*diversity*

MDPI

*Article*

# Do Ecological Patterns Persist in Highly Impacted Urban Wetlands? A Spatiotemporal Analysis of Aquatic Macrophytes and Limnological Variability in a Peruvian Coastal Wetland

**Flavia Valeria Rivera-Cáceda [1], José Antonio Arenas-Ibarra [1,2,*] and Sofía Isabel Urrutia-Ramírez [2]**

[1] Faculty of Environmental Science, Marine Biology Department, Universidad Científica del Sur, Lima 15842, Peru; 180000527@cientifica.edu.pe
[2] Faculty of Biological Sciences, Universidad Ricardo Palma, Lima 15039, Peru; sofia.urrutia@urp.edu.pe
* Correspondence: jarenas@cientifica.edu.pe

## Abstract

Urban coastal wetlands along the Peruvian Pacific coast are increasingly affected by urban expansion, pollution, and hydrological alterations, compromising their ecological integrity. In this context, the spatiotemporal variation of the aquatic macrophyte community and its relationship with limnological conditions and drivers of change were evaluated in the Santa Rosa wetland (Chancay, Lima). The objective is to evaluate the spatiotemporal variation of the aquatic macrophyte community in the Santa Rosa wetland and analyze its relationship with physicochemical limnological variables and drivers of change. Sampling was conducted during two contrasting hydrological seasons in 2022: T1 (low-water season) and T2 (high-water season), at six sampling points (P1–P6). Physicochemical variables (water depth, temperature, pH, conductivity, total dissolved solids—TDS, total suspended solids—TSS, dissolved oxygen—DO, turbidity, nitrate—$NO_3^-$, ammonium—$NH_4^+$, phosphate—$PO_4^{3-}$, and dissolved organic matter—DOM) were measured, and the relative abundance of aquatic macrophytes was evaluated. Drivers of change were identified through direct observation and a structured matrix, with phosphate a PCoA performed to summarize spatiotemporal trends. Data were analyzed using Principal Component Analysis (PCA), Co-inertia analysis, and Multi-Response Permutation Procedures (MRPP). Significant spatiotemporal variation was observed in physicochemical parameters ($p < 0.05$), with moderate covariation between the two matrices (RV = 0.47). A total of ten aquatic macrophyte species were recorded, with higher abundance of *Pontederia crassipes* and *Pistia stratiotes* in T1, and *Hydrocotyle ranunculoides* and *Bacopa monnieri* in T2. The most relevant drivers of change were solid waste, livestock grazing, organic contamination, and urban expansion. Spatial heterogeneity was observed in the drivers of change affecting the Santa Rosa wetland, forming a mosaic of areas with different impact profiles. Despite multiple anthropogenic pressures, the Santa Rosa wetland maintains a limnological structure and a functionally coupled macrophyte community, suggesting that essential ecological processes are maintained within the temporal scope of this study. The observed covariation between physicochemical conditions and vegetation confirms the persistence of essential ecological processes, even within an altered urban context. This study demonstrates that integrating biotic components, limnological variables, and drivers of change is fundamental to understanding and monitoring the ecological dynamics of urban wetlands along the Peruvian coast.

Check for updates

Academic Editor: Lele Liu

Received: 1 March 2026
Revised: 28 March 2026
Accepted: 30 March 2026
Published: 7 April 2026

**Copyright:** © 2026 by the authors. Licensee MDPI, Basel, Switzerland. This article is an open access article distributed under the terms and conditions of the Creative Commons Attribution (CC BY) license.

**Keywords:** aquatic plants; drivers of change; physical-chemical limnology; spatiotemporal variation; Santa Rosa wetland

## 1. Introduction

Wetlands are highly productive ecosystems that provide essential ecosystem services, such as hydrological regulation, pollutant purification, and habitat provision for a wide variety of species, particularly adapted to their hydrological dynamics [1]. They also act as carbon sinks and store $CO_2$ in their biomass and sediments, thereby helping to mitigate climate change [2]. Their ability to retain sediments and nutrients prevents eutrophication, leading them to be referred to as the "kidneys of the landscape" [3]. These ecosystems cover between 8.9% and 9.5% of the Earth's surface and are home to remarkable biological diversity, including endemic and endangered species [4]. However, since 1700, nearly 3.4 million $km^2$ of wetlands have been lost, mainly due to land use change; in the last 150 years, more than 50% have been degraded by urban and agricultural expansion, while pollution from fertilizers and pesticides has deteriorated water quality and affected biodiversity, including aquatic macrophyte habitats [5–8].

The ecological functioning of wetlands is intrinsically linked to the seasonality of the hydrological cycle, physicochemical limnology of the water, and structure of aquatic macrophyte communities [9–12]. Hydrological regimes regulate seasonal fluctuations in water depth, generating variations in physicochemical limnology, redox conditions, and biogeochemical cycles that control nutrient availability [11,13,14]. Water hydrochemistry and seasonality modulate pH, conductivity, salinity, and temperature, thereby affecting the biotic composition and the capacity of wetlands to purify water, maintain biodiversity, and provide ecosystem services [15,16].

The aquatic macrophyte community includes freshwater photosynthetic organisms that are visible to the naked eye; following the life-form classification of Chambers and Maberly [17], freshwater plants can be categorized into several functional groups based on their habit, position on the water column, and access to resources: emergent species, amphibious and floating-leaved species, pleustophytes, and submerged species. The role of aquatic macrophytes in ion absorption and transport between sediment and the water column is key to the nutrient cycle and carbon dynamics, influencing dissolved oxygen concentration, nutrient availability, and water quality [9,18,19]. Likewise, their primary production supports multiple trophic levels, and their physical structures provide habitats, shelters, and substrates for various organisms [19,20]. This multifunctionality contributes to ecosystem stability and resilience to disturbances [21,22].

The composition, distribution, and abundance of aquatic macrophytes respond to interactions among abiotic, biotic, and anthropogenic factors. Among the main hydrological factors, the intensity, duration, and frequency of high and low water levels determine water volume and habitat availability [12,19,21]. Similarly, variability in nutrients, pH, turbidity, and conductivity, together with biotic processes such as competition and herbivory, regulates the richness and dominance of macrophyte species [18,20,23,24]. Eutrophication, fragmentation, and anthropogenic impacts alter these processes to varying degrees [22,25–27].

The wetlands of the Peruvian Pacific are unique ecosystems in the arid subtropical desert, shaped by the interaction between natural processes, such as marine intrusions, El Niño events (ENSO), and exfiltration from coastal aquifers, and human activities, including urban and agricultural runoff, resulting in a complex and dynamic hydrological regime [28–31]. El Niño events are particularly significant geomorphological drivers in this

context, as the marked increase in river discharge and suspended sediment load associated with warm ENSO phases causes overbank flooding that reshapes coastal geomorphology, alters wetland boundaries, and may create new water bodies by inundating coastal depressions [32]. The seasonal hydrology of the Peruvian Pacific coastal wetlands follows a delayed recharge–exfiltration cycle. Rainfall in the Andean headwaters during the austral summer (December–March) recharges the unconfined coastal aquifer via river infiltration and percolation from irrigation canals. Several months later, this groundwater reaches the coast and sustains high water levels between July and September [28–31,33,34]. This lag produces two hydrological phases: a low-water phase (T1), with direct surface inputs and episodic discharge pulses in some sectors, and a high-water phase (T2), when phreatic exfiltration raises the water table and increases water depth across the wetland. The Santa Rosa wetland, the subject of this study, is one of the most important on the central Peruvian coast from a scientific perspective and is characterized by the highest reported floristic diversity among the coastal wetlands of the Department of Lima, with well-structured communities representative of different functional types of macrophytes [35–37]. Although studies have examined the diversity of the Peruvian Pacific wetlands, primarily flora and avifauna [38–40], limnological knowledge, particularly regarding the relationships between water physicochemical parameters and biota, remains poorly resolved. Previous studies addressing the physicochemical characteristics of water in this system have been spatially restricted and temporally discontinuous, precluding robust assessments of seasonal and interannual variability [38–40], and studies that integrate relationships with aquatic biota are mostly limited to benthic invertebrates [41–43]. While some relationships between physicochemical limnology and aquatic macrophytes have been explored [44,45], the role of physicochemical parameters as determinants of the structure of this community and its spatiotemporal dynamics in urban wetlands represents a critical information gap, which is particularly relevant given the increasing degradation of these environments by anthropogenic activities that compromise their diversity and ecosystem services [35,46–49].

In this context, the present study aimed to evaluate the spatiotemporal dynamics of the aquatic macrophyte community in the Santa Rosa wetland during the low-water season (March) and high-water season (September) of 2022, determine its relationship with water physicochemical parameters, and characterize the most important drivers of change. We hypothesize that the contrasting hydrological seasons—low-water season (T1) characterized by reduced water depth and increased ionic concentration, high-water season (T2) driven by aquifer exfiltration with greater depth and ionic dilution—drive significant variation in water physicochemical parameters and in the composition and percent cover of aquatic macrophyte assemblages, with life forms turnover reflecting adaptive responses to these seasonal hydrological filters.

## 2. Materials and Methods

### *2.1. Study Area*

The Santa Rosa wetland is located in the district of Chancay, Huaral, Lima region, Peru (Figure 1). Bordered by El Cascajo (north) and Salinas (east) hills, the agricultural lands of Peralvillo and Salinas Altas (south), and the Pacific Ocean (west), it includes a main lake of approximately 17.5 ha and secondary wetlands [50,51]. The climate is very dry and semi-warm, with mean annual temperatures ranging from 18 to 19 °C, absolute maxima of 25–28 °C, and minima of 12–15 °C [45,52]. Annual precipitation is low, averaging 18 mm and ranging between 0 and 36 mm; from April to December, light drizzle, persistent cloud cover, and high relative humidity (81–97%) predominate, particularly from June onward. From January to March, solar irradiance increases and south-southwesterly winds prevail under the influence of the Humboldt Current, with mean velocities of 8–12 km/h [45,52].

Its hydrological regime depends on an irrigation canal fed by the Chancay River and on groundwater exfiltration, which is recharged by agricultural irrigation [31,50]. The maximum water depth occurs between August and October, then decreases towards austral summer, when the Chancay River floods slightly raise it [42]. Additionally, the wetland receives episodic discharges from an irrigation canal derived from the Chancay River during the austral summer, contributing to short-term hydrological variability [45].

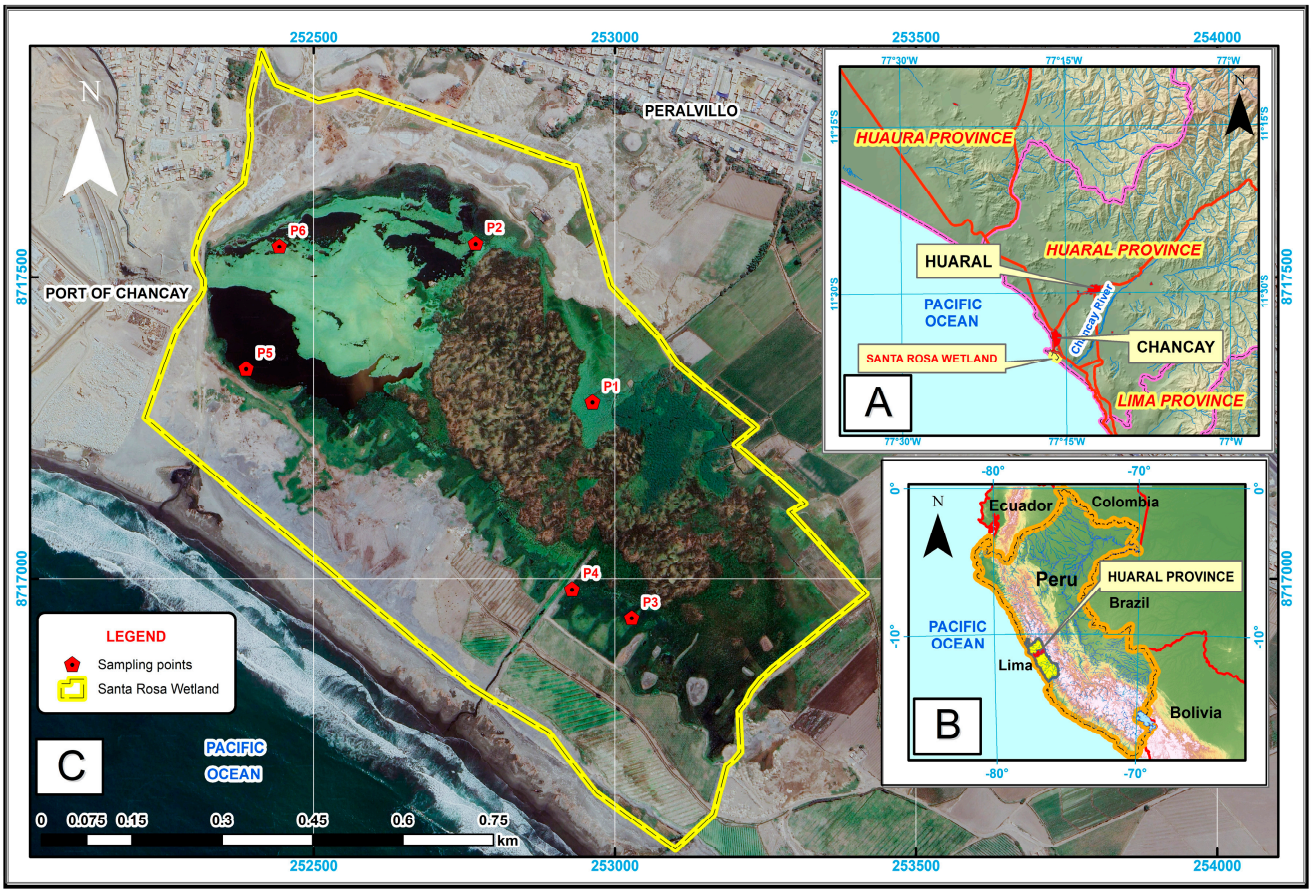



**Figure 1.** Location of the study area. (**A**,**B**) Geographic location in the province of Huaral, district of Chancay, Lima, Peru. (**C**) Santa Rosa wetland environmental conservation area with sampling points (P1–P6).

