14 ± 0.66	4.15 ± 2.40	0.72 ± 0.42	0.91 ± 0.53	6.27 ± 3.62
Laguna Verde	U	5.33 ± 5.77	1.33 ± 0.58	1.33 ± 0.58	1.33 ± 0.58	2.96 ± 0.81	1.03 ± 0.13	1.00 ± 0.03	7.81 ± 2.64
Tunquén	U	16 ± 3.61	1.5 ± 0.50	1.66 ± 0.58	1.66 ± 0.57	5.22 ± 0.58	1.26 ± 0.21	0.91 ± 0.07	6.98 ± 1.14
El Membrillo	U	1.73 ± 99.88	2.11 ± 0.64	3 ± 1.73	2.32 ± 1.34	6.48 ± 3.74	1.44 ± 0.83	0.99 ± 0.57	7.11 ± 4.10
San Jerónimo	U	188 ± 129.97	2.12 ± 1.16	3.33 ± 1.53	2.62 ± 1.17	6.10 ± 1.75	1.88 ± 0.25	0.95 ± 0.03	6.07 ± 0.59
Cartagena	U	507.33 ± 235.82	1.08 ± 0.15	3.66 ± 1.15	1.83 ± 0.17	9.36 ± 3.23	1.84 ± 0.24	0.94 ± 0.06	5.93 ± 1.08
El Yali	P	1710.66 ± 1421.27	1.24 ± 0.32	2.66 ± 0.58	1.51 ± 0.59	5.88 ± 2.31	1.40 ± 0.39	0.95 ± 0.05	4.76 ± 0.41

4. Discussion

4.1 Wetland land use characteristics

Our results showed that the majority of the wetlands studied were surrounded by some kind of anthropogenic activities mostly related to urbanization (8 out of 16), followed by plantation and agriculture, while only a few were surrounded by natural forests (Table A.1). These results are in line with Aguayo (2009) showing that the highest percentages of changes in the use of wetland were due to urbanization and plantations. In the last centuries, most of the landscape changes in Chile were related to the expansion of farming to address national needs and the growing international demand for agricultural products. Further, private forestry, subsidized by the State, promoted the accelerated expansion of industrial plantations. As a consequence, there was a drastic decrease in the area covered by native forest, and a simultaneous increase in grasslands, shrublands and large-scale agriculture and forest plantations (Lara, 2012). In the mountain range of the central coast of Chile, extensive areas of wheat replaced the native forest, and the natural vegetation was used as fuel and cleared for the establishment of crops and grasslands (Millán & Carrasco, 1993). Similarly, also coastal wetlands in Mediterranean Chile have been highly modified, largely due to land-use changes driven by agricultural expansion and urbanization (Ramírez et al., 1983, 1995; Ojeda, 1998). This could partly reflect some regional programs that provided subsidies to obtain soils suitable for agriculture or for the establishment of real estate projects (Ramírez et al., 1983; Solervicens & Elgueta, 1994; Ramírez et al. 1995; Ojeda 1998; Hauenstein et al., 1999).

4.2 Local environmental conditions

Our study revealed clear distinctions between the different soil types. Urban wetlands showed acidic soils with lower salinity and higher values of water content, and some sites also had high values of organic matter. These findings partially agree with previous studies showing that wetlands in urban areas often have acidic waters (Novoa, 2020) and elevated organic matter (Craul & Klein, 1980; Short et al., 1986). Industrial activities combined with decomposition of organic matter can contribute to soil acidification (Matias, 2018). Urbanization significantly impacts organic matter dynamics in urban soils, where sites with more vegetation and less artificial substrates exhibit higher organic matter content (Cadorin, Mello, & Montemezzo, 2013), in line with our findings. This vegetation cover could consist mostly of tolerant species as, according to Coccia (2022) saline and acidic conditions tend to limit native vegetation. Although urban sites in our study showed low levels of salinity, possibly due to geographical position of these wetlands, previous studies show that urban areas tend to have more saline soil, due to contamination from industrial and urban waste (Nightingale, 1970; Fulton et al., 1993) and also urban irrigation practices (Miyamoto et al., 2006). However, if the urban area studied has effective wastewater treatment or is less industrialized, this could explain the low salinity levels observed.

