



## Research

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**Author for correspondence:**

N. Pérez-Méndez

e-mail: [nestor.perez@irta.cat](mailto:nestor.perez@irta.cat)

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# Agricultural policies against invasive species generate contrasting outcomes for climate change mitigation and biodiversity conservation

N. Pérez-Méndez<sup>1</sup>, C. Alcaraz<sup>2</sup>, A. Bertolero<sup>3</sup>, M. Català-Forner<sup>1</sup>,  
L. A. Garibaldi<sup>4,5</sup>, J. P. González-Varo<sup>6</sup>, S. Rivaes<sup>7</sup> and M. Martínez-Eixarch<sup>2</sup>

<sup>1</sup>Sustainable Field Crops Program, IRTA-Amposta, Tarragona, Spain

<sup>2</sup>Marine and Continental Waters Program, IRTA-La Ràpita, Tarragona, Spain

<sup>3</sup>Associació Ornitològica Picapall de les Terres de l'Ebre, Tarragona, Spain

<sup>4</sup>Universidad Nacional de Río Negro, Instituto de Investigaciones en Recursos Naturales, Agroecología y Desarrollo Rural, San Carlos de Bariloche, Argentina

<sup>5</sup>Consejo Nacional de Investigaciones Científicas y Técnicas, Instituto de Investigaciones en Recursos Naturales, Agroecología y Desarrollo Rural, San Carlos de Bariloche, Argentina

<sup>6</sup>Departamento de Biología, IVAGRO, Universidad de Cádiz, Puerto Real, Spain

<sup>7</sup>SEO/Birdlife, Tarragona, Spain

NP, 0000-0001-6264-2920; CA, 0000-0002-2147-4796; LAG, 0000-0003-0725-4049

Direct consequences of biological invasions on biodiversity and the environment have been largely documented. Yet collateral indirect effects mediated by changes in agri-environmental policies aimed at combating invasions remain little explored. Here we assessed the effects of recent changes in water management in rice farming, which are aimed at buffering the impact of the invasive apple snail (*Pomacea maculata*, Lamarck) on greenhouse gas emissions and diversity of waterbird communities. We used observational data from a 2-year field monitoring (2015–2016) performed at the Ebro Delta regional scale. We found that drying rice fields reduced methane emission rates by 82% (2015) and 51% (2016), thereby reducing the contribution of rice farming to climate change. However, there was a marked reduction (75% in 2015 and 57% in 2016) in waterbird diversity in dry fields compared with flooded fields, thus suggesting that post-invasion policies might hinder biodiversity conservation. Our results highlight the need for accounting for potential collateral effects during the policy decision-making process to design efficient agricultural management plans that lessen undesirable agri-environmental outcomes.

## 1. Introduction

Biological invasions are considered a major driver of global change that entail marked environmental and socio-economic costs worldwide [1–3]. In the European Union, economic losses in agriculture associated with the introduction of invasive species amounted to US\$36 billion during 1960–2020 [4]. Direct effects of invasive species on crop production are mostly mediated by weeds, pests and plant pathogens [5]. Strategies for reducing invasive pests include chemical and biological resources, but also changes in management practices which are promoted through agri-environmental policies (AEPs) [3]. Environmental outcomes resulting from AEPs may be controversial as new remedial actions may trigger collateral effects on biodiversity and on biogeochemical cycles. For example, changes of agricultural management at large spatial scales may modify habitat availability for native non-target species [6] as well as the capacity of agroecosystems for carbon sequestration or greenhouse gas emissions [7]. Yet the potential side effects of AEPs on multiple global

environmental issues are rarely considered by policy makers when designing invasive species management plans.

Rice (*Oryza sativa*, L) is a globally important semi-aquatic crop with important implications for global food security, climate change and biodiversity conservation [8]. Flooded rice fields (i.e. rice is grown under flooded conditions) occupy around 12% of the global cultivated area (approx. 164 million hectares) and represent the principal source of food for one third of the world's population [9]. Rice farming is also recognized as one of the main agricultural sources of greenhouse gas emissions, as decomposition of organic matter under anaerobic soil conditions (i.e. flooding conditions) promotes high rates of methane (CH<sub>4</sub>) emission [10–12]. Finally, rice agroecosystems are biodiversity hotspots with special relevance for waterbird conservation as flooding fields act as unique artificial wetlands that provide feeding and breeding habitat to a broad range of species worldwide [13–17]. Strategies for reducing crop damage caused by aquatic invasive pests such as the apple snail (*Pomacea maculata*, Lamarck) include modifications of the flooding dynamics at the landscape scale (e.g. long periods of field drying), which are subsidized by regional governments through specific AEPs [18,19]. Importantly, changes in water management could entail side effects leading to positive outcomes for greenhouse gas emissions but negative outcomes for waterbird conservation. Yet these collateral effects of management actions against invasive species remain largely underexplored, and therefore are not usually considered by policy makers.