The wetland features diverse habitats and plant formations: the main body of water; reed beds dominated by *T. domingensis* and *S. americanus*; beach areas; unvegetated areas; mixed meadows associated with agricultural and urban areas; cleared areas affected by human activities; and shrubby areas [53]. Since 2020, it has been classified as an Environmental Conservation Area under municipal management (Ordinance No. 013-2020-MPH-CM).

### 2.2. Sampling Design

Six sampling points (P1–P6) were established (Figure 1). Sampling was conducted during two contrasting periods of the hydrological cycle: March 2022 (low-water season, T1) and September 2022 (high-water season, T2).

P1 is located in an area influenced by the irrigation canal derived from the Chancay River, which receives agricultural and urban effluents, with a predominance of *Pontederia crassipes* (Mart.). P2 was located at the eastern end of the main lagoon and is covered with *Bacopa monnieri* (L.) Wettst and *P. stratiotes*. P3 is a seasonally saturated area with heterogeneous and seasonal vegetation of *T. domingensis*, *P. crassipes*, and *S. americanus*. P4 corresponds to a deep drainage channel where *P. crassipes* is conspicuous and the sediments

are thick. P5 and P6, at the western and central ends of the lagoon, respectively, are areas with seasonally variable depths and accumulation of organic matter, dominated by *P. stratiotes*, *B. monnieri*, and *H. ranunculoides*.

### *2.3. In Situ Analysis of Physicochemical Parameters*

Depth was measured using a graduated ruler with an accuracy of 1 mm. Physicochemical and limnological parameters were recorded in the water surface layer (0–20 cm), following the National Protocol for Water Resource Monitoring [54]. The temperature (°C) (Standard Method 2550) [55], pH (Standard Method 4500-H+) [55], electrical conductivity (µS/cm) [55] and total dissolved solids (TDS) (Standard Method 2510) [55], and dissolved oxygen (DO) (Standard Method 4500-O) [55] were measured using a YSI Pro multiparameter meter (YSI Inc., Yellow Springs, OH, USA). Turbidity was determined using a Lutron LT-TU2016 turbidimeter (Lutron Electronic Enterprise, Taipei, Taiwan) (Standard Method 2130 B) [55], while total suspended solids (TSS) were measured with a Hach DR900 multiparameter portable colorimeter following the manufacturer's photometric method (Hach, Loveland, CO, USA) (Method 8006) [55]. All measurements were taken between 10:00 a.m. and 2:00 p.m. to minimize diurnal variability in physicochemical and limnological parameters. Determinations were made in accordance with the Standard Methods for the Examination of Water and Wastewater [55]. For nutrient analysis, 500 mL of surface water was collected at each sampling point, selecting representative and homogeneous areas and avoiding substrate disturbance. The samples were stored at 4 °C for transport and subsequent laboratory processing. Phosphate concentration was determined using the ascorbic acid method with a Hanna HI3850 photocolorimeter (HANNA instruments, Smithfield, RI, USA) (Standard Method 4500-P E) [55], ammonium using the Nessler method with a Hanna HI733 photocolorimeter (HANNA instruments, Smithfield, RI, USA) (Standard Method 4500-NH3) [55], and nitrate using the cadmium reduction method with a Hach DR900 photocolorimeter (Hach, Loveland, CO, USA) (Standard Method 4500-NO3-E) [55]. Dissolved organic matter (DOM) was determined as volatile dissolved solids by gravimetry, using 0.45 µm glass fiber filters, vacuum filtration, and subsequent incineration at 550 °C in a muffle furnace for 2 h [56].

### *2.4. Aquatic Macrophytes Abundance Determination and Sample Collection*

#### 2.4.1. Determination of the Relative Abundance of Aquatic Macrophytes

To estimate the relative abundance of aquatic macrophytes, five 0.5 $m^2$ quadrats were placed along an approximately 20 m diagonal belt transect aligned with the depth gradient at each sampling point. The transect extended from the deepest area toward the shallower margins on both sides [57]. This setup ensured representation of the macrophyte life-form zones at each site, including pleustophyte-dominated deeper areas, amphibious transitional zones, and emergent shallow areas. The percentage cover of each species was visually and directly estimated within each quadrat by the same observer throughout the study, using a quantitative direct cover estimation method to reduce inter-observer variability [58].

#### 2.4.2. Species Identification of Aquatic Macrophytes

Aquatic macrophytes were collected at each sampling point and preserved in polyethylene bags with formalin. Identification was based on specialized literature [59] and taxonomic checklists [36,37]. The specimens were deposited in the Herbarium of the Natural History Museum of Ricardo Palma University.

### *2.5. Identification and Classification of Drivers of Change (DOC)*

Drivers of change (DOC) were assessed following a standardized field protocol proposed by Sarkar et al. [60] and adapted by Aponte et al. [35].

In the case of the Santa Rosa wetland, specific modifications were incorporated, and the recommendations from the recently developed methodological framework for coastal wetlands in Peru [47] were also considered in the interpretation of the data. DOCs were classified as direct (immediate physical and chemical impacts) and indirect (sociopolitical, economic, and cultural factors that modulate direct drivers). Direct DOCs included: uncontrolled urban growth (UUG); agricultural (ADG) and livestock (LDG) degradation; introduction of exotic species (IES); chemical (CHC), organic (ORC), and microbiological (MBC) contamination; domestic (DSW) and construction (CSW) solid waste; landfill and drainage (LFD); presence of farms (POF); effluents (EFL); and other disturbances (undermining or extraction) (DIS). The only indirect DOC considered was poor management and inadequate governance policies (GPI). Identification was carried out through direct observation at the six sampling points with photographic recording and GPS georeferencing, based on parallel research and a literature review. Discrepancies between observers were resolved through field discussion and consensus.

### *2.6. Data Analysis*

Principal coordinate analysis (PCoA) was performed using the vegan package in R [61] to evaluate the spatiotemporal variation of the DOCs. Matrices were constructed from the physicochemical parameters and relative abundances of macrophytes by season (T1 and T2) and sampling point (P1–P6), and were summarized in tables and graphs. The relationship between the environmental and biological matrices was evaluated using Spearman's correlation (ρ), considering the value of the statistic and its significance ($p < 0.05$). Physicochemical data were normalized, and the biotic matrix was transformed using the Hellinger procedure, excluding species with a single occurrence [62].

Principal component analysis (PCA) was used to summarize variability and determine spatiotemporal patterns. The significant axes of the PCA were analyzed using multiple response permutation analysis (MRPP) in Vegan package, considering seasons and sampling points as factors of variability [63,64]. The significant axes of an ordination analysis concentrate on the variability of a data matrix and identify the variables that explain it, that is, those most correlated with the axes, allowing statistical inference about this relationship [62]. The relationship between the matrices was analyzed by co-inertia analysis using the ade4 package [65,66]. Co-inertia assesses the covariation of the significant axes of PCA, quantifying shared inertia through the RV coefficient as a measure of the association between both matrices [62]. All analyses were performed using R and RStudio software [67,68].

## 3. Results

### *3.1. Spatiotemporal Variation*

#### 3.1.1. Macrophytes

Ten species of aquatic macrophytes were recorded: *H. ranunculoides*, *P. crassipes*, *P. stratiotes*, *Azolla filiculoides* Lam., *B. monnieri*, *Ludwigia peruviana* (L.) H. Hara, *S. americanus*, *L. gibba*, *Nasturtium officinale* W. T Aiton, and *Paspalum vaginatum* Sw. Following the life-form classification of Chambers and Maberly [17], these species represent three functional groups: pleustophytes (*P. crassipes*, *P. stratiotes*, *A. filiculoides*, *L. gibba*), amphibious and floating-leaved species (*H. ranunculoides*, *B. monnieri*, *N. officinale*), and emergent species (*L. peruviana*, *S. americanus*, *P. vaginatum*). Macrophyte coverage generally ranged from 35% to 100%, with significant variation between T1 and T2 at sampling points P1–P6. At P1, the coverage increased from 88% in T1 to 100% in T2. At P2, there was an increase from 85% in T1 to 100% in T2. In contrast, at P3, coverage decreased slightly from 90% in T1 to 85% in T2. At P4, high coverage was observed in both periods

(95% in T1 and 100% in T2). At P5, coverage showed notable variation, from 60% in T1 to 35% in T2. Finally, at P6, coverage reached 98% in T1 and decreased to 75% in T2 (Figure 2).

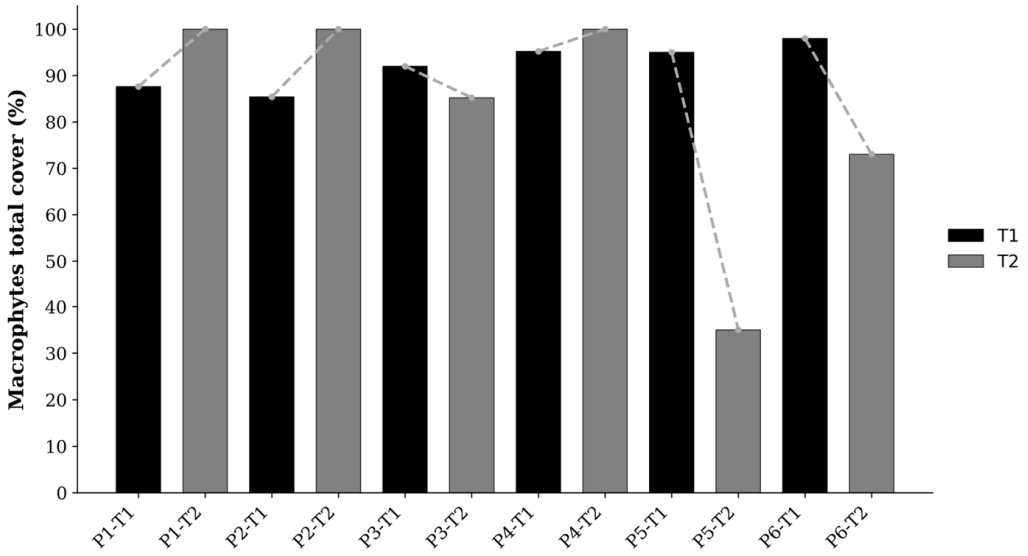



**Figure 2.** Macrophyte total cover (%) at sampling points (P1–P6) in the Santa Rosa wetland during two seasons: T1 (low-water season) and T2 (high-water season).

Regarding the composition and representativeness of aquatic macrophytes at each sampling point and season, shifts in dominance among life-form groups were observed (Figure 3). At P1, dominance changed from *P. crassipes* (65% in T1) to *H. ranunculoides* (55% in T2) (Figure 3A). At P2, *H. ranunculoides* was dominant in T1 (50%); however, in T2, it decreased to 6% and was replaced by *P. stratiotes* (80%) (Figure 3B). At P3, *B. monnieri* dominated in T1 (50%), whereas in T2, it was absent and dominance shifted to *H. ranunculoides* (40%) together with *P. stratiotes* (30%) (Figure 3C). At P4, the dominance of *P. crassipes* (60%) and *L. peruviana* (35%) in T1 decreased in T2, whereas *A. filiculoides* reached 50% (Figure 3D). At P5, the dominance among *B. monnieri* (40%), *H. ranunculoides* (30%), and *P. stratiotes* (25%) in T1 was reduced in T2 to the exclusive presence of *B. monnieri* (35%) (Figure 3E). Finally, at P6, the shared dominance between *P. stratiotes* (60%) and *H. ranunculoides* (35%) in T1 changed in T2 to the sole presence of *P. stratiotes* (75%) (Figure 3F).

The distribution of macrophyte life forms indicated seasonal changes in species composition (Figure 4). At P1, the assemblage exhibited a transition from pleustophyte dominance in T1 to a more balanced contribution of pleustophytes and amphibious/floating-leaved forms in T2 (Figure 4A). At P2, amphibious/floating-leaved forms dominated in T1, whereas pleustophytes dominated in T2 (Figure 4B). At P3, T1 was characterized by a mixed assemblage, with amphibious/floating-leaved and emergent forms predominating. In contrast, in T2, the macrophytes community transitioned towards a mixed pleustophyte-amphibious/floating-leaved structure, with only a minor contribution of emergent forms (Figure 4C). At P4, pleustophytes and emergent forms co-occurred in T1, but in T2 the assemblage became strongly dominated by pleustophytes, with a smaller contribution of amphibious/floating-leaved forms (Figure 4D). At P5, amphibious/floating-leaved forms dominated in both seasons and were the only life form recorded in T2, where emergent and pleustophyte forms were no longer registered, consistent with the marked reduction in total macrophyte cover at this site (Figure 4E). At P6, pleustophytes dominated in both seasons, increasing in relative abundance in T2, when amphibious/floating-leaved forms were no longer recorded (Figure 4F). In general, emer-

gent forms were limited to P3 and P4 and exhibited greater predominance in T1 than in T2.

