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ORIGINAL RESEARCH ARTICLE

Agricultural Runoff Effects on Leaf Litter Decomposition: A Comparative Study in Natural and Constructed Deltaic Mediterranean Wetlands

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Abstract

Wetlands, widely distributed and hotspots of biodiversity, play a crucial role in global biogeochemical cycles and human well-being. However, despite their ecological importance, wetlands worldwide are under threat due to widespread conversion into agricultural fields, leading to changes in hydrology, increased salinity, and more frequent eutrophication. In response to these challenges, constructed wetlands are created to treat agricultural wastewater and mitigate eutrophication. This study aims to assess the effect of natural vs. constructed wetlands on ecosystem functioning (organic matter decomposition of the dominant vegetation: *Phragmites australis* and *Typha angustifolia*). We conducted this study in the Ebro River Delta (NE Spain), which represents a deltaic wetland affected by agricultural land-use changes, examining two constructed and two natural wetlands. Our findings indicate that the influence of agricultural runoff on the decomposition process was similar in both types of wetlands, suggesting that freshwater agricultural runoff has a consistent effect on ecosystem functioning, regardless of its origin, natural vs. constructed. Differences in macroinvertebrate communities associated with leaf litter were likely due to specific conductivity but did not impact decomposition rates. The estimated time required to decompose 95% of the *T. angustifolia* litter produced annually in the studied wetlands ranged from 288 to 856 days. In constructed wetland, this decomposition time exceeded one year, contributing to soil formation and carbon sequestration in wetland ecosystems. Our study suggests that the utilization of constructed wetlands for treating agricultural runoff can aid in mitigating the impacts of agricultural land use in these areas.

Keywords Agricultural Runoff · Coastal Wetlands · Land-use Change · Macroinvertebrates Communities · Organic Matter Decomposition

Introduction

Wetlands, widely distributed ecosystems across coastal areas in diverse settings such as estuaries and deltas (Barbier 2013), are of paramount importance for human well-being. They serve multiple vital functions: as hunting grounds, as nursery habitat for aquatic species critical to commercial fisheries, and as natural defenses against storms (Mitsch et al. 2015). Beyond their significance for human activities, wetlands are crucial for maintaining biodiversity, offering essential wildlife habitats (Bridgham et al. 2006; Barbier et al. 2011). Moreover, they play a pivotal role in global carbon processing (Duan et al. 2018; Ding et al. 2023), nutrient cycling and storage (Fennessy et al. 2008), flood control, aquifer recharge (Whatson et al., 2016), climate regulation, and act as natural filters (Brix 1995). They absorb runoff and filter surface water, making significant contributions to pollution control (Mitsch and Gosselink, 2007).

In deltaic Mediterranean wetlands, emergent macrophytes like *Phragmites*, *Typha* and *Juncus* dominate (Kuehn and Suberkropp 1998), producing substantial amount of organic matter (hereafter OM) that enhances soil accretion

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(Kirschner et al. 2001). The decomposition of this OM significantly influences the physical and chemical properties of wetland soils (Mitsch and Gosselink, 2007). These ecosystems are among the most productive ecosystems on Earth, with primary productivity in wetlands averaging twice that of other terrestrial ecosystems. For instance, wetlands contribute approximately 1800 gCm^{-2} $year^{-1}$, whereas tropical forests yield 800 gCm^{-2} $year^{-1}$, boreal forests 430 gCm^{-2} $year^{-1}$, and tundra only 180 gCm^{-2} $year^{-1}$ (Schlesinger 1997; Pace et al. 2021). Given their high productivity, regional biodiversity, and accessibility, these ecosystems have attracted humans settlement for millennia (Day et al. 2007, 2008), who have transformed them into agricultural fields, especially in the 20th century. This transformation led to the restructuring of primary-producer communities, resulting in shifts in food quality and the routes through which organic carbon is transferred to consumer (Lotze et al. 2006).

Changes in land use, coupled with farming practices like fertilization, often result in river and coastal flooding, compaction, subsidence, alterations in natural hydrology and salinity, eutrophication, and changes in land-water boundaries (Rogers et al. 2013). To mitigate the effects of eutrophication on natural ecosystems, one option is to construct wetlands for treating wastewater from agricultural fields (Brix et al. 2007; Kadlec and Wallace 2008). Constructed wetlands are considered a practical solution to water management challenges in the agricultural sector, and as a result, there are numerous operational constructed wetlands worldwide (Rozema et al. 2016). For instance, in the Ebro Delta, two free-water surface constructed wetlands were constructed to treat the water transported through a network of channels used for irrigating rice fields. These constructed wetlands have successfully reduced the average concentrations of ammonia, nitrite, nitrate, and phosphate by 85%, 94%, 58%, and 58% respectively (Matamoros et al. 2020).