Regarding the ECELS values, all categories showed mediocre scores. The lowest value (bad) was recorded in an urban site, which underscores the poor environmental quality of these sites, due to the anthropogenic factors that reduce biodiversity and degrade ecosystem health (Ehrenfeld, 2000; Zedler & Kercher, 2005). This degradation can be related to habitat fragmentation and led to the loss of ecosystem services commonly observed in urbanized wetlands (Hauenstein et al., 2014; Coccia et al., 2022). Moreover, even though nutrient inputs in urban areas may initially stimulate primary productivity, they can also accelerate litter decomposition and disrupt belowground processes, potentially compromising ecosystem stability in the long term (Turner et al., 2011; Cahoon et al., 2021)

The reduced vegetation cover observed in agriculture wetlands may be associated with elevated soil salinity, which can naturally result from environmental factors – e.g. low precipitation, poor natural drainage, presence of saline groundwater close to the surface – but also can be intensified by agricultural practices – including irrigation with low-quality water and inadequate drainage – causing more stress to these already vulnerable environments (Schoups et al., 2005; Rengasamy, 2006). According to Silva et al. (2019), soil salinization negatively impacts native vegetation, reducing biodiversity and increasing susceptibility to desertification. High salinity levels tend to favor the growth of halophytic species, as observed in this study (e.g., *Distichlis spicata* and *Sarcocornia neei*). These sites also exhibited alkaline soils with low moisture content and organic matter, likely due to the reduced vegetation cover, which plays a crucial role in soil moisture retention (Águas, 2012). Several studies (Souza, Cogo & Vieira, 1998; Santos & Tomm, 2003; Pavinato & Rosolem, 2008; Melo et al., 2021) have showed that less disturbed agriculture soils had higher organic matter content which contribute to nutrient retention in the soil surface layer. In contrast, unsustainable practices, such as intensive agriculture and poor crop rotation, may lead to organic matter decrease, limiting soil fauna and energy flow through the detrital food web (Melo et al., 2021).

Variations in salinity levels between sites may be explained by differences in the wetlands geographic position, irrigation methods, drainage efficiency, and soil composition. Studies by Codevasf (2019) and Gheyi et al. (2010) discuss strategies for managing salinity in agricultural environments, emphasizing appropriate irrigation and drainage practices to prevent soil salinization. Also, recent agricultural innovations in Chile emphasize sustainable water management practices that could be relevant for Mediterranean-climate wetlands. Further studies should address whether agricultural management practices, e.g., if irrigation enhance the risk of secondary salinization and thus of soil degradation (Cordeiro, 2001).

Plantation soils exhibited significant reductions in both vegetation cover and species richness, likely due to high salinity and alkaline soils, which can limit plant establishment (Fahim et al., 2021). However, some studies suggest that isopod species richness is not always dependent on vegetation cover, as habitat complexity and species adaptations play key roles (Zimmer 2004). Low-cover areas can still support high species diversity, namely of specialized species (Emden & Williams, 1974; Lawton & Strong, 1981; Altieri, 1984). Some sites in Pachingo and Litre wetlands exhibited no vegetation cover. Despite *D. spicata* being a salt-tolerant species known to thrive in such conditions, its absence may be attributed to low seed germination rates, reliance on rhizome propagation, and hydrological constraints (USDA NRCS, 2010). Indeed *D. spicata* tolerates salinity, but it requires moist soils for establishment. Soils of the plantation sites were humid, likely due to the organic matter content, however a prolonged flooding, desiccation, or soil instability could still have inhibited the presence of *D. spicata* (USDA NRCS, 2010). Additionally, organic matter in plantation soils may influence microbial and detritivore dynamics, playing a key role in decomposition and nutrient cycling (Potapov, 2021).