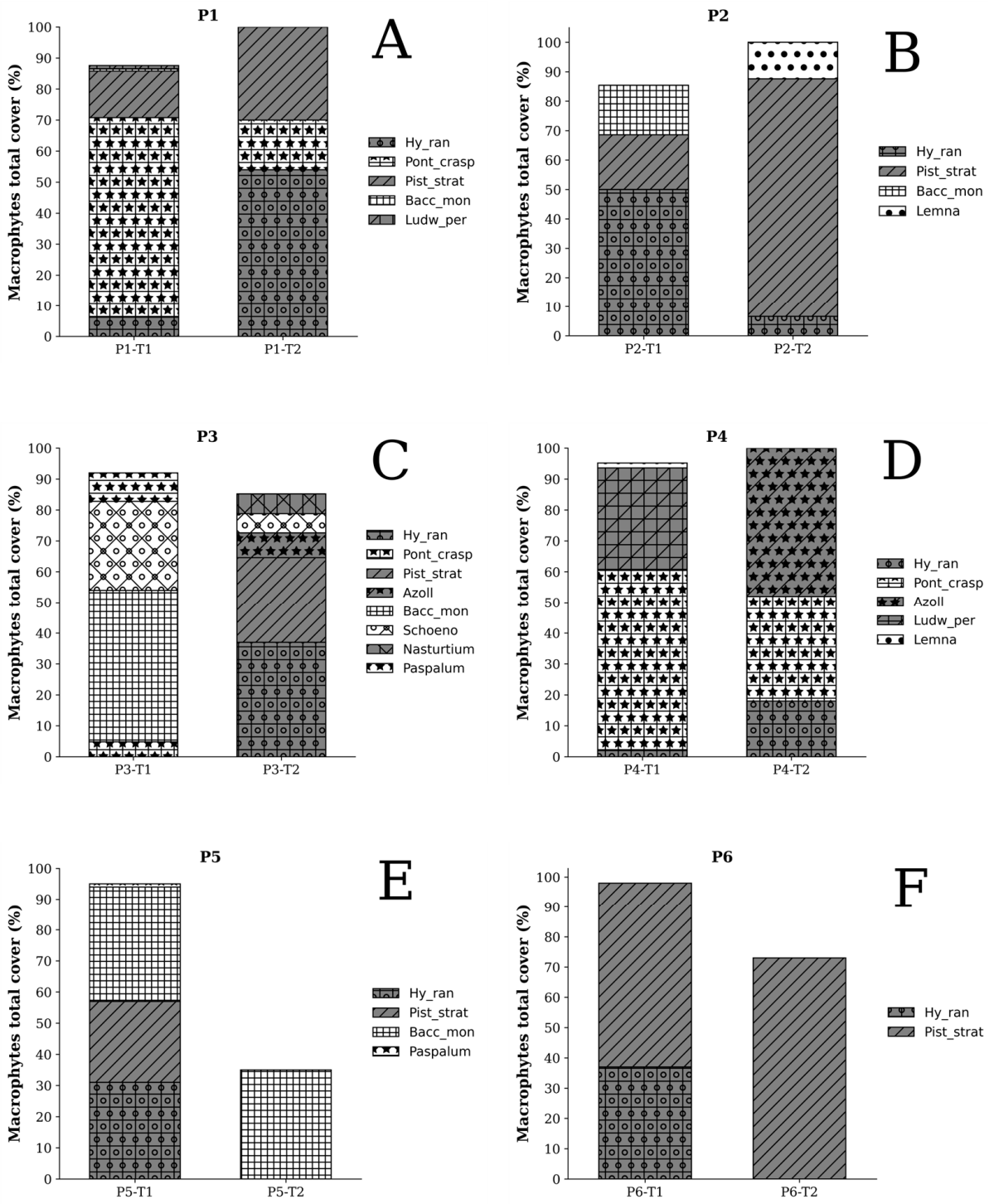



**Figure 3.** Percentage composition of aquatic macrophytes in Santa Rosa wetland during two seasons (T1: low-water season, T2: high-water season) at sampling points: (**A**) P1, (**B**) P2, (**C**) P3, (**D**) P4, (**E**) P5, and (**F**) P6. Hy_ran: *Hydrocotyle ranunculoides*; Pont_crasp: *Pontederia crassipes*; Pist_strat: *Pistia stratiotes*; Bacc_mon: *Bacopa monnieri*; Ludw_per: *Ludwigia peruviana*; Lemna: *Lemna gibba*; Schoeno: *Schoenoplectus americanus*; Nasturtium: *Nasturtium officinale*; Paspalum: *Paspalum vaginatum*; Azoll: *Azolla filiculoides*.

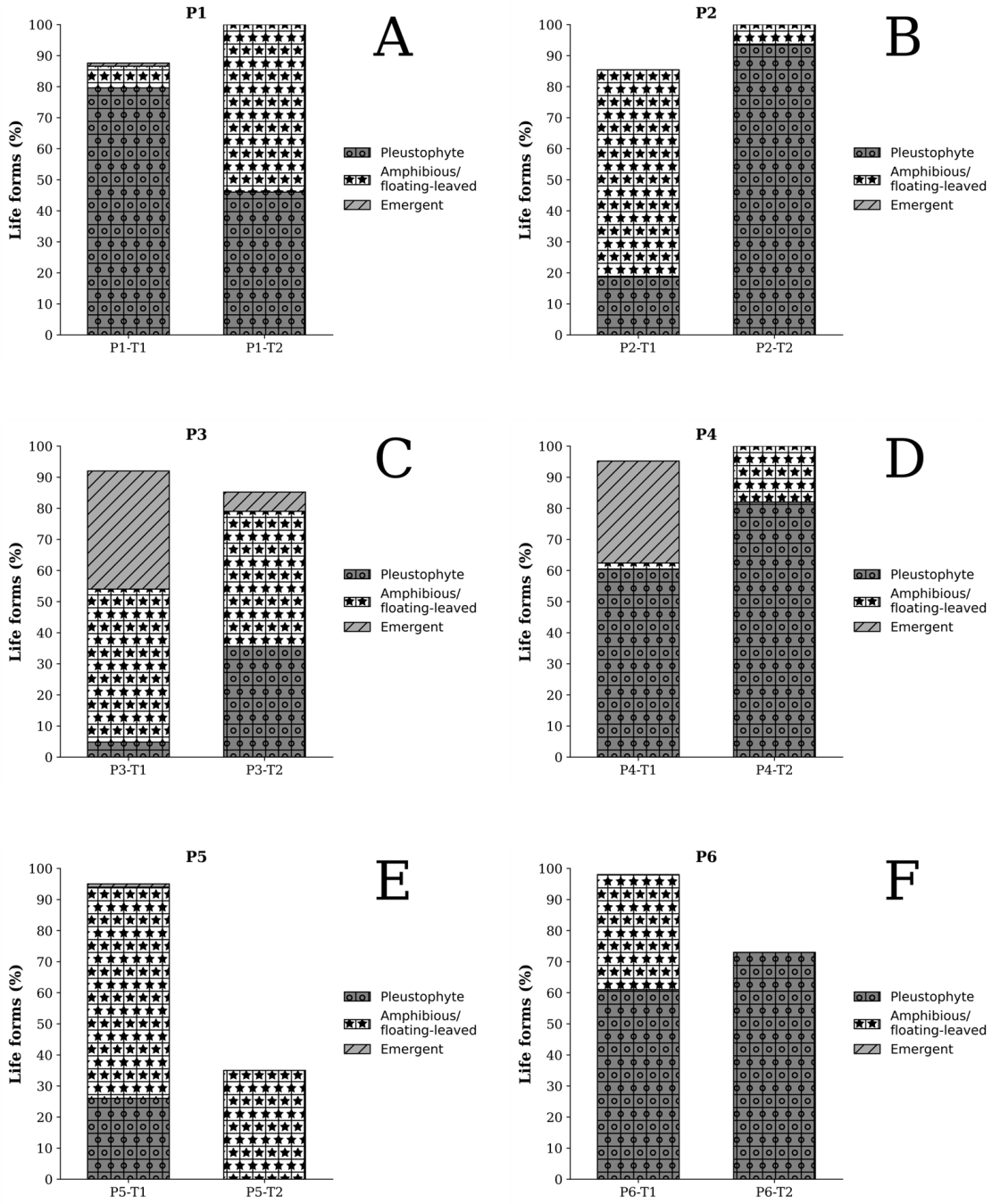



**Figure 4.** Percentage composition of aquatic macrophytes life forms in Santa Rosa wetland during two seasons (T1: low-water season, T2: high-water season) at sampling points: (**A**) P1, (**B**) P2, (**C**) P3, (**D**) P4, (**E**) P5, and (**F**) P6.

### 3.1.2. Physicochemical Parameters

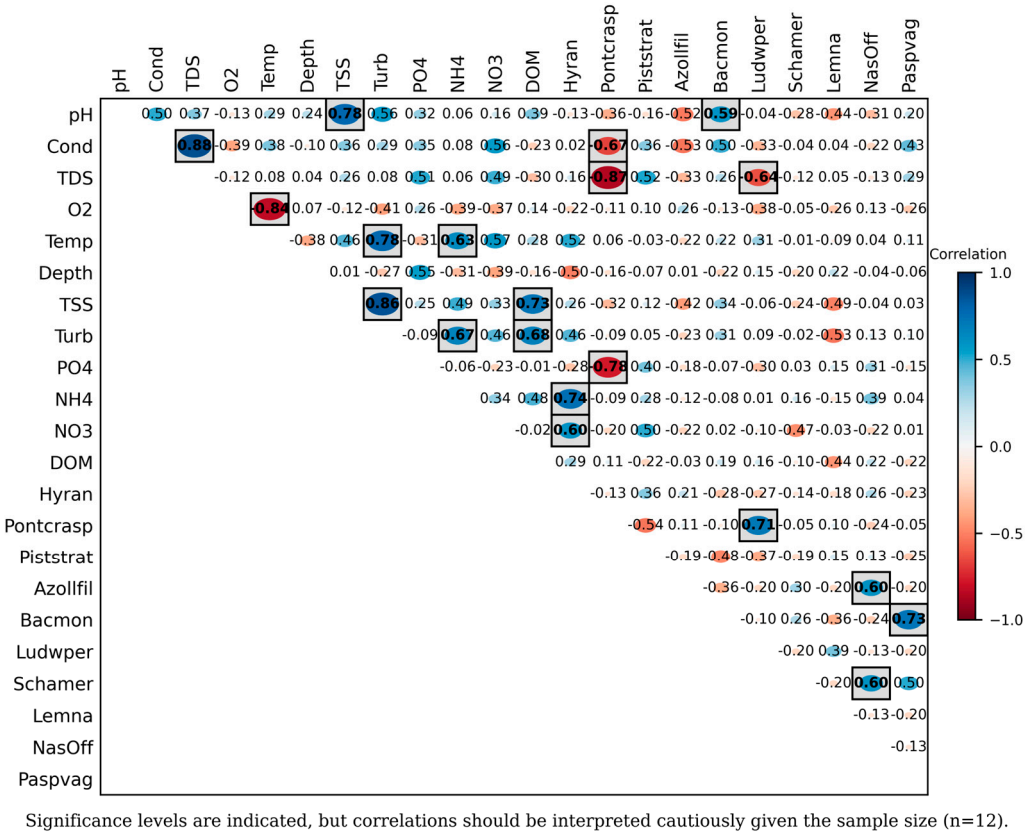
Spatiotemporal variation was observed in physicochemical parameters (Supplementary Materials). Depth increased in T2 compared with T1, except at P4, where it decreased. The temperature decreased in T2 (16.7–22.1 °C) compared with T1 (22–25.9 °C). DO increased significantly in T2 (3.2–8.2 mg/L), reaching the maximum value at P5 (8.5 mg/L) and the minimum at P2 (3.2 mg/L). The pH remained alkaline in both seasons, with a slight decrease in T2. Conductivity and TDS decreased in T2, reflecting a seasonal ionic dilution. Similarly, TSS and turbidity decreased, with the highest values recorded at P1, P6, and P7 during T1, and the lowest at P4 in T2. Regarding nutrients, nitrates showed marked spatial variation in both seasons, while ammonium and phosphates tended to increase in

T2, with maximum at P1 and P3 for ammonium, and at P3 for phosphates. DOM showed a heterogeneous pattern among sampling points, with increases at P3 and P5 and decreases at P1, P2, and P4 in T2 (Supplementary Materials).

### *3.2. Spearman's Correlation Matrix and Drivers of Change*

#### 3.2.1. Spearman's Correlation Matrix

A positive correlation was observed between electrical conductivity and TDS (ρ = 0.88), as well as between TSS and turbidity (ρ = 0.86), TSS and DOM (ρ = 0.73), and turbidity and DOM (ρ = 0.68). O2 was negatively correlated with temperature (ρ = −0.84). P. crassipes and L. peruviana were positively correlated (ρ = 0.72). Regarding the physicochemical–macrophyte relationship, P. crassipes was negatively correlated with conductivity (ρ = −0.67), TDS (ρ = −0.87), and phosphates (ρ = −0.78) (Figure 5).




Significance levels are indicated, but correlations should be interpreted cautiously given the sample size (n=12).