So far, studies conducted in coastal lagoons and wetlands has shown that salinity exerts a negative effect on OM decomposition, primarily through its influence on the invertebrate community, especially shredders (Casagranda et al. 2006; Connolly et al. 2014). In contrasts, it has been reported that both nutrients and salinity can have a positive effect on microbial communities, potentially accelerating OM decomposition in freshwater wetlands (Asselen et al. 2013; Ping et al. 2018; Koceja et al. 2021). In the context of a deltaic Mediterranean coastal lagoon, the rate of OM decomposition was found to be higher during the period of rice field cultivation (0.0005–0.002 degree day^{-1} in spring-summer) compared to autumn-winter (0.0001–0.0007 degree day^{-1}) when the freshwater canals were closed (Menéndez et al. 2004). However, it's worth noting that this effect may vary across different coastal lagoons, depending on their origin, whether constructed or natural, and their specific environmental characteristics. For instance, certain biochemical properties of leaf litter, including their carbon and nitrogen content (e.g., C: N ratios), have been well-documented as significant factors influencing decomposition rates (Valiela et al. 1984; Enríquez et al. 1993). This is due to their influence on shredder preferences and consumption patterns (Arias-Real et al. 2018). Consequently, changes in environmental conditions resulting from freshwater irrigation during rice field cultivation, such as reduced salinity and increased nutrient availability, are likely to have an impact on both biodiversity (including microbial and invertebrate communities) and ecosystem processes, such as the decomposition of leaf litter. Nevertheless, the specific effects of constructed and natural wetlands on invertebrate communities and on the role of OM mineralization in the wetland C cycle remain unclear (Culler et al. 2014; Zhou et al. 2023), leaving some uncertainty in our understanding of these intricate ecological relationships. Kayranli et al. (2010) and Moreno-Mateos et al. (2012) reported that natural wetlands have a higher C sequestration capacity than constructed wetlands, and that restored wetlands are initially a source for C and that it might take decades for them to become sinks.

The objective of this study was to compare the OM decomposition in constructed vs. natural wetlands. To do that, we selected four coastal wetlands, comprising a rice field and a constructed wetland for wastewater treatment (both categorized as constructed wetlands), as well as a bay and a lagoon (both categorized as natural wetlands). These wetlands are located at The Ebro River Delta (NE Spain), which is affected by agricultural land-use changes. The dynamics of OM decomposition of leaf litter from two different plant species, Typha angustifolia and Phragmites australis, which have different carbon-to-nitrogen (C: N) ratios (53.3 in *T. angustifolia*, 36.6 in *P. australis*, (Ágoston Szabó and Dinka 2008), was assessed in order to find out if the quality of the leaf litter affects the decomposition rate and their associated invertebrate assemblages. Our hypotheses were twofold: Firstly, we anticipated a faster rate of decomposition and lower carbon sequestration in the rice field and constructed wetland compared to the natural wetlands. This expectation was based on the fertilization practices used in the rice field and the direct influx of effluents from the rice field into the constructed wetland. Secondly, we expected the impact of nutrient enrichment to be more pronounced in the case of *T. angustifolia*, due to its higher C: N ratios, compared to the decomposition of *P. australis* leaf litter.

Materials and Methods

Study Sites

This study was conducted in the Ebro Delta (330 km^2; hereafter referred to as Delta), in the north-western Mediterranean area (**Fig. 1**). Coastal Mediterranean climate experiences winter temperatures that are moderated by the sea, precipitation occurs mainly during the spring and fall (377 mm/year), and the water level can oscillate over 1 m due to the combined effects of tides, changes in the atmospheric pressure, and winds. The dominant vegetation are *P. australis* and *T. angustifolia*.

Agricultural fields (~62% of the total area), mainly rice fields, are the primary land use, while the natural areas (marshes, sand dunes, coastal lagoons, natural springs and bays) account for 25% of the total area and are directly affected by the fertilization and water regime of the agricultural fields. From May to September an artificial flood system is maintained to provide freshwater to the rice fields. Wastewater from the rice fields circulates through an extensive web of channels and flows out into the constructed and natural wetlands. The constructed wetland, approximately 86.9 ha, was created between 2010 and 2013 to treat the water for rice field irrigation by ACUAMED, the Spanish Ministry of Agriculture, Food and Environment. Thus, water salinity is determined by these freshwater inputs, which lead to relatively low specific conductivity during the cultivation period. Because of low conductivity, the littoral zones of the natural wetlands and channels are mainly covered by *P. australis* and *T. angustifolia.* These species are also the dominant vegetation in the constructed wetlands.