In this study, we used the rice agroecosystem of the Ebro Delta (northeast Spain) as a case study to explore the potential side effects of the application of regional policies to control the invasive apple snail (i.e. post-harvest drying of rice fields) on methane emissions and the diversity of avian communities. Before the accidental introduction of the apple snail in winter 2009/2010, rice farmers were subsidized through AEPs for maintaining rice fields flooded during the post-harvest season (i.e. winter) as the region is one of the three most important wetlands in western Europe in terms of waterbird diversity. As a result, more than 65% of the agricultural surface remained flooded during winter, providing large amounts of habitat for wintering waterbirds [18]. However, the large-scale winter flooding also acted as an important source of methane emission in the region [11,12]. To help in controlling apple snail populations, which cause serious rice yield losses, post-invasion AES policies did not include payment to rice farmers for flooding rice fields, entailing a marked reduction in the flooding surface at the Ebro Delta scale (figure 1). Here, to evaluate the potential side effects of water management changes on methane emission and avian diversity, we used an observational approach based on two-year field monitoring. We specifically characterized methane emission rates and bird diversity throughout the post-harvest season considering two contrasting scenarios in terms of flooding dynamics (flooded versus dry). We hypothesized that the management policies against the apple snail have an indirect positive effect on climate change mitigation (i.e. reduction of methane emission rates), as draining of rice fields introduce aerobic soil conditions that inhibits methane emissions. On the other hand, we expected a negative effect on biodiversity conservation (i.e. reduction of species richness and evenness of communities) as drying fields would provide less suitable habitat for waterbird species. We additionally tested whether the hypothesized

negative effect of field drying on waterbird diversity could be compensated by an increase of non-waterbird diversity.