4.3 Isopoda community and size structure

Our first hypothesis (H1) predicted that isopod abundance and size diversity would be higher in agriculture and plantation wetlands compared to urbanized areas, found a partial support. Although no significant differences were found among the three land-use categories in community size metrics, abundance was significantly higher in agricultural and plantation wetlands. Urbanized wetlands, in contrast, supported lower isopod abundances and exhibited the lowest community structural complexity. *Benthanoides* sp., native to central Chile (Schmalfuss, 2003), was the most abundant taxon and was found across all wetland categories, with the highest values in agriculture wetlands. This highlights its ecological adaptability, suggesting a broad tolerance to varying environmental conditions, including differences in soil pH, salinity, and organic matter (Pérez-Schultheiss, 2009). Similarly, *Tylos* sp., another native isopod (Pérez-Schultheiss, 2009), was the second most abundant species, occurred exclusively in agriculture and plantation wetlands, with particularly high densities in plantations. This species contributed substantially to the differences detected among land-use categories, as it was more abundant in plantation wetlands, suggesting a strong adaptation to physiologically stressful conditions, enabling its persistence in anthropogenically influenced environments. These patterns support H1 and are consistent with studies showing that habitat disturbance alters isopod assemblages, potentially benefiting species with particular ecological traits (Kalisz & Powell, 2004; Pitzalis et al., 2005; Tsukamoto & Sabang 2005; Magura et al., 2008). While forest specialists may decline with urbanization, generalist or urban-adapted species such as *Porcellio scaber* can thrive in these disturbed environments, often in response to increased temperatures and modified microhabitats (Hornung et al., 2007). Exotic species such as *Porcellio scaber*, *Porcellionides* sp., and *Armadillidium* sp. were recorded exclusively in urban sites, although in low abundance. *Armadillidium* sp., being a generalist species (Magura et. al 2008) and a littoral but non-halophile organism (Vandel, 1962), was expected to occur in urban wetlands, where low soil salinity likely provides favorable conditions for its establishment (Khemaissia et al., 2015). However, this species did not emerge as a driving taxa. These findings partially align with Magura et al. (2008), who reported that urban-adapted isopods, such as *P. scaber*, thrive in disturbed environments, although with higher abundances that do not correspond to our results. These patterns may reflect the limited availability of suitable microhabitats and increased environmental stressors such as pollution and habitat fragmentation, which can constrain population densities (Magura et al., 2008). In contrast, the significantly higher isopod abundances in agriculture and plantation wetlands may reflect more favorable conditions for population establishment, such as higher retention of organic matter that could enhance food availability and microbial activity, which are key for sustaining detritivores fauna such as isopods (Loureiro et al., 2006; El-Wakeil, 2015). Although species richness appeared higher in plantation and urban wetlands, compared to agriculture areas, no significant differences were detected in rarefied species richness among land-use categories. Interestingly, Litre wetland, despite lacking any vegetation cover in some sites, exhibited high species richness. This may reflect habitat heterogeneity resulting from shrub removal or exposure, which facilitates colonization by generalist isopods (Vona-Túri & Szmatona-Túri, 2017). The high species richness in some urban wetlands may also be explained by the presence of adaptable generalist and exotic species, such as *Niambia* sp., *Armadillidium* sp., *Porcellio* sp. and *Porcellionides* sp., which thrive in these environments. Likewise, *H. couchii*, another exotic species with high ecological plasticity (Kinahan, 1858; Pérez-Schultheiss et al., 2019), was restricted to plantation wetlands (Salinas Chica, litre and Pachingo) supporting its adaptability to disturbed environments, yet its limited abundance suggests that vegetation structure and microhabitat availability may play a key role in its establishment (NEMESIS, 2025). Little is known about its interactions with different soil types and physiological tolerances. Despite the prevalence and acceleration of urbanization – and its known negative effects on biodiversity – little is understood about whether these impacts manifest similarly across regions (Niemelä et al., 2000). Some studies suggest that soft management in urban green spaces can promote isopod diversity and abundances (Vilisics et al., 2007). However, the increased richness observed in urban wetlands may mask functional homogenization, as generalist and non-native species dominate and displace functionally unique native taxa (Szlavecz et al., 2018).