We selected five study sites distributed along the Delta and divided them into two groups: natural coastal wetlands (2 sites) and constructed wetlands (3 sites). The natural coastal wetlands included the Fangar Bay (40°46'20"N; 0°43'39"E) and the Encanyissada Lagoon (40°39'03"N; 0°41'34"E), while the constructed wetlands included two sites within the constructed wetland, the inflow (40°46'01"N; 0°44'05"E) and outflow (40°46'39"N; 0°43'04"E), and a rice field (40°45'10"N; 0°43'09"E) (**Fig. 1**).

Experimental Design

To analyse the effect of natural and constructed wetlands on leaf litter decomposition and invertebrate community diversity, we used the leaves of *T. angustifolia* and *P. australis* collected at the end of their vegetative cycle in November 2019 from the constructed wetland. Approximately 2 g of air-dried mass (entire leaves of *P. australis* and fragments

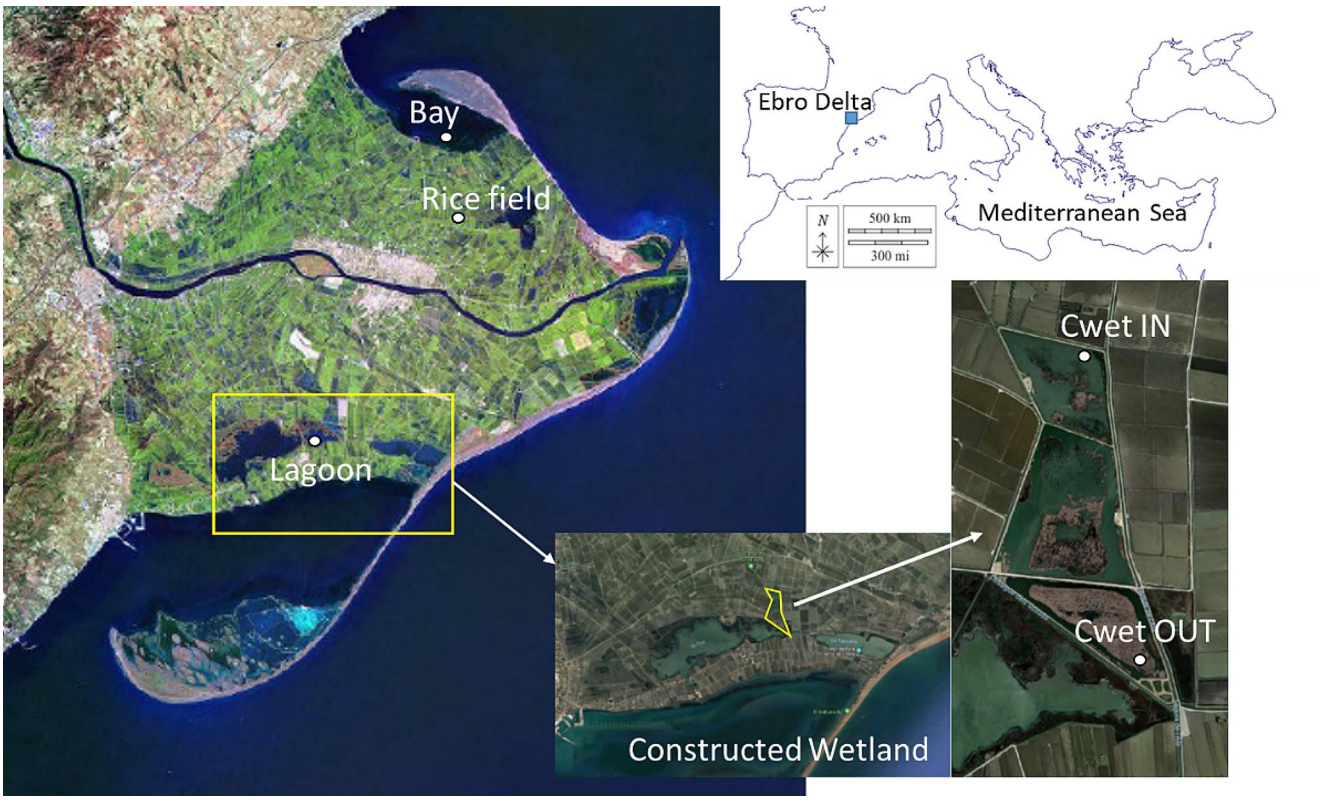

Fig. 1 Location of the study site along the Mediterranean coast, indicating the positions of the wetlands and the specific locations for the litter bag incubations. CWet IN: water input site in the constructed wetland. CWet OUT: water output site in the constructed wetland

of 10 cm of *T. angustifolia* leaves avoiding the apical parts) were enclosed in litter bags with a mesh size of 5 mm to allow access to invertebrates (Bárlocher et al. 2020). There were 18 litter bags at each sampling site ($n = 90$), 9 for each type of leaf litter, tied to strings and attached to a stake. The stakes were placed at all the sampling locations to simulate the natural dynamics of litter from plant senescence, i.e., first floating, then being submerged in the water column until its decomposition. Additionally, 5 litter bags for each type of leaf litter were analysed without being subjected to the immersion phase ($n = 10$).