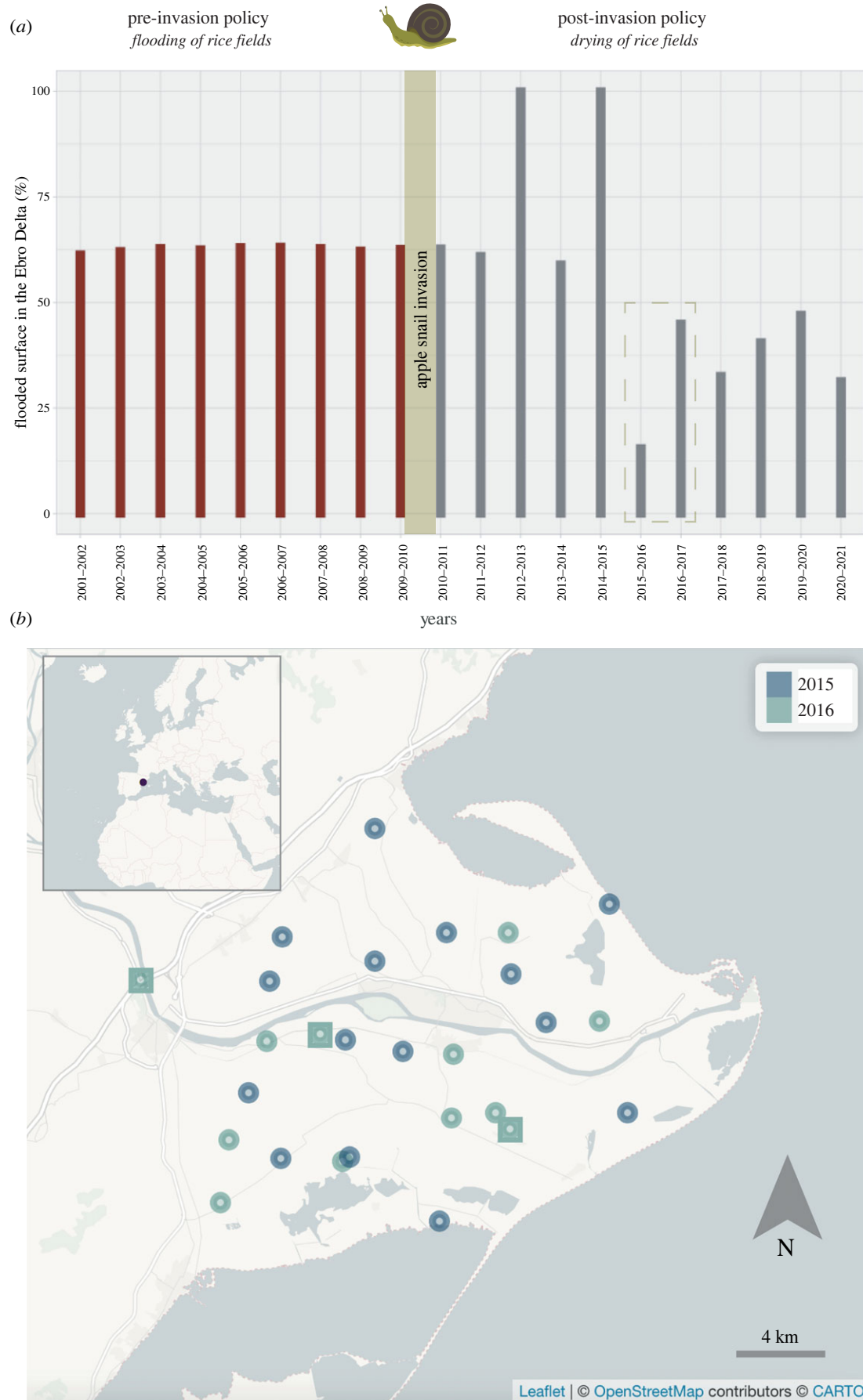
## 2. Material and methods

### (a) Study system

The study was performed in the Ebro Delta, a 320 km<sup>2</sup> area located in northeast Spain (Catalonia, figure 1). The Ebro Delta is considered one of the most important wetlands in the north-west Mediterranean. Around 25% of the region (approx. 8000 ha) is protected as a Natural Park and is included in the Natura 2000 network, in the Ramsar convention and as part of Terres de l'Ebre Biosphere Reserve [16,20]. On the other hand, about 65% of the territory (21 125 ha) is dominated by an agricultural matrix aimed at growing rice during the spring–summer season (from May to September) [21]. Because rice is cultivated under flooded conditions, this agricultural matrix act as an artificial wetland during the rice-growing season. In addition, the Ebro Delta acts as a wintering ground for more than 250 000 waterbird individuals annually foraging in the area (seeds, aquatic plants and invertebrates and fishes) [22]. Yet the availability of foraging habitat for wintering birds largely depends on the water management of rice fields during the post-harvest period (i.e. November to February). Until 2010 post-harvest flooding (of roughly 63% of the overall agricultural surface) was promoted as an environmentally friendly practice supported by the Agri-Environmental Schemes of the European Commission. Implementation through regional agricultural policies included financial support to rice farmers and was aimed at providing wintering waterbird habitat, but also at promoting other agronomic benefits (e.g. straw decomposition) and duck hunting grounds [23–26]. However, the accidental introduction of the invasive apple snail (*P. maculata*, Lamarck) in winter 2009/2010 [27,28] dramatically changed this scenario. Apple snail, which is included among the world's worst invasive species because of its huge impact on rice yields worldwide [27,29], rapidly spread throughout the Ebro Delta. As a response, the post-invasion regional policy (PDR 2014–2020) did not incorporate the AEP's winter flooding as a funded agricultural practice, thus promoting the interruption of flooding during winter. This helped to slow down the invasion process and reduce the agricultural impact during the first stages of the pest invasion process [30], yet it drastically reduced the amount of flooded surface at the regional scale (pre-invasion period = 63%; overall post-invasion period = 55%; last 6 years = 35%; figure 1). Given that most waterbird species surveyed use disproportionately rice fields, natural wetlands (approx. 2500 Ha) and/or salt mines (approx. 1000 Ha) as foraging habitats [22], post-harvest field drying reduced the suitable habitat at the delta regional scale from approximately 16 800 Ha (pre-invasion period) to 10 890 Ha (the last 6 years). It is important to note, however, that despite the application of post-harvest field drying during more than a decade, the apple snail has already occupied all the northern side of the delta and it is currently expanding in a scattered way throughout the southern side [30].

### (b) Characterization of methane emission rates

Methane emission rates (mg m<sup>-2</sup> h<sup>-1</sup>) were estimated in 24 rice fields ( $n = 15$  in 2015;  $n = 9$  in 2016; figure 1), averaging  $2.15 \pm 0.08$  (mean  $\pm$  s.d.) hectares. Fields were selected to represent two contrasting scenarios from the point of view of the flooding state: dry fields (i.e. water layer height = 0 cm in more than 95% of the field surface) versus flooded fields (i.e. water layer height greater than 0 cm in more than 95% of the field surface) (electronic supplementary material, figure S1). It is worth noting that flooding state of a given field can change across the



**Figure 1.** Study system. (a) Flooding patterns promoted by AEPs in the Ebro Delta to control populations of the invasive apple snail (*P. maculata*). Bars show the temporal distribution of flooded surface (%) relative to total rice surface in the Ebro Delta during the post-harvest rice seasons in both the pre-invasion and post-invasion period. Flooding data were obtained from Pernollet *et al.* [18] (period 2002–2012) and from the Generalitat de Catalunya (period 2016–2021) (<https://govern.cat/govern/docs/2021/02/22/13/01/c10b629b-e1fd-4f35-8a7e-ecb7d145398e.pdf>). The framed bars with a dashed line indicate the sampling years of this study. (b) Spatial and temporal distribution of sampling locations in the Ebro Delta. Circles represent rice fields where greenhouse gas emissions and avian diversity were quantified, whereas squares indicate fields where only avian diversity was characterized. (Online version in colour.)



post-harvest season, thus the flooding state of the studied fields was characterized in each fortnightly survey. Sampling locations were widely distributed across the Ebro Delta to capture environmental variability in terms of biotic and abiotic conditions at the regional scale (figure 1).