Our second hypothesis (H2), that smaller body sizes would be associated with low soil pH and high salinity due to physiological constraints did not found support. We found no significant correlation between isopod body-size range and soil type, which aligns with previous findings indicating no consistent global pattern in body size variation among isopods (Karagkouni et al., 2016). However, it contradicts studies such as Wolters and Ekschmitt (1997), which suggest that soil moisture may influence body size. In this study, the smallest individuals were found in urban wetlands like Laguna Verde, and also in agricultural wetlands like Quilimarí. These environments may impose physiological stress that limits growth, as previously reported in studies of metal-polluted sites where isopods exhibited reduced size and survival (Donker et al., 1993; Jones & Hopkin, 1997). These patterns support the hypothesis that energy trade-offs under stress may prioritize survival overgrowth (Sibly & Calow, 1989). On the other hand, the largest individuals were observed in plantation wetlands such as Salinas Chica – drier and more alkaline – supporting the idea that larger body sizes confer desiccation resistance due to lower surface-to-volume ratios (Remmert, 1981; Olalla-Tárraga et al., 2009; Entling et al., 2010). Although isopods are generally more abundant in alkaline soils, body size does not appear to be systematically affected by soil acidity (Wolters & Ekschmitt, 1997), as both small and large individuals were recorded in acidic urban areas. This highlights the complexity of interpreting size-related patterns. Overall, the lack of consistent size patterns across land-use types suggests that size-based and taxonomic responses may be decoupled in disturbed systems (Coccia & Farina, 2022), and that body size may be shaped by phylogenetic or life-history constrains rather than directly by abiotic variables (Potapov, 2021).

5. Conclusion

Despite the high number of endemic species in Chile, its native vegetation has been strongly modified by anthropogenic disturbances such as agricultural activity and urban expansion (Figueroa et al., 2013). Growing urbanization is exerting greater pressure on the environment, especially around urban centers. In our study, urban areas although supporting richer plant communities, faced greater soil acidification had smaller sized isopods and a higher presence of invasive species. This suggests that urban wetlands exhibit signs of ecological stress, which can result in reduced ecosystem functioning. In contrast, plantations and agriculture areas showed higher salinity and alkalinity levels, limiting the vegetation cover but hosted both small and large isopods. Larger body sizes can confer physiological advantages in such environments — such as reduced water loss — but may also indicate simplified communities adapted to stress, rather than diverse, resilient ecosystems.

These findings emphasize the need for sustainable management policies to preserve the coastal wetlands of Chile, especially in light of increasing urbanization. Implementing protected areas and promoting sustainable land-use practices could help mitigate some of the adverse effects observed in urban and plantation areas – e.g. the presence of invasive species, smaller body sizes, that are linked to stress – by maintaining habitat quality, supporting native biodiversity, and promoting more balanced invertebrate communities (Zedler & Kercher, 2005; Ministerio de Medio Ambiente, 2010).

Our results showed that terrestrial isopods respond differently to land-use change depending on the metric used. This highlights the value of combining taxonomic and size-based approaches to better assess biodiversity and ecosystem responses, and reveals the complexity of ecological responses in

Mediterranean coastal wetlands. Incorporating size-based indicators may improve future monitoring and inform conservation strategies in human-impacted landscapes. In a rapidly changing landscape like central Chile, integrating these perspectives is essential to understand ecological resilience and guide effective wetland conservation.