**Figure 5.** Spearman's correlation matrix of association between biotic and abiotic variables. The color scale and ellipse size represent the direction and strength of the correlation coefficient (ρ), respectively. Those values marked marked in bold indicate the level of significance of the correlation ($p < 0.05$). Cond: conductivity; TDS: total dissolved solids; O2: dissolved oxygen; Temp: temperature; TSS: total suspended solids; Turb: turbidity; DOM: dissolved organic matter; Hyran: Hydrocotyle ranunculoides; Pontcrasp: Pontederia crassipes; Piststrat: Pistia stratiotes; Bacmon: Bacopa monnieri; Ludwper: Ludwigia peruviana; Lemna: Lemna gibba; Schamer: Schoenoplectus americanus; NasOff: Nasturtium officinale; Paspvag: Paspalum vaginatum; Azollfil: Azolla filiculoides.

#### 3.2.2. Drivers of Change

The analysis of drivers of change (DOC) in the Santa Rosa Wetland revealed a spatially heterogeneous pattern of impact (Table 1). P4 had the highest number of DOCs (12), followed by P1 and P5 (9 each), P6 (7), and P2 and P3 (6 each). Livestock degradation (LDG), characterized by cattle, sheep, goats, and pig farming, was present at all sampling points. Similarly, the introduction of exotic species (IES), such as dogs and free-grazing livestock (cattle, sheep, and goats), occurred at all locations. Organic contamination (ORC),

characterized by food and agricultural waste, livestock manure, urban effluents, as well as microbiological contamination (MBC), indicated by the presence of thermotolerant coliforms, was also recorded at all sampling points. Governance failures (GPI), including poor inter-institutional participation and coordination, which led to deficiencies in the effective protection of wetlands, were considered common across all sampling points. Chemical contamination (CHC), characterized by the use of pesticides and fertilizers, was recorded at P1, P4, P5, and P6. Domestic solid waste (DSW) was found at P4, P5, P6, and P1 during T1, and construction solid waste (CSW) and landfill and drainage (LFD) were recorded at P1, P4, and P5. Other DOCs showed localized distribution: uncontrolled urban growth (UUG) was recorded only at P4; agricultural degradation associated with land use change for agriculture (ADG) at P3 and P4; pig and cattle farms (POF) in P2; effluents (EFL) at P1, and other disturbances (DIS), such as undermining or extraction, only in P4 in T2.

**Table 1.** Drivers of change identified at the sampling points. UUG: Uncontrolled urban growth; ADG: Agricultural degradation; LDG: Livestock degradation; IES: Introduction of exotic species; CHC: Chemical contamination; ORC: Organic contamination; MBC: Microbiological contamination; DSW: Domestic solid waste; CSW: Construction solid waste; LFD: Landfill and drainage; POF: Presence of farms: EFL: Effluents; GPI: Governance failures associated with poor management and inadequate policies; and DIS: Other disturbances (undermining or extraction). Note: 1 = presence of driver; 0 = absence.

| DOC | UUG | ADG | LDG | IES | CHC | ORC | DSW | CSW | LFD | POF | MBC | GPI | EFL | DIS |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| T2P1 | 0 | 0 | 1 | 1 | 1 | 1 | 0 | 1 | 1 | 0 | 1 | 1 | 0 | 0 |
| T2P2 | 0 | 0 | 1 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 1 | 1 | 0 | 0 |
| T2P3 | 0 | 1 | 1 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 |
| T2P4 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0 | 1 | 1 | 0 | 1 |
| T2P5 | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0 | 1 | 1 | 0 | 0 |
| T2P6 | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 0 | 0 | 0 | 1 | 1 | 0 | 0 |
| T1P1 | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0 | 1 | 1 | 1 | 0 |
| T1P2 | 0 | 0 | 1 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 1 | 1 | 0 | 0 |
| T1P3 | 0 | 1 | 1 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 |
| T1P4 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0 | 1 | 1 | 0 | 0 |
| T1P5 | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0 | 1 | 1 | 0 | 0 |
| T1P6 | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 0 | 0 | 0 | 1 | 1 | 0 | 0 |

#### 3.2.3. Principal Coordinates Analysis (PCoA)

The ordination based on the drivers of change (DOC) matrix revealed spatial heterogeneity in the pressures affecting the Santa Rosa Wetland (Figure 6). The first three axes accounted for 80.88% of the variability (PCoA1 = 53.41%, PCoA2 = 17.87%, and PCoA3 = 9.59%). Sampling points formed discrete, well-defined groups, in which the primary source of variation was spatial location rather than the temporal dimension, suggesting that pressure profiles persisted at each point throughout the analyzed period. Axis PCoA1 separated P2 and P3 from the remaining sampling points, while axis PCoA2 distinguished P3 from P4. Four subgroups emerged based on DOC incidence: one comprising P1, P5, and P6, which share a similar DOC profile, and three points (P2, P3, and P4) that clustered separately, each exhibiting a distinctive set of drivers of change. The EFL driver was characteristic of P1-T1, whereas CHC, DSW, and LFD were characteristic of P5 and P1-T2. Regarding POF, it was associated with P2 across both T1 and T2. Lastly, UUG and DIS were associated with P4 (T1 and T2). Collectively, these results indicate that driver variability in the wetland is predominantly spatial, resulting in a mosaic of zones with differentiated impact profiles.

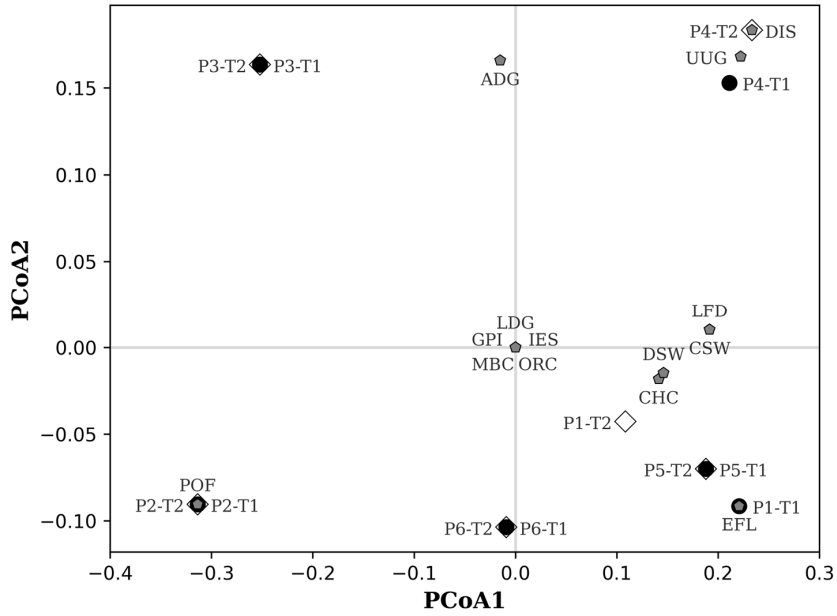



**Figure 6.** Principal Coordinates Analysis (PCoA) ordination of sampling points based on the drivers of change (DOC). Circles and diamonds represent sampling points across T1, low-water season and T2, high-water season respectively, pentagons represent drivers of change. UUG: Uncontrolled urban growth; ADG: Agricultural degradation; LDG: Livestock degradation; IES: Introduction of exotic species; CHC: Chemical contamination; ORC: Organic contamination; MBC: Microbiological contamination; DSW: Domestic solid waste; CSW: Construction solid waste; LFD: Landfill and drainage; POF: Presence of farms: EFL: Effluents; GPI: Governance failures associated with poor management and inadequate policies; and DIS: Other disturbances (undermining or extraction).

### *3.3. Relationship Between Biotic and Abiotic Matrices*

#### 3.3.1. Principal Component Analysis (PCA)

The first three principal components (PC1, PC2, and PC3) of the principal component analysis (PCA) of the abiotic matrix explained 74.55% of the total variability (PC1: 34.47%; PC2: 21.06%; PC3: 19.02%). PC1 retained the variability in turbidity, temperature, and total suspended solids ($\rho = -0.941$, $-0.864$, and $-0.850$, respectively), indicating a temporal gradient. PC2 concentrated the variability of conductivity and TDS ($\rho = 0.794$ and 0.792, respectively), showing a gradient of ionic concentrations between sampling points and seasons. PC3 retained the variability of pH and phosphates ($\rho = -0.750$ and $-0.872$, respectively), associated with a gradient of acidity and nutrient availability (Figure 7). The PCA of the biotic matrix explained 76.87% of the variability in the first three components (PC1: 31.20%; PC2: 27.15%; and PC3: 18.51%). PC1 retained the variability of *P. stratiotes*, *P. crassipes*, and *L. peruviana* ($\rho = -0.830$, $-0.788$, and 0.728, respectively), This reflects a primary spatial gradient separating the *Pistia-Hydrocotyle* assemblage—which characterized stations such as P5, P6, and P1-T2, associated with higher nutrient concentrations (as supported by the physicochemical PCA)—from the *P. crassipes-L. peruviana* assemblage characterizing station P4 and P1-T1, linked to deeper and more oxygenated conditions. In PC2, *B. monnieri* and *S. americanus* exhibited concentrated variability ($\rho = -0.891$ and $-0.625$, respectively), indicating a secondary spatiotemporal pattern with more mixed ecological aspects, involving amphibious (*B. monnieri*) and emergent (*S. americanus*) species that characterized specific points (e.g., P5-T2 and P3-T1) in shallower palustrine environments. PC3 was associated with the occurrence of *A. filiculoides* ($\rho = 0.875$), indicating favorable conditions for this species (Figure 8). Statistically significant differences were found in the MRPPs between seasons (PC1) and sampling points (PC3) of the abiotic matrix ($p < 0.04$ and $p < 0.037$, respectively) and between sampling points for PC2 of the biotic matrix ($p < 0.047$).

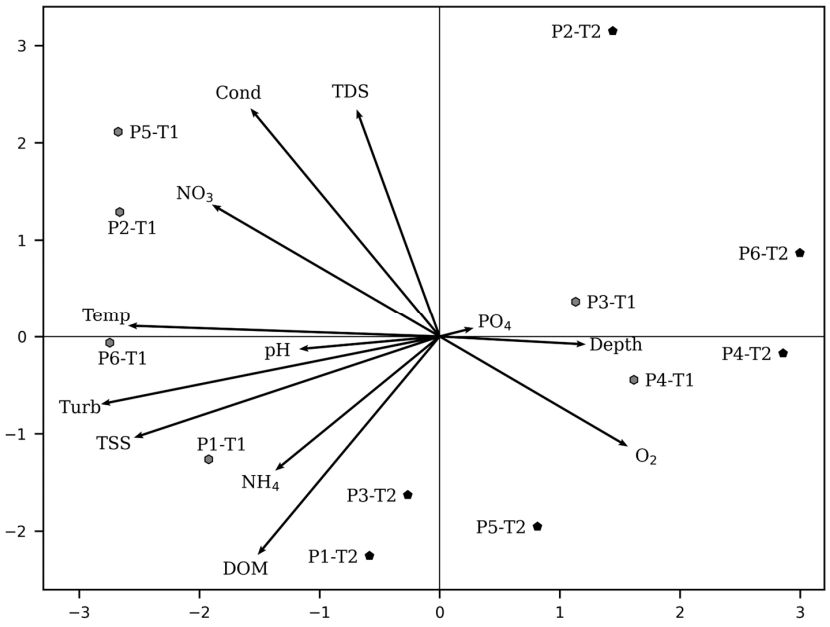



**Figure 7.** Principal Component Analysis (PCA) for physicochemical variables. Black arrows represent physicochemical variables, hexagons and pentagons represent sampling points across T1, low-water season and T2, high-water season respectively. Cond: conductivity; TDS: total dissolved solids; $O_2$: dissolved oxygen; Temp: temperature; TSS: total suspended solids; Turb: turbidity; DOM: dissolved organic matter; $NO_3$: nitrate; $PO_4$: phosphate; $NH_4$: ammonium.

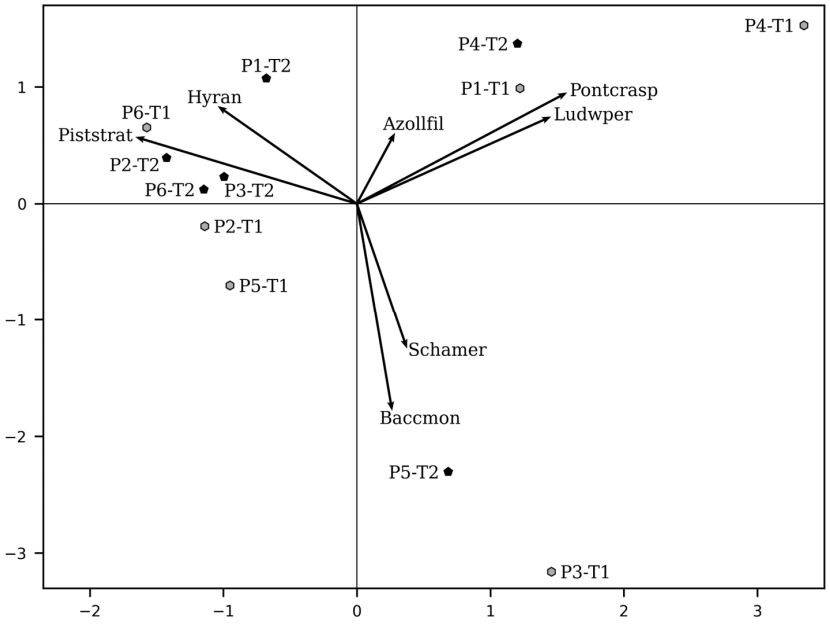



**Figure 8.** Principal Component Analysis (PCA) for macrophytes. Black arrows represent macrophyte species, hexagons and pentagons represent sampling points across T1, low-water season and T2, high-water season respectively. Hyran: *Hydrocotyle ranunculoides*; Pontcrasp: *Pontederia crassipes*; Piststrat: *Pistia stratiotes*; Azollfil: *Azolla filiculoides*; Baccmon: *Bacopa monnieri*; Ludwper: *Ludwigia peruviana*; Schamer: *Schoenoplectus americanus*.