At each sampling site, we conducted three sampling campaigns, during which we retrieved three litter bags of each type of leaf litter: the first after 9 days, the second after 30 days and the final after 66 days (these campaigns took place between May and July). At each time and location, we measured the specific conductivity of water, pH and dissolved oxygen using a WTW multiparametric sensor (WTW GmbH, Weilheim, Germany). After each retrieval, the litterbags were kept in zip-lock bags, tagged and transported refrigerated to the laboratory where they were processed. In addition, the water temperature was continuously monitored at each sampling point throughout the study period using HOBO Pendant temperature loggers (Onset Computer Corporation, Bourne, MA, USA).

Analytical Procedures

To characterize the invertebrate community, leaf material from each litter bag was rinsed with distilled water and the invertebrates from the litter bags were separated using a 500-µm mesh sieve and preserved in 70% ethanol. Individuals were counted and identified in the laboratory to the lowest possible taxonomic level (i.e. genus or family level) using a dissecting microscope. Identifications were carried out following Tachet et al. (2010).

To characterize leaf litter decomposition, the remaining material from each bag was weighed and dried at 60 °C for 48 h until constant weight. We determined the relationship between dry mass (DM) and fresh mass. A weighted fraction of dried litter was combusted at 400 °C for 5 h to determine the ash content in order to calculate the ash-free dry mass (AFDM). The decomposition rate, k, was calculated with a negative exponential model $W_t = W_o e^{-kt}$ (Olson 1963), where W_o is the initial percentage of the AFDM, W_t is the remaining AFDM at time t, k is the decomposition rate (degree-days^{-1}, dd^{-1}, or days^{-1}, d^{-1}) and t is expressed as the accumulated heat based on the mean daily temperatures between each sampling date (Stout 1989) or elapsed days of incubation. The time to take loss the 95% of the initial mass (t_{95}) was calculated from the decomposition rate in days ($t_{95} = 3/k$).

To analyze dissolved nutrients, water samples were taken and passed through glass fiber filters (Whatman GF/F). Nitrate concentration was assessed using ion chromatography (COMPACT IC1.1 Metrohm). Ammonium levels were determined via the manual salicylate method, while soluble reactive phosphorus (SRP) was measured using the molybdate method (APHA 2005).

Additional sub-samples were ground with an MM200 mixer mill and used to analyse litter nutrient content. Contents of C and N were determined from the initial content of each type of leaf litter using a CHNS Analyser (Carlo Erba Inc., Milan, Italy).

Statistical Analysis

We performed an analysis of covariance (ANCOVA) followed by Tukey's test for AFDM percentage remaining after each sampling date, with wetland and the type of leaf litter as fixed factors and the degree-days as a covariate to detect differences in the decomposition rates. An analysis of variance (ANOVA) followed by Tukey's test was used to determine significant differences in the number of macroinvertebrates associated with leaf litter. Statistical calculations were performed using the CSS-Statistica package and the subprogram ANOVA/MANOVA.

We used a non-metric multidimensional scaling ordination (nMDS) and Bray-Curtis dissimilarity to ordinate invertebrate communities based on relative abundances of individual taxa using PAST package (Hammer et al. 2001). Differences in invertebrate communities among the wetlands and litter type were assessed by analysis of similarities (two-way ANOSIM; Clarke 1993), followed by pairwise comparisons between the groups as a post-hoc test (Bonferroni correction) using the same program. Before all the statistical analyses, the distributional properties of the data were assessed to check the assumptions of normality and homoscedasticity. The Shapiro–Wilk test and Levene's test were applied to evaluate normality and homoscedasticity, respectively (Legendre and Legendre 1998). Data were log + 1 transformed when necessary to meet these assumptions. The significance level was set at $p < 0.05$ for all tests.