Only methane was considered as previous studies showed that emissions of additional greenhouse gases (i.e. nitrous oxide and carbon dioxide) are negligible during the post-harvest season in our study system [12,31]. Three sampling points were established for each rice field, which were randomly distributed across the field. The height of the water layer was also measured in the same sampling points. Gas sampling was conducted on a monthly basis from October to December, using non-steady-state gas chambers [11,32]. The characteristics of the chambers as well as the procedure for chamber deployment and field sampling plan are detailed by Martínez-Eixarch *et al.* [11]. In brief, the chambers, were made of polyvinylchloride (PVC) structure covered by transparent plastic and they were equipped with a thermometer to monitor temperature within the chamber in each gas sample extraction. To avoid soil disturbance during gas sampling, blocks were installed in the field to support wooden boards to access the chamber. All the rice fields were randomly sampled within the same day and consistently from 10.00 to 15.00 to minimize variability derived from the daily emission variation [33]. During the sampling procedure, each gas sample was transferred overpressured to pre-evacuated 12.5 ml vials (Labco Ltd., Buckinghamshire, UK) and sent to laboratory. Methane (CH<sub>4</sub>) concentration was determined using a Thermo Trace 2000 (Thermo Finnigan Scientific, USA) gas chromatograph equipped with a flame ionization detector (GC-FID). The calibration of the gas chromatograph was carried out using a CH<sub>4</sub> standard in nitrogen provided by Carbueros Metalicos SA (Spain). The emission rates of methane were obtained from the change of concentration of the respective gas in chambers over the 30 min sampling period in each chamber. The emission rate was estimated by the slope of the linear regression between gas concentration and sampling time. The increase of temperature in the head-space of the chamber was considered to correct methane concentration of each sample.

### (c) Characterization of avian communities

Characterization of avian communities was conducted during the post-harvest rice season (i.e. winter, November to January) in 2015 and 2016. We selected a total of 27 rice fields ( $n = 15$  in 2015;  $n = 12$  in 2016); i.e. the same 24 fields selected for estimating greenhouse gas emissions plus three additional fields in 2016. Diurnal bird surveys were performed every two weeks, with the first sampling date (i.e. 11 November 2015 and 7 November 2016) coinciding with the third fortnight after harvesting ( $n = 6$  fortnightly surveys per year for two years, totaling 12 fortnightly surveys). With the help of binoculars and a telescope all recorded individuals during 15 min censuses were identified at the species level by expert ornithologists of the research group (A. Bertolero & S. Rivaes) [34]. In addition, each species was categorized as a waterbird or a non-waterbird species. Waterbirds included gulls (Fam. Laridae), herons and ibis (Fam. Ardeidae and Threskiornithidae) and waders (Orders Charadriiformes and Gruiformes), whereas the non-waterbirds included songbirds (Order Passeriformes), raptors (Order Falconiformes) and an assortment of a few surveyed species that cannot be included in any of the previous categories ('others' functional group) [35] (electronic supplementary material, table S1). The functional groups are composed by phylogenetically related species (same family or order) with shared common functional traits (e.g. body size or foraging behaviour), except for the 'others' group (see [36,37] for similar aggrupations).

### (d) Field and landscape characterization

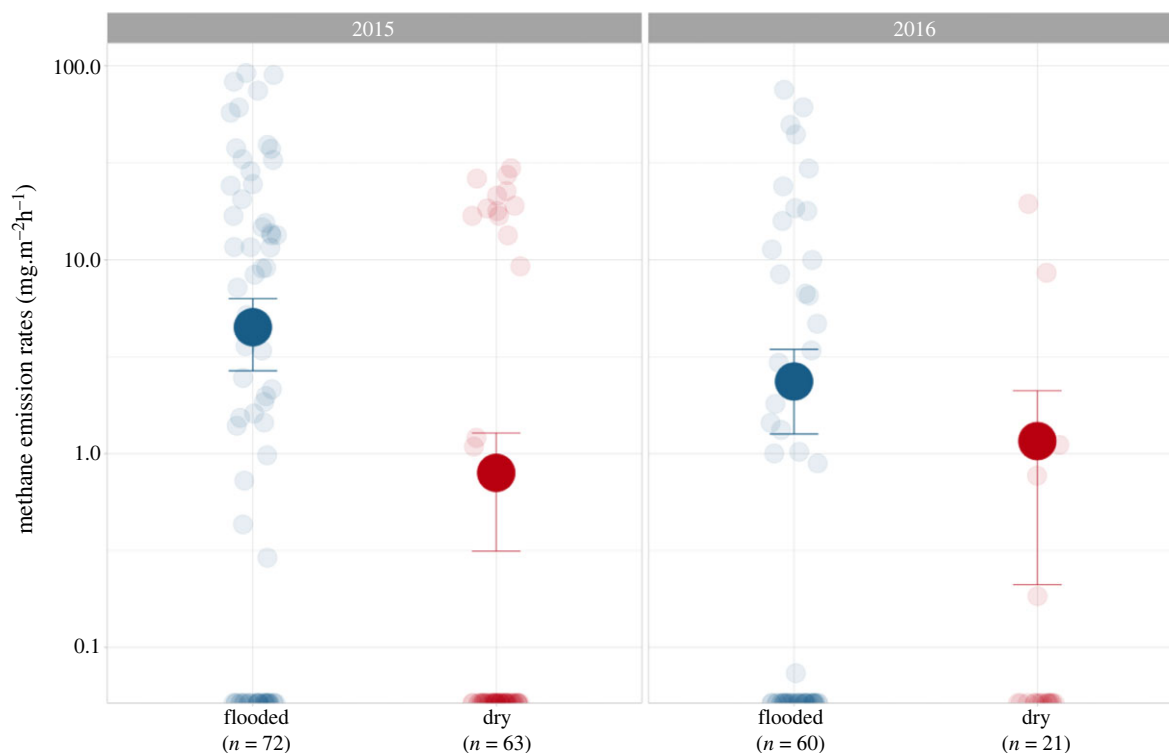
In every single visit, the following habitat characteristics were characterized at the field scale as they are expected to influence bird diversity: (i) two semi-quantitative measures indicating the proportion of the field surface with water and with straw residues (i.e. 0%, 5%, 25%, 50%, 75% and 100% for each measure), (ii) the mean height of the water layer (average of three random points for each field), and (iii) whether the rice straw were or were not incorporated into the soil. In order to control for the landscape influence on avian diversity patterns, we also characterized the composition of the surrounding habitat for each focal field (1 km buffer). We used the QGIS software and the land-use shape layer (10 m resolution) provided by the regional government of Catalonia (<https://territori.gencat.cat/>; year 2017). Specifically, we estimated the surface area (ha) occupied by the following habitat features: (i) rice matrix, (ii) agricultural matrix (no rice), (iii) natural wetlands, (iv) urban areas and (v) other habitat features (which included shrublands and coastal habitat).