### 3.3.2. Co-Inertia

The co-inertia analysis obtained a total inertia of 8.693, retaining 88.97% of the co-inertia (E1: 50.48%, E2: 25.84%, and E3: 12.65%). The correlation (covariation) between both matrices was high for axes 1 and 2, and moderate for axis 3 (ρE1 = 0.86, ρE2 = 0.82, ρE3 = 0.67). An RV coefficient of 0.468 was obtained, representing moderate covariation

between the matrices. A main gradient (E1) was observed, defined by TDS, Cond, TSS, Turb, $PO_4^{3-}$, and $NO_3^-$ (positive correlations) and Depth (negative correlation). E2 showed a gradient of pH and $PO_4^{3-}$ (positive correlation) and $NH_4^+$ and $NO_3^-$ (negative correlation). The axis structure reflects the pattern observed in the PCA, with the dilution of the main ions increasing with depth. *P. stratiotes* showed covariation with TDS, Cond, $NO_3^-$, $PO_4^{3-}$, and pH. *B. monnieri* was associated with pH and $PO_4^{3-}$, whereas *H. ranunculoides* was more related to shallow environments with high concentrations of $NH_4^+$, TSS, and Turb. *A. filiculoides* was associated with deeper and less mineralized environments, whereas *P. crassipes* was more related to less mineralized environments (Figure 9). Good agreement was observed between the biotic and abiotic matrices in P3-T2, P2-T2, P6-T1, and P5-T2 (short arrows) (Figure 10), while in P6-T2, P1-T1, P5-T1, P3-T1, and P4-T2, disagreement was observed between both matrices (long arrows). Spatiotemporally, P2-T1, P5-T1, and P6-T1 were associated with the Cond, TDS, pH, and $PO_4^{3-}$ gradient and the presence of *P. stratiotes* and *B. monnieri*; P4-T1 and P4-T2 were associated with low mineralization and the presence of *P. crassipes* and *A. filiculoides*, whereas P1-T2 and P3-T2 were associated with *H. ranunculoides* and high concentrations of $NH_4$ and turbidity, while P5-T2 aligned with the pH gradient and *B. monnieri*. In general, a pattern of ionic dilution was observed with increasing depth from T1 to T2, along with a temporal gradient in physicochemical limnology and aquatic macrophytes (Figures 9 and 10). Although moderate covariation was observed between the matrices, it was not statistically significant ($p > 0.05$) owing to variability within each sampling point.

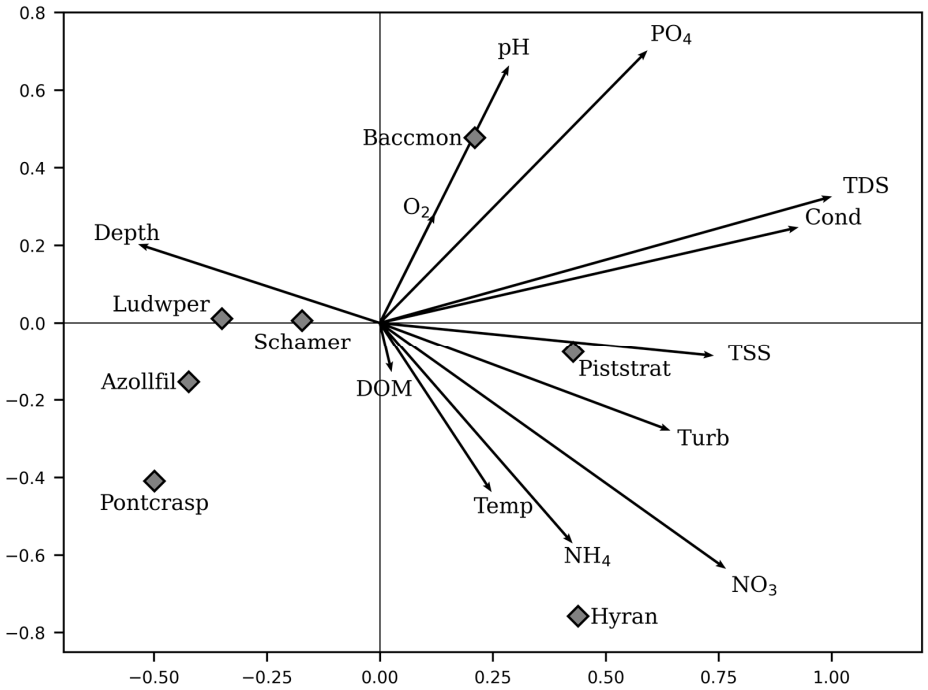



**Figure 9.** Co-inertia biplot (axes 1–2): covariance-optimized structural matching between limnological variables and macrophyte assemblages. Black arrows represent physicochemical variables, diamonds represent macrophyte species. Arrow length indicates the strength of the contribution to the co-inertia axes. Cond: conductivity; TDS: total dissolved solids; O2: dissolved oxygen; Temp: temperature; TSS: total suspended solids; Turb: turbidity; DOM: dissolved organic matter; $NO_3$: nitrate; $PO_4$: phosphate; $NH_4$: ammonium; Hyran: *Hydrocotyle ranunculoides*; Pontcrasp: *Pontederia crassipes*; Piststrat: *Pistia stratiotes*; Azollfil: *Azolla filiculoides*; Baccmon: *Bacopa monnieri*; Ludwper: *Ludwigia peruviana*; Schamer: *Schoenoplectus americanus*.

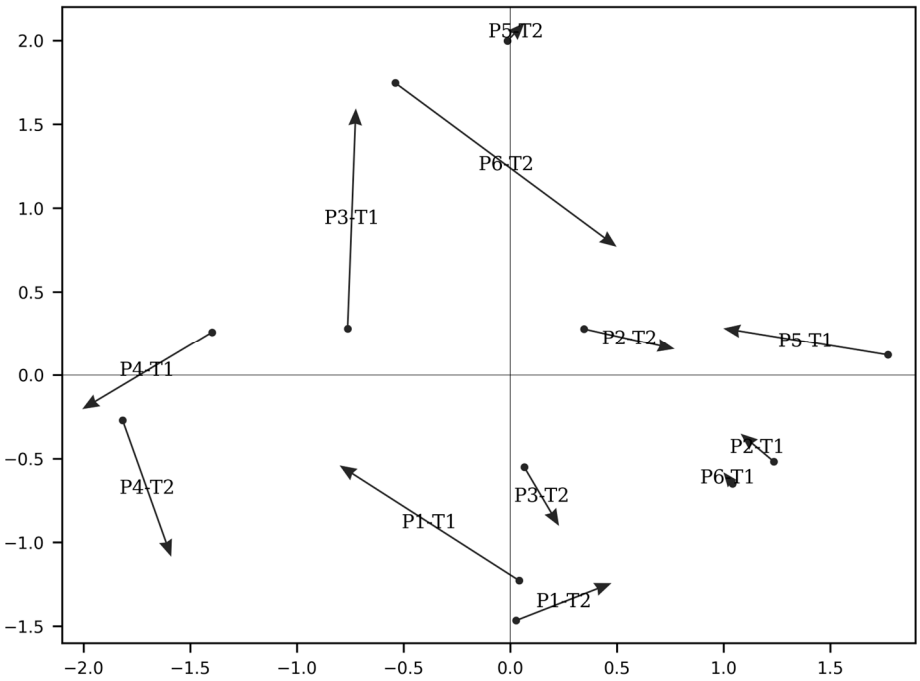



**Figure 10.** Temporal trajectories showing changes between sampling periods. Arrows connect T1 (low-water season) and T2 (high-water season) positions for each sampling point. Short arrows indicate strong agreement between the biotic and abiotic matrices; long arrows indicate disagreement.

## 4. Discussion

The ecological dynamics of wetlands are characterized by hierarchical interactions among the hydrological regime, physicochemical limnology, and the structure of the aquatic macrophyte community, with linkages that vary spatially and temporally [9,13,18]. This relationship is not unidirectional; aquatic macrophytes and their effects on physicochemical heterogeneity can feed back into the system, altering or buffering the hydrological regime [17,19,20]. The alternation of these processes defines a wetland as an ecological system [12], and it is within this dynamic that anthropogenic impacts become evident [22,23,69].

The Santa Rosa wetland is severely affected by various drivers of change that compromise its integrity, as are most coastal wetlands on the Lima, which are predominantly urban [35]. However, the fundamental ecological pattern of a wetland, the spatiotemporal variability of physicochemical limnology, and the macrophyte community, modulated by the different phases of hydrological seasonality, remain recognizable, with some reservations, as demonstrated in this study, which is the first of its kind in Peruvian coastal wetlands.

The results show that the Santa Rosa wetland is subject to a combination of direct and indirect drivers of change that act synergistically, generating cumulative pressures on its ecological integrity. The simultaneous presence of factors such as waste accumulation, livestock farming, and the introduction of exotic species has been widely documented as a major cause of tropical wetland degradation [60,70]. These direct drivers act on habitat structure and water quality, whereas indirect factors, such as poor governance and unplanned urban expansion, perpetuate these impacts by weakening institutional capacities to mitigate them [35,71]. Evidence suggests that when these drivers coexist spatially, ecological effects intensify, hindering natural recovery processes [72]. The binary presence/absence coding of drivers of change, adopted following the standardized protocols for tropical and Peruvian coastal wetlands [35,47,60], assigns equal statistical weight to drivers of potentially different magnitude. While the PCoA ordination demonstrates that

this approach effectively captures the spatial distribution and co-occurrence of drivers across the wetlandscape, it does not quantify their differential ecological impact. Future studies should incorporate semi-quantitative intensity scales [47] to enable more nuanced assessments of cumulative effects.

In the Santa Rosa wetland, physicochemical limnology showed marked seasonal variations, with higher concentrations of total suspended solids (TSS), turbidity, temperature, conductivity, and total dissolved solids (TDS) during the low-water season (T1), attributable to the influence of the irrigation canal in T1 and the ionic dilution effect associated with greater water depth in T2. This pattern is characteristic of shallow wetlands with seasonal connectivity [13,73]. However, the temporal component of the hydrological regime exerts a more decisive control over the system's limnological dynamics [9,21], which, in the present study, is reflected in increased depth and seasonal ionic dilution. While the atmospheric regime of the central Peruvian coast—persistent cloud cover and reduced evapotranspiration during austral winter versus high solar radiation and increased evaporation during austral summer [34]—modulates the seasonal water balance, it operates as a background climatic condition rather than as a discrete driver of wetland seasonality. In the Santa Rosa wetland, the primary control over limnological variability is the hydrological cycle of aquifer recharge and exfiltration driven by upstream Andean precipitation [42]. Future studies incorporating direct atmospheric measurements would help quantify the relative contribution of evapotranspiration to seasonal water level fluctuations. It is important to note that the recorded nutrient concentrations and physicochemical parameters, which may seem high compared to pristine wetlands, constitute the current reference condition for a system embedded in an agricultural-urban matrix. Rather than anomalies, these values reflect the current hydrological regime and changes in the physicochemical variables that govern the wetland's seasonal variability. The ionic content of this system is modulated by both local processes (natural and anthropogenic) and the specific hydrological characteristics of the Santa Rosa wetland basin. In general, the ionic concentrations (conductivity and TDS) of the Santa Rosa wetland are lower than those of other coastal wetlands, such as the Pantanos de Villa, Humedales de Ventanilla, and wetlands in La Libertad region [38,41,43,44]. The conductivity values recorded in the Santa Rosa wetland (992–3280 µS/cm in T1 and T2) were below the limiting thresholds for dominant floating species, such as *P. stratiotes* and *P. crassipes* [74,75]. Regarding dissolved oxygen (DO), the recording of critical hypoxia thresholds in T1 (<2 mg/L) and recovery to physiologically acceptable levels for macrophytes and biota in general in T2 shows the alternation of states and their amplitudes between seasons [74]. This pattern is particularly evident at certain sampling points, characteristic of structurally complex wetlands with lacustrine and palustrine sectors [18,20]. Persistent hypoxia at specific sampling points results from the synergistic accumulation of oxygen-consuming processes linked to the hydrological transition: residual organic load from agricultural effluents, the turnover from amphibious to pleustophyte growth forms with consequent decomposition of amphibious biomass [76], and the increased surface coverage of interlocking floating macrophytes that restricts atmospheric gas exchange and limits light penetration [77,78]. These coupled oxygen-depleting mechanisms are characteristic of palustrine wetlands subject to fluctuating hydrological regimes [18,20].