Results

The specific water conductivity varied highly among the wetlands, being the highest in the bay, followed by the lagoon, the constructed wetland and then the rice field (**Table 1**). The pH varied between the minimum value observed in the rice field (7.4) and the maximum value obtained in the lagoon (9.2). The dissolved oxygen also varied highly during the incubation period in all the wetlands except in the lagoon,

Table 1 Minimum-maximum values of the water characteristics of the studied wetlands throughout the study period. Cond: conductivity, O2: percentage of dissolved oxygen saturation, T: temperature. CWet IN: water input site in the constructed wetland. CWet OUT: water output site in the constructed wetland

	CWet IN	CWet OUT	Bay	Lagoon	Rice-field
Cond*, *mS/cm	1.7–2.2	1.8–2.9	11.2–47.7	4.1–13.2	0.7–1.4
pH	8.0-8.6	8.1–8.4	8.8–8.8	8.7–9.2	7.4–8.9
O_2*, *%	48–148	67–111	62–135	102–120	52–185
T*, *°C	21.3–25.2	20.5–25.2	23.6–27.0	21.2–26.8	21.4–26.0
NO_3^-*, *mg/L	0.01–0.27	0.01–0.11	0.01–0.30	0.01–0.11	0.10–5.74
NH_4^+*, *mg/L	0.01–0.27	0.01–0.12	0.01–0.03	0.04–0.14	0.01–0.73
PO_4^{3-}mg/L	0.006–0.024	0.003–0.020	0.005–0.019	0.007–0.014	0.033–0.067

where the lowest variability was detected and was always over 100% of oxygen saturation (Table 1). No remarkable differences were observed between the average daily water temperatures during the study period, which ranged from 20 to 27 °C.

Leaf Litter Decomposition

There were no significant differences observed among the decomposition rates in *T. angustifolia* between the wetlands (ANCOVA, $F = 1.59$ Tukey's HSD test, $P \geq 0.84$). However, for *P. australis*, differences were significant among the lagoon and the rice field, the water inlet cell (CWet IN) of the constructed wetland and the bay (ANCOVA, Tukey's HSD test, $P < 0.05$) (**Fig. 2**).

The time it takes to decompose the 95% of mass in the case of *P. australis*, it varies from 58 to 150 days. However, for *T. angustifolia*, ranges from 288 to 854 days, being greater than one year in wetlands constructed for the treatment of rice field effluents (854 and 444 days in the water inlet and outlet respectively).

The initial C: N ratio of leaf litter was three times higher for *T. angustifolia* (97.5) than for *P. australis* (27.7), showing a high nitrogen content in relation to carbon content in *P. australis* leaf litter when compared to *T. angustifolia* leaf litter. The decomposition rate (k, **Fig. 2**) was higher and more variable for *P. australis* (from 0.0007 to 0.0020 dd^{-1}, and from 0.0199 to 0.0512 d^{-1}) than for *T. angustifolia* (from 0.0002 to 0.0004 dd^{-1}, and from 0.0035 to 0.0104 d^{-1}; ANCOVA, $F = 58.36$, $P = 10^{-6}$) in all the studied wetlands.

Macroinvertebrate Assemblages

Thirty-six invertebrate taxa were found in the litter bags containing *T. angustifolia* and *P. australis* leaf litter in the wetlands studied. We did not find differences in the number of macroinvertebrates per litter bag between the two types of leaf litter ($F = 0.90$, $p > 0.05$). However, differences were observed between the sites (ANOVA, $F = 19.02$, $p < 0.05$), with higher quantities from the bay (Tukey's HSD test, $P = 0.001$) than for the rest of the wetlands (**Fig. 3**).

The non-metric multidimensional scaling analysis (**Fig. 4**) did not show differences in invertebrate communities between the two types of leaf litter (two-way ANOSIM, $R = 0.10$, $P = 0.21$). However, invertebrate communities were different among the bay, the lagoon and the constructed wetlands (rice field plus both cells of the constructed wetland) (two-way ANOSIM, $R = 0.79$, $P = 0.0001$; Bonferroni corrected P values < 0.05). In the constructed wetlands, the invertebrate community inside the litterbags was dominated by chironomids (more than 60%), accompanied by gastropods from the genus *Physa* in the rice field. The most abundant taxa in the bay (80%) were amphipods of the genus