### (e) Data analysis

To evaluate the effect of water management during the post-harvest season on methane emission rates, we applied a generalized linear mixed-effect modelling (GLMM) approach. The model included the rate of methane emission as the response variable and the interaction between the flooding state (i.e. dry versus flooded) and sampling year as the main fixed factors ( $n = 216$  observations). We included this interaction to test for temporal consistency in the effects of water management on methane rates. The initial temperature in the chamber and the height of the water layer were included as covariates. Finally, because we established three repeated measures (sampling units) per plot and fortnightly survey, we included a nested random factor with fortnightly survey nested within plot identity. Because data distribution was markedly left-skewed with clustered zero values (i.e. no methane emission), we used a tweedie distribution with a *log* link function [38].

To evaluate the impact of changes of winter flooding patterns on waterbird diversity, we firstly estimated both species richness and Hill evenness (i.e. a modified Simpson index or  $q_2$ ) for each sampled rice field across fortnightly surveys by using the iNext R package (Hill numbers) [39–41]. Both components are complementary and summarize properly the biodiversity profiles of sampled rice fields. Richness indicates the total number of species detected whereas Hill evenness is interpreted as the effective number of dominant species in the community [39,41]. We then applied two GLMMs for modelling species richness ( $n = 162$  observations) and Hill evenness ( $n = 162$  observations), respectively. Both models included the interactions between flooding state and sampling year and between flooding state and fortnightly survey as main fixed factors, whereas local (i.e. field size, height of the water layer and the incorporation of straw residues) and landscapes variables (i.e. amount of rice matrix in the 1 km buffer) were included as covariates. Given the high correlation detected among the rest of local and landscape variables, they were not included in the model to avoid potential multicollinearity issues (electronic supplementary material, figure S2). Both models included the plot identity as a random factor whereas a negative binomial and a Gaussian distribution were used for the species richness and evenness model, respectively. All above-mentioned analyses were repeated by including not only the diversity of waterbirds, but also the diversity of the whole avian community (i.e. waterbirds + raptors + songbirds + 'others').

In order to assess how flooding patterns affected the abundance of the different functional groups of birds, we also applied a GLMM by using the overall abundance (i.e. the sum of abundances across fortnightly surveys) of each species in each flooding state as the response variable ( $n = 208$  observations).



**Figure 2.** Methane emission rates ( $\text{mg m}^{-2} \text{h}^{-1}$ ) in dry (red points; 2015:  $n = 63$ ; 2016:  $n = 21$ ) and flooded rice fields (blue points; 2015:  $n = 72$ ; 2016:  $n = 60$ ) in the Ebro Delta. Small semi-transparent points indicate observed methane emission rate for each field and fortnightly survey. Large solid points and error bars indicate the estimated emission rate and the standard errors provided by the GLMM, respectively. Note the log-scaled y-axis. (Online version in colour.)

The interaction between flooding state and the functional group, and the interaction between flooding state and the sampling year, were included as fixed factors. Then, we incorporated the identity of the species as a random factor and used a negative binomial distribution of errors. Finally, to test if, as hypothesized, reduced avian diversity in dry fields is compensated by an increase of avian biomass, we estimated the overall biomass of bird communities for each plot and fortnightly survey. We obtained the weight (kg) for each surveyed species from the AVONET database [42], and then multiplied by their abundances. We then applied a GLMM by using the total biomass of birds for each fortnightly survey and plot as the response variable ( $n = 162$  observations). The plot surface and the interactions between flooding state and year and flooding state and fortnightly survey were incorporated as fixed factors. Avian biomass was square root transformed to improve model fitness. Finally, the identity of the plot was incorporated as a random factor and a tweedie distribution with a log link function was specified. We used R software (v4.1.2) [43] and the *glmmTMB* R package to perform all the GLMMs [44]. In addition, the *DHARMA* package was used to check for potential patterns in model residuals by using both a visual inspection of residual plots and the formal tests provided by the library [45]. We also used *emmeans* for computing contrast between factor levels [46], and *tydiverse* for both data management and visualization [47].