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APPENDIX

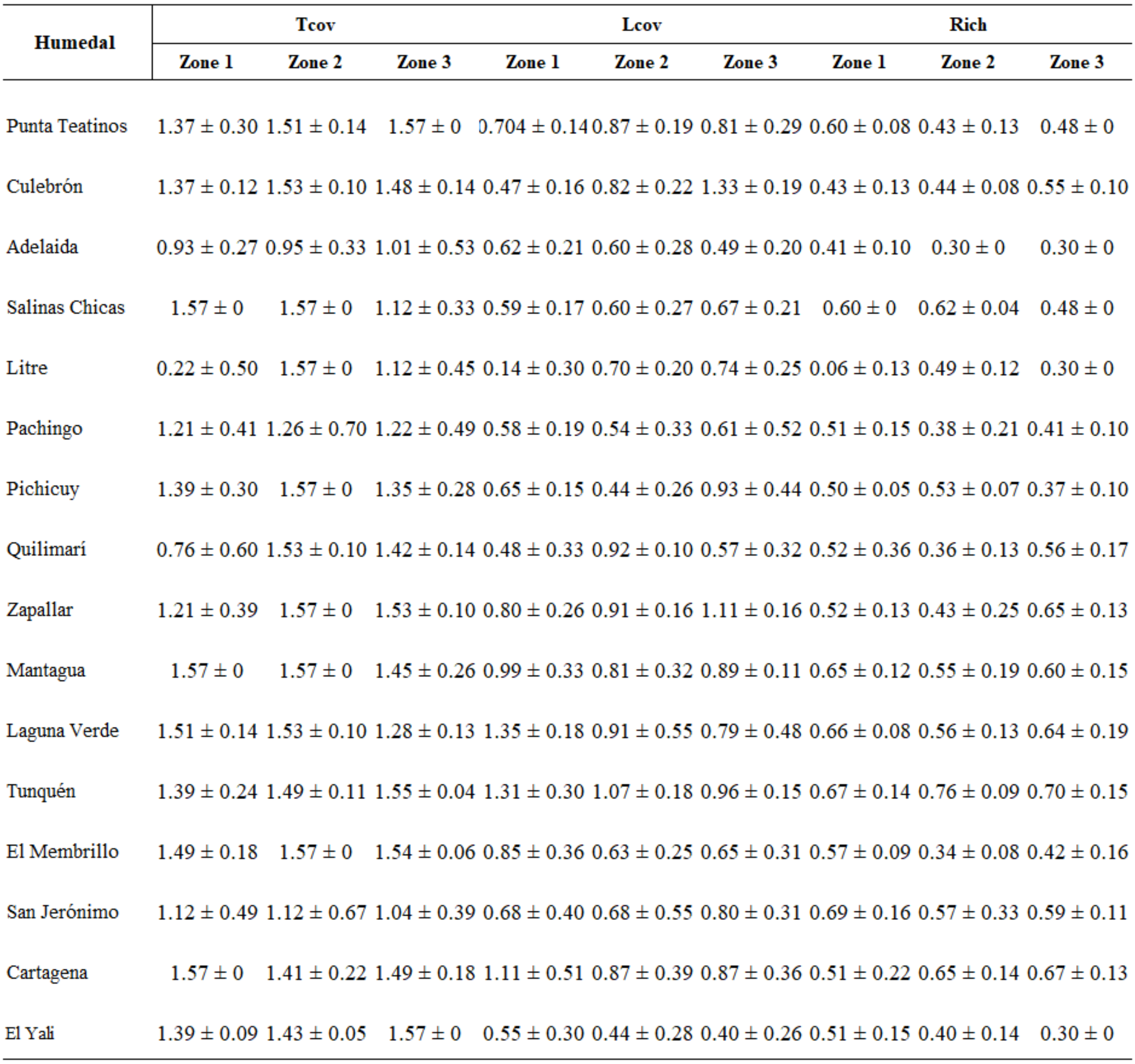
Table A.1. Percentage of land use types (A = agriculture; P = plantation and U = urban) surrounding each of the 16 coastal wetlands studied in Mediterranean Chile.

Wetlands	**Category**	**% of Land Use**
Punta Teatinos	A	15.81
Culebrón	U	45.47
Adelaida	A	10.72
Salinas Chicas	P	21.96
Litre	P	25.58
Pachingo	P	12.81
Pichicuy	P	13.77
Quilimarí	A	8.39
Zapallar	U	43.25
Mantagua	U	5.01
Laguna Verde	U	31.3
Tunquén	U	22.47
El Membrillo	U	23.33
San Jerónimo	U	28.67
Cartagena	U	43.91
El Yali	P	19.28
Chigualoco	F	41
Teniente	F	43.08

Table A.2. Mean values of ECELS and environmental variables (pH, salinity, water content, and organic matter) measured in the three sampled zones of each wetland.