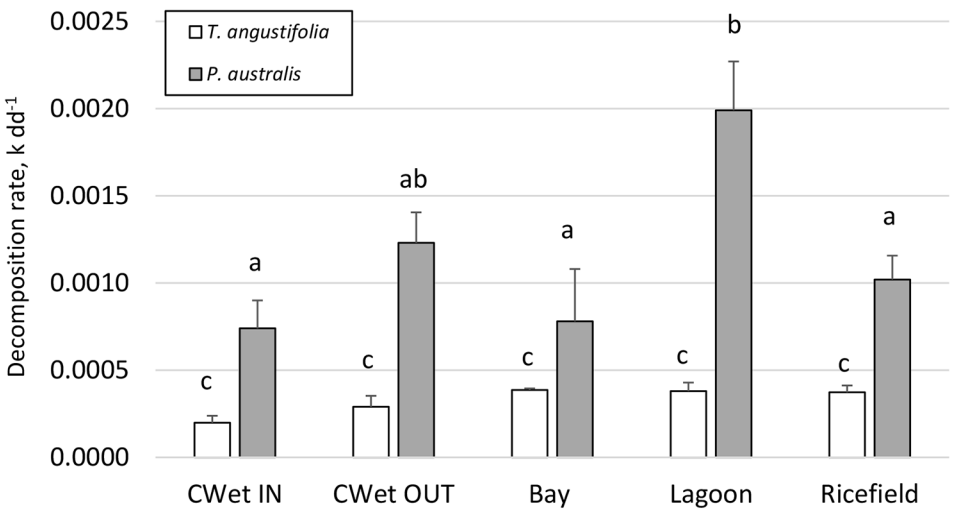

Fig. 2 Decomposition rates (k) expressed as mean ± SE, per degree-day basis (dd^{-1}) for *P. australis* and *T. angustifolia* in the examined wetlands. Different letters above the bars signify significant differences as determined by the ANCOVA analysis and subsequent Tukey's significant difference test ($P < 0.05$). CWet IN: water input site in the constructed wetland. CWet OUT: water output site in the constructed wetland

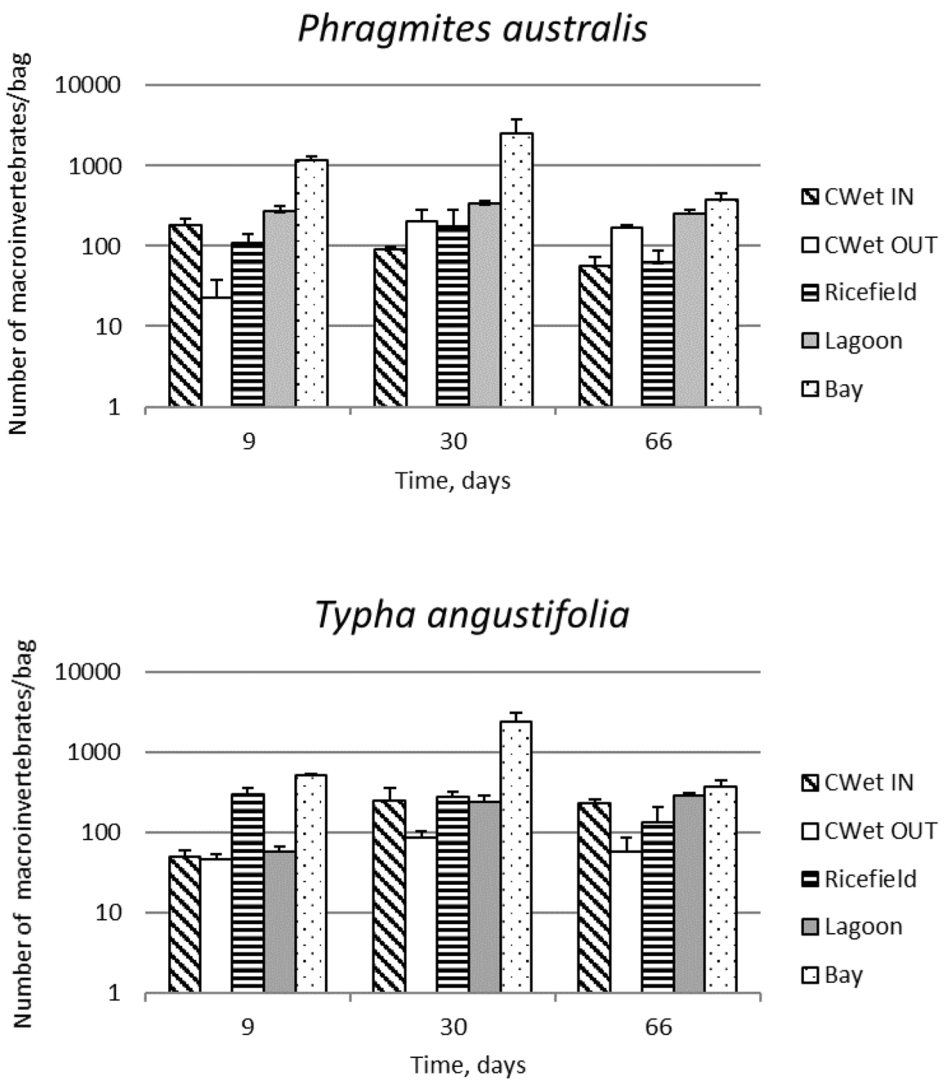

Fig. 3 Macroinvertebrate density (individuals/bag, mean ± SE) found in the studied wetlands. CWet IN: water input site in the constructed wetland. CWet OUT: water output site in the constructed wetland

Corophium, whereas the lagoon community was dominated by isopods from the species *Lekanesphaera hookeri*.