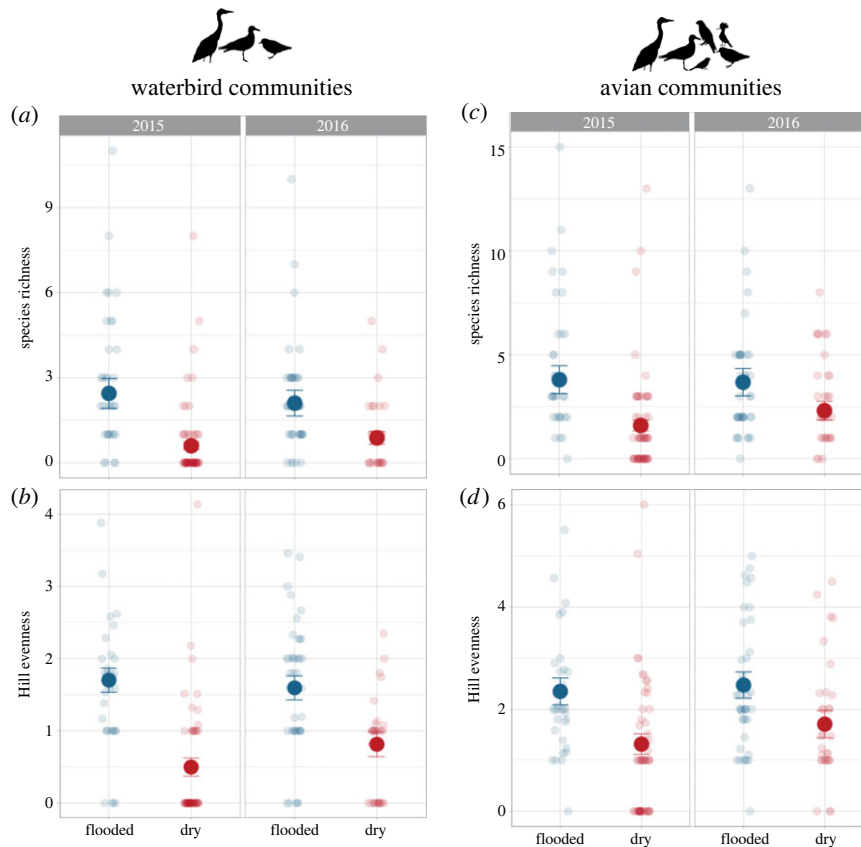
### 3. Results

We found evidence that, during the post-harvest rice period, field drying reduces methane emission rate when compared to flooded fields ( $\chi^2 = 5.4$ ,  $p = 0.019$ ; figure 2; electronic supplementary material, table S2). This pattern was consistent across years as shown by the non-significant interaction between the year of sampling and flooding state of fields ( $\chi^2 = 0.9$ ,  $p = 0.340$ ).

Specifically, the percentage reduction in methane emission rates in dry fields when compared to flooded fields was 82% in 2015 (flooded fields =  $4.5 \pm 1.8 \text{ mg m}^{-2} \text{h}^{-1}$  versus dry fields =  $0.8 \pm 0.5$ ) and 51% in 2016 ( $2.4 \pm 1.1 \text{ mg m}^{-2} \text{h}^{-1}$  versus  $1.2 \pm 0.9$ ). In addition, we found statistical evidence that the height of the water layer had a significant negative effect on methane emission rates ( $\chi^2 = 5.4$ ,  $p = 0.018$ ), whereas temperature had a strong positive effect ( $\chi^2 = 16.2$ ,  $p < 0.001$ ).

Our results showed very strong evidence of reduced waterbird richness in dried fields when compared to flooded fields ( $\chi^2 = 26.8$ ,  $p < 0.001$ ; electronic supplementary material, table S3) during the post-harvest season. This pattern was consistent between years as evidenced by the non-significant effect of the sampling year  $\times$  flooding state interaction ( $\chi^2 = 2.3$ ,  $p = 0.132$ ). On average, avian richness was reduced by ca 75% in 2015 (flooded fields =  $2.5 \pm 0.5$  versus dry fields =  $0.6 \pm 0.1$ ) and by 57% in 2016 ( $2.1 \pm 0.4$  versus  $0.9 \pm 0.2$ ). The GLMM output showed a statistically significant interaction between fortnightly survey and flooding state ( $\chi^2 = 15.8$ ,  $p < 0.001$ ), showing a positive relationship between avian richness and fortnightly survey for flooded fields ( $t = 2.7$ ,  $p = 0.008$ ) but a negative relationship for dry fields ( $t = -2.9$ ,  $p = 0.004$ ) (electronic supplementary material, figure S3). Finally, there was a strong positive effect of field size on waterbird richness ( $\chi^2 = 18.5$ ,  $p < 0.001$ ), and the extent of the rice matrix around focal fields did not influence it ( $\chi^2 = 0.1$ ,  $p = 0.779$ ). The results were quite consistent when we included also the non-waterbird species in the analysis (i.e. raptors, songbirds and 'others' groups) (figure 3; electronic supplementary material, table S4).