Between T1 (low-water season) and T2 (high-water season), seasonal replacement of *P. crassipes* by *H. ranunculoides*, of *P. stratiotes* by *B. monnieri*, and of *B. monnieri* by *P. stratiotes* and *H. ranunculoides* was observed, reflecting a relative increase in marsh species in response to water retreat. However, this pattern varies among sectors of the wetland, where geomorphological heterogeneity and water connectivity modulate the magnitude of the coverage change between sampling point. This pattern coincides with observations in

other neotropical and urban aquatic systems, where macrophyte communities maintain a stable functional composition in the face of broad environmental gradients, and the homogenization of physical and chemical conditions limits floristic turnover and favors the dominance of a few species [79]. These results are consistent with studies showing that the seasonal dynamics of aquatic macrophytes in shallow wetlands respond strongly to hydrological regimes, especially the flood and retreat cycle, which favors amphibious and emergent species during periods of decreasing water depth [21,80]. The alternation in dominance between pleustophytes and amphibious/emergent species reflects adaptive mechanisms in response to water variability and habitat availability, which have been documented in neotropical and intertropical systems with marked seasonality [24,73].

The relationship between physicochemical limnology and the distribution of aquatic macrophytes in the Santa Rosa wetland was particularly evident in specific areas. The ionic concentration gradient (pH, conductivity, TDS) was associated with the dominance of *P. stratiotes* and *B. monnieri*, whereas *H. ranunculoides* was limited by increasing depth. The association of *H. ranunculoides* with shallow, turbid, ammonium-rich environments reflects its status as a floating marsh plant with floating or creeping stems in saturated soils [81]. In contrast, *P. crassipes* showed greater affinity for waters with lower pH, TDS, and conductivity. Likewise, a moderate covariation structure was evident between the biotic and abiotic matrices, reflecting specific patterns of change in particular sectors and macrophyte species across seasons and sampling points. Although these patterns were not consistent across the entire matrix, the relationship was not statistically significant. These findings are consistent with previous studies that have documented the sensitivity of aquatic macrophytes to environmental gradients, especially in shallow systems, where fluctuations in pH, nutrients, and conductivity strongly influence species distribution [69]. The specific association of *P. crassipes* with less mineralized waters and *P. stratiotes* with more mineralized environments reflects contrasting ecophysiological strategies in response to environmental conditions [20,22]. In the lagoon sector of the Santa Rosa wetland, Muñoz et al. [82] described the dynamics of *P. stratiotes*, the dominant floating species in that environment, whose growth is influenced by local concentrations of ammonium, phosphate, and phosphorus, highlighting the relevance of physicochemical limnology in the structuring of aquatic macrophytes in wetlands. Likewise, the influence of phosphorus on species assembly coincides with reports indicating that increased nutrient concentrations can induce abrupt changes in the dominance of pleustophyte or amphibious/emergent growth forms [17,25,83,84]. In the Santa Rosa wetland, agricultural activity has been suggested to promote an increase in the coverage of *P. crassipes*, *P. stratiotes*, and *Lemna gibba* [51,85], with high values of chemical oxygen demand (COD), biological oxygen demand (BOD), total nitrogen (TN), and total phosphorus (TP) being recorded [45,86]. The concentrations of nitrate, ammonium, and phosphate in the Santa Rosa Wetland exhibited a spatial and temporal nutrient gradient, associated with variations in macrophyte composition and cover. Although there are no previous records of the degree of trophic status and its progressive increase, the levels recorded in 2022 suggest a system tending toward eutrophication and hypertrophy, with spatial and temporal variability. This condition is not an anomaly, but rather a hydrochemical pattern characteristic of coastal wetlands immersed in agricultural-urban matrices [39,87,88]. The structure of the plant community responds to nutrient availability by assimilating excess nutrients and maintaining ecosystem activity despite anthropogenic pressure [26,89]. Although the results enabled the identification of associations between nutrient gradients and macrophyte distribution, it was not possible to establish causal relationships with respect to the system's eutrophication.

The absence of a statistically significant global RV coefficient ($p > 0.05$) does not indicate a lack of ecological coupling between the biotic and abiotic matrices, but rather reflects

the inherent spatial heterogeneity of the Santa Rosa wetland. The site-specific drivers of change and the structure of the co-inertia trajectories indicate that Santa Rosa operates as a wetlandscape [90,91] composed of lacustrine and palustrine functional units [92], each sustaining distinct macrophyte growth forms that alternate in space and time [17] at differentiated rates of change. In this context, biotic–abiotic coupling is not distributed homogeneously across the wetland, but rather concentrated in specific patches with particular functional identity and distinct environmental filters, including water volume and origin, organic and nutrient loading, agricultural activity and drainage dynamics, and mineralization gradients. Additional modulators include preferential herbivory on *P. crassipes* and *H. ranunculoides* [53,93] and sediment inputs from agricultural effluents, all of which contribute to within-site variability that a single global statistic cannot resolve. It should also be noted that the RV coefficient is inherently sensitive to sample size and the number of variables considered [94]. The observed pattern of spatially structured, locally strong associations thus constitutes a key finding in itself, underscoring the need for patch-based monitoring and management approaches in urban wetlands.

It is important to note that 2022 corresponds to the second year of an unusual three-year La Niña event (2020–2023). During La Niña phases, precipitation in the Chancay basin may be reduced, potentially limiting aquifer recharge and altering the wetland's hydrological regime [95]. The spatiotemporal turnover documented in this study therefore reflects a single hydrological cycle under these specific ENSO conditions, and the described patterns should be interpreted in this interannual context. Furthermore, climate change projections for the region suggest potential alterations in precipitation patterns and aquifer recharge dynamics that could directly affect the hydrological driver underlying the observed turnover.

Effective wetland conservation requires an understanding of their ecological functioning and the factors that shape their structure and dynamics, including not only the biotic communities they support but also the subsystems that compose them. Nevertheless, reductionist approaches persist in limnological studies, which tend to focus only on major open-water bodies such as lagoons or shallow lakes, overlooking functionally key environments, such as marshlands, floodplains, seasonal plains, and transitional microhabitats [9,10,69]. Previous limnological studies of Peruvian coastal wetlands have focused on the main lagoons and a few shallow environments, overlooking marsh environments [42,43]. Indeed, coastal wetlands such as Santa Rosa constitute "wetlandscapes," that is, mosaic landscapes composed of multiple functional units with specific physicochemical conditions and differential ecological responses [60,91]. The Santa Rosa wetland exhibits marked internal limnological heterogeneity, with marsh and lake sectors showing contrasting physicochemical dynamics. These sectors can be interpreted as differentiated functional patches, the coexistence of which contributes to the structural and functional stability of the wetland, without implying a formal classification of landscape units.

The results show that different sectors of the Santa Rosa Wetland respond differentially to water quantity and quality, which in turn modulates the structure of the macrophyte community. However, this pattern is only moderately explained by the intrinsic heterogeneity of the wetland and by the effects of drivers of change. This finding contributes to understanding the limnology–biota relationship, an aspect scarcely addressed in the literature on Peruvian wetlands, which has mostly favored descriptive or ecological community approaches without examining functional relationships with limnological dynamics [96,97]. This gap limits the ability to diagnose threats, such as drying, hydrological fragmentation, and climate change, which requires an understanding the coupling between abiotic and biotic matrices [27].

This study contributes to the existing body of knowledge by demonstrating that the limnological structure of wetlands cannot be reduced to the main lagoon. Rather, it must be understood as an integrated system, where areas currently undergoing degradation continue to perform key ecological functions. Despite the multiple drivers of change, waste accumulation, livestock farming, introduction of exotic species, and urbanization, the Santa Rosa wetland continues to sustain essential processes such as nutrient retention, habitat provision, and hydrological regulation, demonstrating the persistence of functional integration among temporal dynamics, physicochemical limnology, and aquatic macrophyte communities [60,70]. The findings suggest that the maintenance of critical functions does not depend exclusively on the best preserved areas but also on the interaction between impacted and less impacted sectors, which has direct implications for the design of conservation, restoration, and climate change adaptation strategies [3,12].

Given the increasing degradation of coastal wetlands along the Peruvian Pacific, understanding the relationship among the hydrological cycle, limnology, and aquatic macrophytes, as addressed in this study, is key to redefining conservation priorities, identifying functional thresholds, and guiding interventions toward maintaining fundamental ecological processes, even in highly transformed landscapes. It is also suggested that the most variable sites, or those with a less clear explanatory relationship between biotic and abiotic processes, may be subject to unique local drivers of change. This opens up a particularly important line of research aimed at establishing causal links between the drivers of change and their effect on community-environment coupling. It is recommended that future studies address this question with extended temporal coverage.

## 5. Conclusions

The Santa Rosa wetland maintains recognizable ecological patterns despite sustained anthropogenic pressures. The seasonal hydrological cycle–contrasting high-water (T2) and low-water (T1) phases driven by aquifer recharge and exfiltration dynamics–remains the primary structuring force of physicochemical limnology. The aquatic macrophyte community responds coherently to seasonal changes in water depth, ionic concentration, and nutrient availability, with a functional turnover between pleustophytes in lacustrine sectors during high water and amphibious/emergent species in palustrine sectors during low water, evidencing functional coupling between biota and environmental conditions. Drivers of change are predominantly spatial in their distribution, generating a mosaic of differentiated impact profiles that modulate this coupling at local scales. By integrating limnological dynamics, macrophyte community structure, and drivers of change—an approach scarcely applied in Peruvian coastal wetlands—these findings provide a scientific foundation for identifying functional thresholds, guiding conservation strategies in increasingly threatened urban wetland systems along the Peruvian Pacific coast.

**Supplementary Materials:** The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/d18040214/s1.

**Author Contributions:** Conceptualization, J.A.A.-I. and F.V.R.-C.; methodology, J.A.A.-I. and F.V.R.-C.; software, J.A.A.-I.; validation, J.A.A.-I. and S.I.U.-R.; formal analysis, J.A.A.-I.; investigation, F.V.R.-C., S.I.U.-R. and J.A.A.-I.; resources, J.A.A.-I.; data curation, J.A.A.-I. and F.V.R.-C.; writing—original draft preparation, F.V.R.-C. and J.A.A.-I.; writing—review and editing, F.V.R.-C., J.A.A.-I. and S.I.U.-R.; visualization, J.A.A.-I. and F.V.R.-C.; supervision, J.A.A.-I.; project administration, J.A.A.-I.; funding acquisition, J.A.A.-I. All authors have read and agreed to the published version of the manuscript.

**Funding:** This study was conducted as part of the project: Environmental degradation vs. ecosystem response: A case study in the Santa Rosa wetland from a limnological perspective (004-2020-PRO99),

winner of the 2020 Cabieses Teaching Seed Fund Grant from the Universidad Científica del Sur, and is part of Flavia Valeria Rivera Cáceda's Licentiate Thesis in Marine Biology under the guidance of Dr. José Antonio Arenas Ibarra.

**Institutional Review Board Statement:** Not applicable.

**Data Availability Statement:** The original contributions presented in this study are included in the article and Supplementary Materials. Further inquiries can be directed to the corresponding authors.

**Acknowledgments:** Materials and equipment acquired by the project and others loaned for use by Terra Aqua Peru S.A.C. were used for field sampling and laboratory analysis. Chemical determinations were performed at Wet Experimental Laboratory No. 104 of the Universidad Científica del Sur. José Antonio Arenas Ibarra used protected hours for research at the Universidad Científica del Sur, the Institute of Natural Resources and Ecology, and the Ecology Laboratory at the Universidad Ricardo Palma. The facilities of the Ecology Laboratory of the Faculty of Biological Sciences and their infrastructure and research resources were used for the final draft. Access to academic databases at the Universidad Científica del Sur and Universidad Ricardo Palma were used. The authors gratefully acknowledge the National Forest and Wildlife Service (SERFOR) for facilitating the necessary research permits, and Williams Jurado Zevallos of the Santa Rosa Chancay Wetland Surveillance Committee, for providing the community consent to conduct research in the wetland. Special thanks are due to James López Visag, Martín Muñoz, Kaory Tinoco, Bruno Reátegui, and Edwin Pinto for their assistance during fieldwork and sample processing in the laboratory, as well as to Dámaso Ramírez for his guidance on field sampling techniques and his assistance in the taxonomic identification of aquatic macrophytes. We also would like to thank the examination committee of the licentiate thesis of Flavia Valeria Rivera Cáceda, for their suggestions on the thesis from which this article originates. Thanks are likewise extended to the anonymous reviewers of this article, whose observations and comments helped to improve this manuscript.