Humedal	**ECELS**			**pH**			**Salinity (mS/uS)**			**% Hum**			**% MO**		
	Zone 1	**Zone 2**	**Zone 3**	**Zone 1**	**Zone 2**	**Zone 3**	**Zone 1**	**Zone 2**	**Zone 3**	**Zone 1**	**Zone 2**	**Zone 3**	**Zone 1**	**Zone 2**	**Zone 3**
Punta Teatinos	71	70	73	9.54	9.08	8.74	5.21	16.73	1.4	7.3	23.39	18.26	1.98	7.29	1.8
Culebrón	42	61	55	8.19	9.68	10.09	1.88	6.54	0.46	0.98	18.6	23.31	1.02	1.7	0.81
Adelaida	72	62	60	8.4	8.15	8.56	1.49	11.28	1.48	4.39	0.4	0.17	1.21	2.82	0.74
Salinas Chicas	65	48	53	8.48	9.13	9	3	3.18	8.93	53.65	65.33	4.27	4.47	2.85	0.56
Litre	63	63	68	8.56	8.27	8.93	6.66	16.37	3.99	13.18	36.52	6.03	2.79	6.39	0.93
Pachingo	47	51	46	7.98	8.93	8.84	13.74	4.33	1.06	0.89	6.05	0.14	4.68	1.35	0.6
Pichicuy	59	55	61	6.65	6.47	7.11	0.05	0.09	0.09	0.59	0.24	0.18	1.19	1.45	0.28
Quilimarí	38	68	53	7.37	7.7	6.79	0.88	1.45	0.08	9.84	15.93	0.73	2.5	1.89	0.26
Zapallar	31	21	34	8.18	5.73	8.46	5.2	26.05	1.05	4.8	46.62	11.54	1.15	12.16	1.12
Mantagua	61	58	63	7.91	7.96	7.63	0.12	3.82	0.33	29.32	26.57	4.72	2.15	9.56	1.4
Laguna Verde	54	66	48	7.11	6.39	6.39	0.15	0.15	0.16	24.05	7.81	4.15	3.65	10.92	2.95
Tunquén	62	71	66	7.39	7.03	8.76	0.47	0.35	6.85	24.36	42.97	32.45	6.46	12.38	15.53
El Membrillo	56	38	66	6.04	5.93	6.99	1.16	6.11	0.57	20.67	7.47	20.95	8.19	3.82	1.44
San Jerónimo	65	42	32	6.5	7.13	6.46	3.13	0.74	0.04	49.06	32.91	0.57	16.84	2.8	0.3
Cartagena	60	62	51	7.71	7.86	8.86	0.95	2.48	1.29	24.16	9.87	11.32	4.86	11.91	1.33
El Yali	58	38	48	6.63	5.88	6.81	3.57	9.28	1.62	17.16	30.34	10.06	4.07	6.88	2.86

Table A.3. Mean ± SD of ecological environmental conditions (Tcov= total vegetation cover; Lcov= live plant biomass and Rich= plant species richness across the 3 zones per site.