We consistently found that species evenness (i.e. Hill evenness) was also lower in dry fields than in flooded fields for both years ( $\chi^2 = 34.5$ ,  $p < 0.001$ , electronic supplementary



**Figure 3.** Avian diversity patterns in dry (2015:  $n = 57$ ; 2016:  $n = 31$ ) and flooded rice fields (2015:  $n = 33$ ; 2016:  $n = 41$ ) in the Ebro Delta. Left panels (*a,b*) show diversity patterns for waterbird communities, whereas the right panels (*c,d*) show patterns for the whole avian communities (i.e. also including songbirds, raptors and the 'others' functional groups). Small semi-transparent points indicate observed richness (*a,c*) and Hill evenness (*b,d*) for each field and fortnightly survey. Large solid points and error bars indicate the estimated means and the standard errors provided by the GLMMs, respectively. (Online version in colour.)

material, table S3). Specifically, we observed a reduction of evenness by 71% (flooded fields =  $1.7 \pm 0.2$  versus dry fields =  $0.5 \pm 0.1$ ) in 2015 and by 49% in 2016 ( $1.6 \pm 0.2$  versus  $0.8 \pm 0.2$ ). The sampling year  $\times$  flooding state interaction was not statistically significant ( $\chi^2 = 2.7$ ,  $p = 0.097$ ), suggesting that the patterns were consistent for both years. Fortnightly survey and flooding state showed a statistically significant interaction mirroring the results for species richness ( $\chi^2 = 13.8$ ,  $p < 0.001$ ), i.e. a positive relationship in flooded fields ( $t = 3.2$ ,  $p = 0.002$ ) and a negative trend in dry fields ( $t = -1.9$ ,  $p = 0.050$ ) (electronic supplementary material, figure S3). Field size also positively influenced species evenness ( $\chi^2 = 8.0$ ,  $p = 0.005$ ), whereas the amount of rice matrix showed no evidence of influence ( $\chi^2 = 0.001$ ,  $p = 0.972$ ). The results remained consistent when the additional non-waterbird species were included in the GLMM (figure 3; electronic supplementary material, table S4).

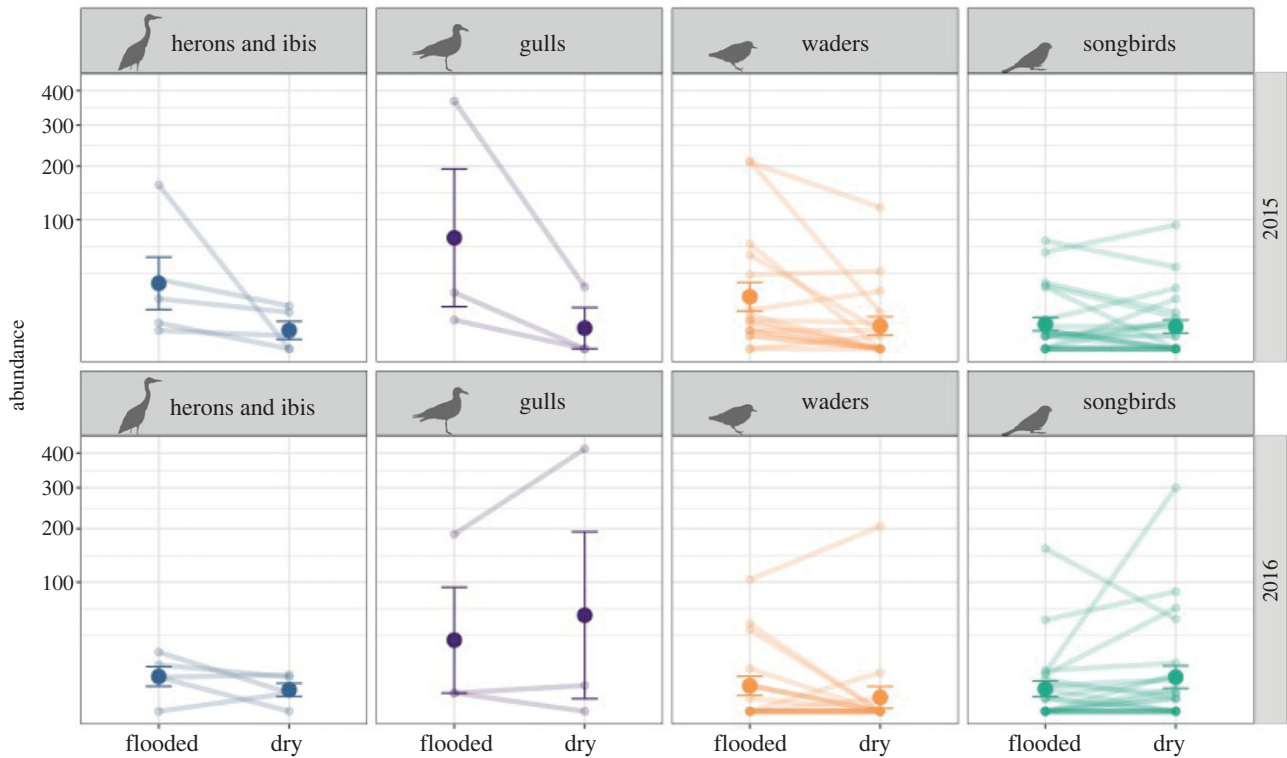
The abundance of birds in rice fields was highly influenced by water dynamics ( $\chi^2 = 6.1$ ,  $p = 0.014$ ; electronic supplementary material, table S5), yet its effect differed among groups as shown by the bird group  $\times$  flooding state interaction ( $\chi^2 = 16.9$ ,  $p = 0.004$ ). Specifically, we found a negative effect of field drying in herons and ibis ( $t = -2.4$ ,  $p = 0.018$ ) and waders ( $t = -3.7$ ,  $p < 0.001$ ), while no effect was detected for gulls ( $t = -1.2$ ,  $p = 0.228$ ), songbirds ( $t = 1.2$ ,  $p = 0.238$ ), raptors ( $t = -0.7$ ,  $p = 0.493$ ) and the 'others' functional group ( $t = 0.09$ ,  $p = 0.92$ ) (figure 4). The pattern was consistent for both years as shown by the non-significant interaction between flooding state and sampling year ( $\chi^2 = 3.0$ ,  $p = 0.081$ ). Finally, overall biomass of avian communities was also influenced by the flooding state of fields ( $\chi^2 = 16.9$ ,  $p < 0.001$ ; electronic