**Conflicts of Interest:** The authors declare no conflicts of interest.

## Abbreviations

The following abbreviations are used in this manuscript:

| | |
|---|---|
| Temp | Temperature |
| TDS | Total dissolved solids |
| TSS | Total suspended solids |
| DO | Dissolved oxygen |
| O2 | Dissolved oxygen |
| Cond | Conductivity |
| Turb | Turbidity |
| DOM | Dissolved organic matter |
| $NO_3$ | Nitrate |
| $PO_4$ | Phosphate |
| $NH_4$ | Ammonium |
| DOC | Drivers of change |
| PCA | Principal Component Analysis |
| PCoA | Principal Coordinate Analysis |
| MRPP | Multiple Response Permutation Procedure |
| UUG | Uncontrolled urban growth |
| ADG | Agricultural degradation |
| LDG | Livestock degradation |
| IES | Introduction of exotic species |
| CHC | Chemical contamination |
| ORC | Organic contamination |
| DSW | Domestic solid waste |
| CSW | Construction solid waste |

| | |
|---|---|
| LFD | Landfill and drainage |
| POF | Presence of farms |
| MBC | Microbiological contamination |
| GPI | Governance failures associated with poor management and inadequate policies |
| EFL | Effluents |
| DIS | Other disturbances (undermining or extraction) |
| Hy_ran | *Hydrocotyle ranunculoides* |
| Pont_crasp | *Pontederia crassipes* |
| Pist_strat | *Pistia stratiotes* |
| Bacc_mon | *Bacopa monnieri* |
| Ludw_per | *Ludwigia peruviana* |
| Lemna | *Lemna gibba* |
| Schoeno | *Schoenoplectus americanus* |
| Nasturtium | *Nasturtium officinale* |
| Paspalum | *Paspalum vaginatum* |
| Azoll | *Azolla filiculoides* |
| Hyran | *Hydrocotyle ranunculoides* |
| Pontcrasp | *Pontederia crassipes* |
| Piststrat | *Pistia stratiotes* |
| Azollfil | *Azolla filiculoides* |
| Baccmon | *Bacopa monnieri* |
| Ludwper | *Ludwigia peruviana* |
| Schamer | *Schoenoplectus americanus* |
| NasOff | *Nasturtium officinale* |
| Paspvag | *Paspalum vaginatum* |

## References


1. Grizzetti, B.; Lanzanova, D.; Liquete, C.; Reynaud, A.; Cardoso, A.C. Assessing Water Ecosystem Services for Water Resource Management. *Environ. Sci. Policy* **2016**, *61*, 194–203. [CrossRef]
2. Moomaw, W.R.; Chmura, G.L.; Davies, G.T.; Finlayson, C.M.; Middleton, B.A.; Natali, S.M.; Perry, J.E.; Roulet, N.; Sutton-Grier, A.E. Wetlands in a Changing Climate: Science, Policy and Management. *Wetlands* **2018**, *38*, 183–205. [CrossRef]
3. Mitsch, W.J.; Gosselink, J.G. Wetlands: Human Use and Science. In *Wetlands*, 5th ed.; Wiley: Hoboken, NJ, USA, 2015; pp. 3–26.
4. Davidson, N.C.; Finlayson, C.M. Updating Global Coastal Wetland Areas Presented in Davidson and Finlayson (2018). *Mar. Freshw. Res.* **2019**, *70*, 1195. [CrossRef]
5. Davidson, N.C. How Much Wetland Has the World Lost? Long-Term and Recent Trends in Global Wetland Area. *Mar. Freshw. Res.* **2014**, *65*, 934–941. [CrossRef]
6. Fluet-Chouinard, E.; Stocker, B.D.; Zhang, Z.; Malhotra, A.; Melton, J.R.; Poulter, B.; Kaplan, J.O.; Goldewijk, K.K.; Siebert, S.; Minayeva, T.; et al. Extensive Global Wetland Loss over the Past Three Centuries. *Nature* **2023**, *614*, 281–286. [CrossRef]
7. IPBES. *Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*; Zenodo: Bonn, Germany, 2019.
8. Xu, T.; Weng, B.; Yan, D.; Wang, K.; Li, X.; Bi, W.; Li, M.; Cheng, X.; Liu, Y. Wetlands of International Importance: Status, Threats, and Future Protection. *Int. J. Environ. Res. Public Health* **2019**, *16*, 1818. [CrossRef]
9. An, S.; Yin, S.; Verhoeven, J.T.A.; Jeelani, N. Chapter 29—Wetlands. In *Wetzel's Limnology*, 4th ed.; Jones, I.D., Smol, J.P., Eds.; Academic Press: San Diego, CA, USA, 2024; pp. 999–1013.
10. Mitsch, W.J.; Gosselink, J.G. Wetland Biogeochemistry. In *Wetlands*, 5th ed.; Wiley: Hoboken, NJ, USA, 2015; pp. 179–214.
11. Rasmussen, T.C.; Deemy, J.B.; Long, S.L. Wetland Hydrology. In *The Wetland Book: I: Structure and Function, Management, and Methods*; Finlayson, C.M., Everard, M., Irvine, K., McInnes, R.J., Middleton, B.A., van Dam, A.A., Davidson, N.C., Eds.; Springer: Dordrecht, The Netherlands, 2018; pp. 201–216.
12. Tiner, R.W. *Wetland Indicators: A Guide to Wetland Identification, Delineation, Classification, and Mapping*, 2nd ed.; Taylor & Francis: Boca Raton, FL, USA, 2017.
13. Mitsch, W.J.; Gosselink, J.G. Wetland Hydrology. In *Wetlands*, 5th ed.; Wiley: Hoboken, NJ, USA, 2015; pp. 111–160.
14. Reddy, K.R.; DeLaune, R.D. *Biogeochemistry of Wetlands: Science and Applications*; CRC Press: Boca Raton, FL, USA, 2008.

15. Davidson, N.C.; Middleton, B.A.; McInnes, R.J.; Everard, M.; Irvine, K.; Van Dam, A.A.; Finlayson, C.M. Introduction to the Wetland Book 1: Wetland Structure and Function, Management, and Methods. In *The Wetland Book: I: Structure and Function, Management, and Methods*; Springer: Dordrecht, The Netherlands, 2018; pp. 3–14.
16. Ramsar Convention Secretariat. Ramsar Convention on Wetlands. In *Global Wetland Outlook: State of the World's Wetlands and Their Services to People*; Ramsar Convention Secretariat: Gland, Switzerland, 2018.
17. Chambers, P.A.; Maberly, S.C. Chapter 24—Freshwater Plants. In *Wetzel's Limnology*, 4th ed.; Jones, I.D., Smol, J.P., Eds.; Academic Press: San Diego, CA, USA, 2024; pp. 759–816.
18. Bornette, G.; Puijalon, S. Response of Aquatic Plants to Abiotic Factors: A Review. *Aquat. Sci.* **2011**, *73*, 1–14. [CrossRef]
19. Mitsch, W.J.; Gosselink, J.G. Wetland Vegetation and Succession. In *Wetlands*, 5th ed.; Wiley: Hoboken, NJ, USA, 2015; pp. 215–255.
20. Siddiqui, A.J.; Jahan, S.; Adnan, M.; Ashraf, S.A.; Singh, R. Macrophytes and Their Role in Wetland Ecosystems. In *Aquatic Macrophytes: Ecology, Functions and Services*; Kumar, S., Bauddh, K., Singh, R., Kumar, N., Kumar, R., Eds.; Springer Nature: Singapore, 2023; pp. 119–138.
21. Maltchik, L.; Rolon, A.S.; Schott, P. Effects of Hydrological Variation on the Aquatic Plant Community in a Floodplain Palustrine Wetland of Southern Brazil. *Limnology* **2007**, *8*, 23–28. [CrossRef]
22. Freedman, B.; Lacoul, P. Environmental Influences on Aquatic Plants in Freshwater Ecosystems. *Environ. Rev.* **2006**, *14*, 89–136. [CrossRef]
23. Dar, N.A.; Pandit, A.K.; Ganai, B.A. Factors Affecting the Distribution Patterns of Aquatic Macrophytes. *Limnol. Rev.* **2014**, *14*, 75–81. [CrossRef]
24. Rameshkumar, S.; Radhakrishnan, K.; Aanand, S.; Rajaram, R. Influence of Physicochemical Water Quality on Aquatic Macrophyte Diversity in Seasonal Wetlands. *Appl. Water Sci.* **2019**, *9*, 12. [CrossRef]
25. Dubey, D.; Kumar, S.; Dutta, V. Impact of Nutrient Enrichment on Habitat Heterogeneity and Species Richness of Aquatic Macrophytes: Evidence from Freshwater Tropical Lakes of Central Ganga Plain, India. *Int. J. Environ. Sci. Technol.* **2022**, *19*, 5529–5546. [CrossRef]
26. Florescu, L.I.; Catana, R.D.; Mihai, R.A.; Dumitrache, A.C.; Moldoveanu, M.M. Macrophyte Community Distribution in Relation to Anthropization Influences and Phytoplankton Development in an Urban Lake Chain. *Water* **2024**, *16*, 3467. [CrossRef]
27. Madzivanzira, T.C.; Coetzee, J.A.; Dalu, T. Factors Structuring Aquatic Macrophytes. In *Aquatic Macrophytes: Ecology, Functions and Services*; Kumar, S., Bauddh, K., Singh, R., Kumar, N., Kumar, R., Eds.; Springer Nature: Singapore, 2023; pp. 21–52.
28. Cerna-Arrue, A.; Aponte, H. El origen de los humedales peruanos y su relación con los procesos geológico-ambientales de Sudamérica: Una revisión. *Caldasia* **2024**, *46*, 314–325. [CrossRef]
29. Díaz, A.; Sáenz-Zúñiga, L.; Zutta, B. Formación de un humedal en la costa norte del Peru: Estabilidad biofísica y diversidad biológica. *Rev. Peru Biol.* **2021**, *28*, e21132. [CrossRef]
30. Dirección de Geología Ambiental y Riesgo Geológico from INGEMMET. *Estudio Hidrogeológico del Refugio de Vida Silvestre Los Pantanos de Villa, Chorrillos—Lima*; Instituto Geológico, Minero y Metalúrgico—INGEMMET: Lima, Peru, 2019.
31. Verones, F.; Bartl, K.; Pfister, S.; Jiménez Vílchez, R.; Hellweg, S. Modeling the Local Biodiversity Impacts of Agricultural Water Use: Case Study of a Wetland in the Coastal Arid Area of Peru. *Environ. Sci. Technol.* **2012**, *46*, 4966–4974. [CrossRef]
32. Herrera, L.E.; Xu, H. Formación y evolución de la Laguna La Niña 2017, utilizando imágenes satelitales Modis-Aqua. *Bol. Inst. Mar Peru* **2019**, *34*, 105–114.
33. Pulido-Capurro, V.M. Estado actual de la conservación de los hábitats de los Pantanos de Villa, Lima, Peru. *Arnaldoa* **2018**, *25*, 679–702. [CrossRef]
34. Young, K.R. El Ecosistema. In *Los Pantanos de Villa: Biología y Conservación*; Universidad Nacional Mayor de San Marcos, Museo de Historia Natural: Lima, Peru, 1998; pp. 3–12.
35. Aponte, H.; Gonzales, S.; Gomez, A. Impulsores de cambio en los humedales de América Latina: El caso de los humedales costeros de Lima. *South Sustain.* **2020**, *1*, e023. [CrossRef]
36. Aponte, H.; Cano, A. Estudio florístico comparativo de seis humedales de la costa de lima (Peru): Actualización y nuevos retos para su conservación. *Rev. Latinoam. Conserv.* **2013**, *3*, 15–27.
37. Gonzales, S.; Aponte, H.; Cano, A. Update of the Vascular Flora of the Santa Rosa Wetland—Chancay (Lima, Peru). *Arnaldoa* **2019**, *26*, 867–882.
38. Aguirre-Soto, L.; Nieto-Juárez, J.I. Evaluación de La Calidad Del Agua En El Humedal Costero Pantanos de Villa, Chorrillos, Peru. *Tecnia* **2024**, *34*, 11–25. [CrossRef]
39. Benites-Alfaro, E.; Mendivil, V.T.; Farfán, E.E.; Rodriguez, C. Trophic Status Index of the Waters of the Pantanos de Villa Protected Natural Reserve, Chorrillos, Peru. *J. Crit. Rev.* **2020**, *7*, 447–453.
40. Huaman Vilca, S.; Lucen, M.; Paredes Vite, M.; Changanaqui Alfaro, D. Evaluación de la calidad del agua de la Laguna Marvilla en los Pantanos de Villa (Lima—Peru). *South Sustain.* **2020**, *1*, e019. [CrossRef]