Discussion

Leaf Litter Decomposition in Constructed and Natural Wetlands

We hypothesized that the decomposition would be faster in constructed wetlands than in natural wetlands due to the fertilization practices in the rice fields and the effluent from these areas flowing into the constructed wetlands. However, this hypothesis was not supported by our results. No differences were observed between the types of wetlands in the rate of decomposition of *T. angustifolia* leaf litter. In the case of *P. australis*, the decomposition process was faster in the lagoon than in the rice field, the water inflow site of the constructed wetland and the bay. There were no differences observed in the decomposition rates of the two types of litter between the water inflow and outflow of the wetland constructed for treating agricultural discharges. However, the efficiency of nitrogen and phosphorus removal at the inlet and outlet sites averaged more than 80% and 58%, respectively (Matamoros et al. 2020). Therefore, our results indicated that the effect of agricultural runoff on the decomposition process and on the detrital pathway in our studied deltaic coastal wetlands was more or less uniform and independent of their origin. This observation suggests that the freshwater agricultural runoff received by all the wetlands during rice field cultivation tends to homogenize its effect on ecosystem function, although some differences were noted in environmental water characteristics such as conductivity. The differences in water conductivity were the main reason for the differences found in the macroinvertebrate communities of the leaf litter in the studied wetlands, although they

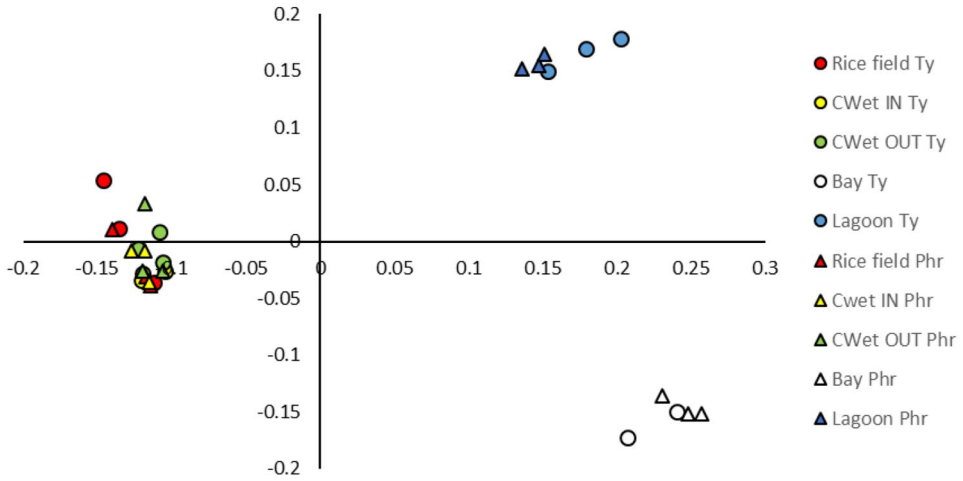

Fig. 4 Nonmetric multidimensional scaling ordination plot of macroinvertebrate assemblages colonizing *P. australis* (Phr) and *T. angustifolia* (Ty) leaf litter incubated in litterbags in the studied wetlands. CWet IN: water input site in the constructed wetland. CWet OUT: water output site in the constructed wetland. Stress: 0.14

were only marginally associated with decomposition. The density of macroinvertebrates found inside the litter bags did not correlate with the decomposition rates. The maximum density was observed in the bay, mainly linked to the amphipods from the genus *Corophium*, although no differences were observed in the decomposition rates among the wetlands studied (except the lagoon, but only for *P. australis*). Only in the studied lagoon with intermediate conductivity (4–13 mS cm^{-1}), the composition of macroinvertebrate assemblages seemed to directly correlate with the high rate of decomposition observed for *P. australis*, but not for *T. angustifolia.* In this sense, Quintino et al. (2009) found only small differences in the rates of decomposition of leaf litter in wetlands among salinity ranges less than 16%.