Humedal	**Tcov**			**Lcov**			**Rich**		
	Zone 1	**Zone 2**	**Zone 3**	**Zone 1**	**Zone 2**	**Zone 3**	**Zone 1**	**Zone 2**	**Zone 3**
Punta Teatinos	1.37 ± 0.30	1.51 ± 0.14	1.57 ± 0	0.704 ± 0.14	0.87 ± 0.19	0.81 ± 0.29	0.60 ± 0.08	0.43 ± 0.13	0.48 ± 0
Culebrón	1.37 ± 0.12	1.53 ± 0.10	1.48 ± 0.14	0.47 ± 0.16	0.82 ± 0.22	1.33 ± 0.19	0.43 ± 0.13	0.44 ± 0.08	0.55 ± 0.10
Adelaida	0.93 ± 0.27	0.95 ± 0.33	1.01 ± 0.53	0.62 ± 0.21	0.60 ± 0.28	0.49 ± 0.20	0.41 ± 0.10	0.30 ± 0	0.30 ± 0
Salinas Chicas	1.57 ± 0	1.57 ± 0	1.12 ± 0.33	0.59 ± 0.17	0.60 ± 0.27	0.67 ± 0.21	0.60 ± 0	0.62 ± 0.04	0.48 ± 0
Litre	0.22 ± 0.50	1.57 ± 0	1.12 ± 0.45	0.14 ± 0.30	0.70 ± 0.20	0.74 ± 0.25	0.06 ± 0.13	0.49 ± 0.12	0.30 ± 0
Pachingo	1.21 ± 0.41	1.26 ± 0.70	1.22 ± 0.49	0.58 ± 0.19	0.54 ± 0.33	0.61 ± 0.52	0.51 ± 0.15	0.38 ± 0.21	0.41 ± 0.10
Pichicuy	1.39 ± 0.30	1.57 ± 0	1.35 ± 0.28	0.65 ± 0.15	0.44 ± 0.26	0.93 ± 0.44	0.50 ± 0.05	0.53 ± 0.07	0.37 ± 0.10
Quilimarí	0.76 ± 0.60	1.53 ± 0.10	1.42 ± 0.14	0.48 ± 0.33	0.92 ± 0.10	0.57 ± 0.32	0.52 ± 0.36	0.36 ± 0.13	0.56 ± 0.17
Zapallar	1.21 ± 0.39	1.57 ± 0	1.53 ± 0.10	0.80 ± 0.26	0.91 ± 0.16	1.11 ± 0.16	0.52 ± 0.13	0.43 ± 0.25	0.65 ± 0.13
Mantagua	1.57 ± 0	1.57 ± 0	1.45 ± 0.26	0.99 ± 0.33	0.81 ± 0.32	0.89 ± 0.11	0.65 ± 0.12	0.55 ± 0.19	0.60 ± 0.15
Laguna Verde	1.51 ± 0.14	1.53 ± 0.10	1.28 ± 0.13	1.35 ± 0.18	0.91 ± 0.55	0.79 ± 0.48	0.66 ± 0.08	0.56 ± 0.13	0.64 ± 0.19
Tunquén	1.39 ± 0.24	1.49 ± 0.11	1.55 ± 0.04	1.31 ± 0.30	1.07 ± 0.18	0.96 ± 0.15	0.67 ± 0.14	0.76 ± 0.09	0.70 ± 0.15
El Membrillo	1.49 ± 0.18	1.57 ± 0	1.54 ± 0.06	0.85 ± 0.36	0.63 ± 0.25	0.65 ± 0.31	0.57 ± 0.09	0.34 ± 0.08	0.42 ± 0.16
San Jerónimo	1.12 ± 0.49	1.12 ± 0.67	1.04 ± 0.39	0.68 ± 0.40	0.68 ± 0.55	0.80 ± 0.31	0.69 ± 0.16	0.57 ± 0.33	0.59 ± 0.11
Cartagena	1.57 ± 0	1.41 ± 0.22	1.49 ± 0.18	1.11 ± 0.51	0.87 ± 0.39	0.87 ± 0.36	0.51 ± 0.22	0.65 ± 0.14	0.67 ± 0.13
El Yali	1.39 ± 0.09	1.43 ± 0.05	1.57 ± 0	0.55 ± 0.30	0.44 ± 0.28	0.40 ± 0.26	0.51 ± 0.15	0.40 ± 0.14	0.30 ± 0

Table A.4. List of total abundance of terrestrial isopods (Isopoda: Oniscidea) collected during the 2019spring survey in the 16 studied wetlands.

Species **Wetlands**	***Scyphoniscus sp.***	***Af. Scyphoniscus***	***Benthanoides sp.***	***Halophiloscia couchii***	***Porcellionides sp***	***Porcellio***	***Tylos***	***Niambia***	***Armadillidium***	**Total**
Punta Teatinos	0	0	3142	0	0	0	1294	21	0	4457
Culebrón	0	0	339	0	0	0	0	3	0	342
Adelaida	0	0	2252	0	0	0	0	6	0	2258
Salinas Chicas	3	0	748	7	0	0	3534	5	0	4297
Litre	5	0	645	2	0	0	59	10	0	721
Pachingo	0	0	237	2	0	0	127	7	0	373
Pichicuy	0	0	1730	0	0	0	0	4	0	1734
Quilimarí	0	0	337	0	0	0	0	47	0	384
Zapallar	2	0	1325	0	0	1	0	66	0	1394
Mantagua	0	0	114	0	0	0	0	1	0	115
Laguna Verde	0	0	14	0	1	0	0	0	1	16
Tunquén	0	0	39	0	0	0	0	0	9	48
El Membrillo	0	0	61	0	0	0	0	4	108	173
San Jerónimo	0	0	344	0	14	122	0	14	70	564
Cartagena	2	0	1394	0	0	35	0	125	3	1559
El Yali	0	199	4886	0	0	0	4	33	0	5122
Total	12	199	17607	11	15	158	5018	346	191	23557