41. Canales Cuadros, H.; Cabrera Carranza, C.; Arana Maestre, J. Macroinvertebrados acuáticos y calidad de agua en el Área de Conservación Regional Humedales de Ventanilla, Callao. *Rev. Inst. Investig. Fac. Minas Metal Cienc. Geogr.* **2022**, *25*, 295–301. [CrossRef]
42. Castillo-Velásquez, R.-M.; Huamantinco-Araujo, A.-A. Variación espacial de la comunidad de macroinvertebrados acuáticos en la zona litoral del humedal costero Santa Rosa, Lima, Peru. *Rev. Biol. Trop.* **2020**, *68*, 50–68. [CrossRef]
43. Córdova-Tello, C.A.; Huamantinco-Araujo, A.A. Efecto de las variables fisicoquímicas en la comunidad de coleópteros acuáticos en un humedal costero, Lima, Peru. *Acta Biol. Colomb.* **2023**, *28*, 204–219. [CrossRef]
44. Bopp, G.; Peláez, F. Evaluación de La Flora Vascular de Los Humedales Costeros de La Libertad, Peru. *Manglar* **2019**, *16*, 151–156. [CrossRef]
45. Jindo, K.; Morikawa, M. Innovative Feasibility Study for the Reclamation of the Cascajo Wetlands in Peru Utilizing Sustainable Technologies. *Water* **2020**, *12*, 1097. [CrossRef]
46. Aldana-Domínguez, J.; Rojas Quezada, C.; Munizaga, J.; Moschella, P.; Stamm, C.; Martínez, C. Urban Wetland Losses and Land-Use Conservation Challenges in Three Latin American Cities. In *Urban Wetlands in Latin America: Protection, Conservation, Innovation, Restoration, and Community for Sustainable and Water Sensitive Cities*; Rojas Quezada, C., Ed.; Springer Nature: Cham, Switzerland, 2024; pp. 3–18.
47. Aponte, H.; Coello-Sarmiento, M.-P.; Montes-Iturrizaga, D. New Insights About the Drivers of Change in the Coastal Wetlands of Peru: Results of a Rapid Field Survey. *Water* **2025**, *17*, 1473. [CrossRef]
48. Romero-Mariscal, G.; Garcia-Chevesich, P.A.; Morales-Paredes, L.; Arenazas-Rodriguez, A.; Ticona-Quea, J.; Vanzin, G.; Sharp, J.O. Peruvian Wetlands: National Survey, Diagnosis, and Further Steps toward Their Protection. *Sustainability* **2023**, *15*, 8255. [CrossRef]
49. Urbina, D.; Rivera-Cáceda, F.; Aponte, H. ¿Se Están Reduciendo Los Humedales de La Costa Del Pacífico Suramericano? El Caso de Los Humedales de Lima. *Rev. Acad. Colomb. Cienc. Ex. Fis. Nat.* **2022**, *46*, 985–998. [CrossRef]
50. Ramirez, D.W.; Aponte, H.; Cano, A. Flora vascular y vegetación del humedal de Santa Rosa (Chancay, Lima). *Rev. Peru. Biol.* **2011**, *17*, 105–110. [CrossRef]
51. Zamora-Aranda, A.; Aponte, H. Evaluation of Two Ecosystem Services Provided by a Pistia Stratiotes Population on the Pacific Coast of South America. *Biology* **2024**, *13*, 573. [CrossRef]
52. Castillo Polo, L.A. *Humedales Costeros de La Región Lima*, 1st ed.; Gobierno Regional de Lima: Lima, Peru, 2010.
53. Ochoa-Balbacea, D.O.; Gonzales, S.; Apeño, A.; Aponte, H. Hábitats prioritarios en un humedal costero del Pacífico: Propuesta de un índice basado en servicios ecosistémicos. *Rev. Acad. Colomb. Cienc. Exactas Físicas Nat.* **2022**, *46*, 182–191.
54. ANA Protocolo Nacional Para El Monitoreo de La Calidad de Los Recursos Hídricos Superficiales. 2016. Available online: https://sinia.minam.gob.pe/documentos/protocolo-nacional-monitoreo-calidad-recursos-hidricos-superficiales (accessed on 3 February 2022).
55. Baird, R.; Rice, E.W.; Eaton, A.D.; Bridgewater, L.; Federation, W.E. *Standard Methods for the Examination of Water and Wastewater*; American Public Health Association: Washington, DC, USA, 2017.
56. Hauer, F.R.; Lamberti, G.A. (Eds.) *Methods in Stream Ecology*, 2nd ed.; Academic Press/Elsevier: San Diego, CA, USA, 2006.
57. Titus, J.E. Submersed Macrophyte Vegetation and Distribution Within Lakes: Line Transect Sampling. *Lake Reserv. Manag.* **1993**, *7*, 155–164. [CrossRef]
58. Dengler, J.; Dembicz, I. Should We Estimate Plant Cover in Percent or on Ordinal Scales? *Veg. Classif. Surv.* **2023**, *4*, 131–138. [CrossRef]
59. Cirujano, S.; Meco Molina, A.; García Murillo, P. Flora Acuática Española. Hidrófitos Vasculares. Available online: https://bibdigital.rjb.csic.es/records/item/16128-flora-acuatica-espanola-hidrofitos-vasculares?offset=1 (accessed on 28 March 2025).
60. Sarkar, P.; Salami, M.; Githiora, Y.; Vieira, R.; Navarro, A.; Clavijo, D.; Padgurschi, M. A Conceptual Model to Understand the Drivers of Change in Tropical Wetlands: A Comparative Assessment in India and Brazil. *Biota Neotrop.* **2020**, *20*, e20190913. [CrossRef]
61. Oksanen, J.; Simpson, G.L.; Blanchet, F.G.; Kindt, R.; Legendre, P.; Minchin, P.R.; O'Hara, R.B.; Solymos, P.; Stevens, M.H.H.; Szoecs, E.; et al. Vegan: Community Ecology Package. Available online: https://cran.r-project.org/package=vegan (accessed on 12 December 2025).
62. Borcard, D.; Gillet, F.; Legendre, P. *Numerical Ecology with R*; Springer International Publishing: Cham, Switzerland, 2018.
63. Legendre, P.; Legendre, L. *Numerical Ecology*, 3rd English ed.; Developments in Environmental Modelling; Elsevier: Amsterdam, The Netherlands, 2012.
64. Mielke, P.W.; Berry, K.J. Description of MRPP. In *Permutation Methods: A Distance Function Approach*; Mielke, P.W., Berry, K.J., Eds.; Springer: New York, NY, USA, 2001; pp. 9–65.
65. Doledec, S.; Chessel, D. Co-Inertia Analysis: An Alternative Method for Studying Species-Environment Relationships. *Freshw. Biol.* **1994**, *31*, 277–294. [CrossRef]

66. Dray, S.; Dufour, A.-B.; Thioulouse, J.; Chessel, D.; Jombart, T.; Pavoine, S.; Lobry, J.R.; Ollier, S.; Borcard, D.; Legendre, P.; et al. Ade4: Analysis of Ecological Data: Exploratory and Euclidean Methods in Environmental Sciences R package version 1.7-23. Available online: https://cran.r-project.org/package=ade4 (accessed on 12 December 2025).
67. R Core Team. *R: A Language and Environment for Statistical Computing*; R Core Team: Vienna, Austria, 2020.
68. RStudio Team. *RStudio: Integrated Development for R*; RStudio: Vienna, Austria, 2020.
69. Agidie, A.; Wondie, A.; Beneberu, G. Wetland Disturbance Level and Macrophyte Composition in Natural Wetlands and Paddy Fields in the Eastern Part of Lake Tana, Ethiopia. *Heliyon* **2024**, *10*, e40701. [CrossRef]
70. Galatowitsch, S.M. Natural and Anthropogenic Drivers of Wetland Change. In *The Wetland Book*; Finlayson, C.M., Milton, G.R., Prentice, R.C., Davidson, N.C., Eds.; Springer: Dordrecht, The Netherlands, 2018; pp. 359–367.
71. Gomez Vilca, A.J.; Aponte, H.; Gonzales, S. ¿Cómo Proteger Los Humedales Costeros Peruanos? Una Respuesta a Partir de Un Modelo Conceptual de Sus Impulsores de Cambio. *Bol. Investig. Mar. Costeras* **2023**, *52*, 125–142. [CrossRef]
72. Pacioglu, O.; Tuṣa, I.M.; Sidoroff, M.E.; Iţcuṣ, C. The Best Management Practices in Agriculture for Protection of Inland Water Ecosystems. In *Encyclopedia of Inland Waters*, 2nd ed.; Mehner, T., Tockner, K., Eds.; Elsevier: Oxford, UK, 2022; pp. 58–67.
73. Meerhoff, M.; Beklioğlu, M. Chapter 26—Shallow Lakes and Ponds. In *Wetzel's Limnology*, 4th ed.; Jones, I.D., Smol, J.P., Eds.; Academic Press: San Diego, CA, USA, 2024; pp. 859–892.
74. Hadad, H.R.; Mufarrege, M.M.; Di Luca, G.A.; Maine, M.A. Salinity and pH Effects on Floating and Emergent Macrophytes in a Constructed Wetland. *Water Sci. Technol.* **2018**, *2017*, 270–275. [CrossRef]
75. Haller, W.T.; Sutton, D.L.; Barlowe, W.C. Effects of Salinity on Growth of Several Aquatic Macrophytes. *Ecology* **1974**, *55*, 891–894. [CrossRef]
76. Bianchini, I., Jr.; Cunha-Santino, M.; Panhota, R. Oxygen Uptake from Aquatic Macrophyte Decomposition from Piraju Reservoir (Piraju, SP, Brazil). *Braz. J. Biol.* **2011**, *71*, 27–35. [CrossRef]
77. Poi de Neiff, A.; Neiff, J.J.; Casco, S.L. Leaf litter decomposition in three wetland types of the Paraná River floodplain. *Wetlands* **2006**, *26*, 558–566. [CrossRef]
78. Bunch, A.J.; Allen, M.S.; Gwinn, D.C. Spatial and Temporal Hypoxia Dynamics in Dense Emergent Macrophytes in a Florida Lake. *Wetlands* **2010**, *30*, 429–435. [CrossRef]
79. Pereto, S.C.A.S.; Padial, A.A. Macrophyte Functional Composition Is Stable across a Strong Environmental Gradient of a Neotropical Floodplain. *Acta Bot. Bras.* **2021**, *35*, 62–69. [CrossRef]
80. Rolon, A.; Homem, H.; Maltchik, L. Aquatic Macrophytes in Natural and Managed Wetlands of Rio Grande Do Sul State, Southern Brazil. *Acta Limnol. Bras.* **2010**, *22*, 133–146. [CrossRef]
81. Amaral, M.; Bittrich, V.; Faria, A.; Anderson, L.; Aona, L. *Guia de Campo Para Plantas Aquáticas e Palustres Do Estado de Sao Paulo; Série Manuais Práticos em Biologia*; Holos, Editora: Sao Paulo, Brazil, 2008.
82. Muñoz, M.; Tinoco, K.; Arenas, J.; Aponte, H. Dinámica de La Planta Acuática Pistia Stratiotes y Su Incorporación a Un Sistema de Producción de Compost. *Technol. Water Sci.* **2026**, *17*, 77–123. [CrossRef]
83. Rejmankova, E. The Role of Macrophytes in Wetland Ecosystems. *J. Ecol. Field Biol.* **2011**, *34*, 333–345. [CrossRef]
84. Saluja, R.; Garg, J.K. Macrophyte Species Composition and Structure along Littoral Region in Relation to Limnological Variables of a Tropical Wetland Ecosystem. *Chem. Ecol.* **2017**, *33*, 499–515. [CrossRef]
85. Aponte, H.; Jiménez, R.; Alcántara, B. Challenges for Management and Conservation of Santa Rosa Wetland (Lima-Peru). *Cientifica* **2012**, *9*, 257–264.
86. Loayza-Basan, K.; Castillejos Paniagua, K.; Mestas Valero, R.; Quiliche Duran, J.P.J. Estudio de la Teledetección y Caracterización Fisicoquímica del Humedal "El Cascajo", Santa Rosa, Chancay-Lima, Peru. *Infinitum* **2017**, *7*, 52–58. [CrossRef]
87. Leiva, J.V.Y.; Ibarra, J.A. Evaluación de la calidad del agua en el canal Surco de la Zona de Reglamentación Especial del Humedal Los Pantanos de Villa (Lima, Peru). *South Sustain.* **2024**, *5*, e090. [CrossRef]
88. Mendoza-Caballero, W.; Quiroz-Santos, R.L. Evaluación de los estados tróficos de la laguna principal del área de conservación regional Albufera de Medio Mundo, Huaura—Lima. *Biotempo* **2022**, *19*, 149–164. [CrossRef]
89. Jyothi, P.V.; Sureshkumar, S. Patterns of Vegetation Dynamics across Mild Disturbance Gradient in a Freshwater Wetland System in Southern India. *Wetlands* **2018**, *38*, 807–817. [CrossRef]
90. Mullen, C.; Bertassello, L.E.; Rao, P.S.C.; Müller, M.F. From Wetlands to Wetlandscapes: Remote Sensing Calibration of Process-Based Hydrological Models in Heterogeneous Landscapes. *Hydrol. Process.* **2022**, *36*, e14739. [CrossRef]
91. Mitsch, W.J.; Gosselink, J.G. Wetland Definitions. In *Wetlands*, 5th ed.; Wiley: Hoboken, NJ, USA, 2015; pp. 27–44.
92. Cowardin, L.M.; Carter, V.; Golet, F.C.; LaRoe, E.T. *Classification of Wetlands and Deepwater Habitats of the U.S.*; U.S. Department of the Interior, Fish and Wildlife Service: Washington, DC, USA, 1992.
93. Domínguez-Machín, M.E.; Silva-López, G. Consumo de plantas por el ganado vacuno en una zona de Manglar-Espartal de Alvarado, Veracruz, México. *For. Veracruzana* **2020**, *22*, 7–11.
94. Adams, D.C. Evaluating Modularity in Morphometric Data: Challenges with the RV Coefficient and a New Test Measure. *Methods Ecol. Evol.* **2016**, *7*, 565–572. [CrossRef]

95. Lavado-Casimiro, W.S.; Felipe, O.; Silvestre, E.; Bourrel, L. ENSO Impact on Hydrology in Peru. *Adv. Geosci.* **2013**, *33*, 33–39. [CrossRef]
96. Gómez-Sánchez, R.; Cuba, D.; Aponte, H. Sobre la necesidad de descentralización y diversificación de la investigación en humedales costeros peruanos. *Biologist* **2022**, *20*, 121–150. [CrossRef]
97. Rivera, G.; Gonzales, S.; Aponte, H. Wetlands of the South American Pacific Coast: A Bibliometric Analysis. *Wetl. Ecol. Manag.* **2021**, *30*, 869–877. [CrossRef]

**Disclaimer/Publisher's Note:** The statements, opinions and data contained in all publications are solely those of the individual author(s) and contributor(s) and not of MDPI and/or the editor(s). MDPI and/or the editor(s) disclaim responsibility for any injury to people or property resulting from any ideas, methods, instructions or products referred to in the content.