Effect of Leaf Litter Characteristics on the Decomposition Rate

The decomposition process depends not only on the physical and chemical characteristics of wetlands, but also on the biochemical properties of litter. One of these biogeochemical properties is the C: N ratio. Microorganisms have a high nitrogen content relative to litter and, as a consequence, the nitrogen demand for decomposition frequently exceeds the nutrient content of the litter (Enriquez et al., 1993). *Typha* has been suggested to be one of the toughest and most resistant emergent macrophytes to decomposition in wetlands (Alvarez and Bécares 2006). The higher C: N ratio of *Typha* compared to that of *Phragmites* found in our study confirms this suggestion, explaining the five-times lower decomposition rate found for *T. angustifolia* when compared to that for *P. australis.* Under these conditions of a lower nitrogen content relative to carbon content in the litter, dissolved inorganic nutrients may be an important determinant of the decomposition rate (Verhoeven et al. 1996). We expected that the fertilization in the rice fields and the consequent nutrient enrichment of the large amount of freshwater circulating among the wetlands would lead to more increased decomposition rates in *T. angustifolia* than in *P. australis.* This was not the case, as the decomposition rates of *T. angustifolia* in our study (0.0035–0.0104 d^{-1}) were similar to those previously published for *Typha* spp. in natural wetlands (0.001–0.0104 d^{-1}, Vymazal 1995) and constructed wetlands (0.0020–0.0043 d^{-1}, Alvarez and Bécares, 2001). Furthermore, no differences were found in the decomposition rates between the constructed and natural wetlands, although, as we have mentioned before, we would have expected the constructed wetlands to have received a higher nutrient load than the natural wetlands. It is highly probable that rice field effluents contain pesticides (fungicides and insecticides), which could adversely affect the activity of fungi and invertebrates, the primary decomposers of plant litter in these systems.

The *P. australis* decomposition rates in our study were higher (0.0007–0.0020 dd^{-1}) than those observed previously in the Tancada Lagoon in the Ebro Delta (0.0004–0.0008 dd^{-1}) (Menéndez et al. 2004). This is probably related to the litter bag mesh size used. We used 5 mm, while the study from the Tancada Lagoon used 2 mm, which was too small to give large invertebrate's access to the detritus inside. The highest decomposition rate found in our study in the lagoon with *P. australis* leaf litter was most likely related to the dominance of the shredder *L. hookeri* inside the litter bags, which contributes to litter decomposition by reducing its particle size, providing a greater surface area for leaching and microbial action (Casagranda et al. 2006). Similar results have been previously reported in other coastal wetlands and estuaries, where the dominance of some specific macroinvertebrate species determines the decomposition rate of leaf litter (Menéndez et al. 2019; Feio et al., 2021).

By utilizing the decomposition rates, we can calculate the time required for 95% of the leaf litter to decompose. For

T. angustifolia, this period exceeded one year in wetlands constructed for the treatment of rice field effluents, contributing to sediment accretion and carbon sequestration in the wetland. As a consequence, we can affirm that the replacement of some rice fields with wetlands could contribute to solving the subsidence problem of the Ebro Delta that has been observed in recent years (Fatorić and Chelleri 2012), conferring another important ecosystem service to wetlands constructed for the treatment of agricultural effluents in deltaic ecosystems. Day et al. (2011) concluded that coastal wetlands with high sediment input, mainly riverine, are the only ones likely to survive the predicted accelerated sea-level rise. However, the river deltas are highly vulnerable owing to the trapping of sediment in reservoirs upstream and floodplain engineering (Syvitski et al. 2005).

Conclusions

The effect of freshwater discharges from rice fields appears to result in a similar decomposition rate of OM, regardless of whether the wetlands are natural or constructed, and irrespective of their effect on the water conductivity. These differences in water conductivity were the main reason for the distinct macroinvertebrate assemblages in leaf litter in the studied wetlands, although they were only marginally associated with decomposition. Planting diverse species in constructed wetlands with different leaf-litter traits, such as C: N ratio, contributes to the availability of detritus over time for microbial communities and the detritivore's food web, enhancing biogeochemical cycles. We can affirm that the replacement of some rice fields for wetlands could contribute to solving the subsidence problem of the Ebro delta observed in the last years, adding another important ecosystem service to the constructed wetlands for the treatment of agricultural effluents in the deltaic ecosystems. In deltaic environments affected by agricultural land use, the replacement of part of the farming fields for constructed wetlands contributes to the creation of soil mitigating the effect of the climate change.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s13157-024-01837-0.

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Author Contributions Rebeca Arias-Real and Margarita Menéndez contributed to the study conception and design. Material preparation, data collection and analysis were performed by Rebeca Arias-Real, Xavier Herbera and Margarita Menéndez. The first draft of the manuscript was written by Rebeca Arias-Real and Margarita Menéndez and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

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Data Availability The datasets generated during the current study are available from the corresponding author on reasonable request.

Competing Interests The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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