supplementary material table S6), showing higher values for flooded than for dry fields. Despite this trend tended to be consistent between years (i.e. higher avian biomass in flooded than in dry fields), it was only statistically significant for the first year, as shown by the significant interaction between flooding state and year ( $\chi^2 = 13.5$ ,  $p < 0.001$ ) (electronic supplementary material, figure S4).

## 4. Discussion

Direct environmental consequences of biological invasions have been largely documented [1–3], yet the collateral effects of invasive species—via changes in AEPs—on important environmental issues such as climate change and biodiversity conservation remain unexplored. Here we show that large-scale changes in the dynamics of field flooding promoted to control the invasive apple snail in the Ebro Delta entails contrasting outcomes in terms of climate change mitigation and conservation of bird diversity. Specifically, field drying during the post-harvest rice season largely slow down methane emission rates across the post-harvest season, thus reducing the contribution of rice farming to climate change. On the other hand, we found a markedly reduction of waterbird diversity and abundance in dry fields when compared to flooded fields, suggesting that post-invasion policies might hinder conservation of biodiversity in the studied region.

Rice farming contributes 48% of cropland greenhouse gas emissions globally [48], mainly because of its unique flooding system. In the studied rice-growing region, two-thirds of



**Figure 4.** Abundance of the different functional groups of birds in flooded and dry rice fields. Semi-transparent points and lines indicate the accumulated abundance at the end of the post-harvest season for each bird species of different functional groups. Solid points and error bars indicate averaged accumulated abundance ( $\pm$  s.e.) across species. Note that raptors and the 'others' functional groups were not included as they were very few abundant. The y-axis is square root scaled. (Online version in colour.)

methane emissions occur during the flooded post-harvest season [12], yet our results show that field drying drastically reduced mean emission rates by 82% and 51% in 2015 and 2016, respectively, (figure 2). Alternate wetting and dryings [49] has been already documented as an efficient strategy to reduce methane emission during the growing season as aerobic conditions inhibit methanogenic archaea activity [50]. Our results suggest that implementing field drying in the fallow season can be an efficient strategy to counteract the contribution of rice farming to methane emissions which is aligned with Belenguer-Manzanedo *et al.* [31]. Despite the significant effect of flooding regime on methane emissions, it was remarkable the presence of zero emissions under flooded conditions contrasting with some emissions detected in dry fields (figure 2). The cluster of zero methane emissions under flooded conditions mainly corresponds to the late autumn or winter period [11,31], when soil temperature in rice fields is below the optimal range (15–30°C) for methanogenesis [51]. This is coherent with the strong positive effect of temperature on methane emission rates found in our study, which is explained by an increased activity of methanogenic microbial activity during warm days [11,52,53]. On the other hand, methane emissions in dry fields can be explained by patches of saturated soil during the drying of the fields, where despite the absence of a layer of water, soil moisture preserves anaerobic conditions and thus methanogenesis. As previously reported in Martínez-Eixarch *et al.* [11] our results indicate also a negative effect of the water layer height on methane emissions rates, most likely resulting from a process of methanotrophy in the water column [11]. This suggests that maintaining a high layer of water might help in reducing the emissions of methane associated to rice fields that remain flooded during the post-harvest season.

The positive effect of AEP-mediated field drying on climate change mitigation markedly contrasted with the negative outcome in terms of waterbird conservation. Feeding habitat provided by flooded rice agroecosystems during the post-harvest season have been documented to play a crucial role in attracting a broad range of migratory and resident waterbird species during the non-breeding season [18] (but see [19]). Our results show that flooded fields attracted more waterbird species (75% in 2015 and 57% in 2016) than dry fields, probably as a consequence of higher availability of feeding resources [54]. Similarly, species evenness (i.e. Hill evenness) was also higher in flooded fields, indicating a higher number of common abundant species than in dry fields (figure 3). Reduced diversity (i.e. richness and evenness) resulted not only from the loss of low tolerant species to field drying, but also from a consistent decrease of abundance of almost all individual species of herons and ibis, gulls and waders (figure 4). Reduced diversity of waterbird communities in dry fields was not compensated by an increment of non-waterbird species, as differences between flooded and dry fields remained quite similar when we included songbirds, raptors and other low-abundant species in our analysis (figure 3). In fact, contrarily to the observed pattern for waterbirds, the abundance of non-waterbird species was quite similar in both dry and flooded fields (figure 4). Reduced diversity was neither compensated by an increment of biomass of drought tolerant species, but instead overall biomass was also reduced in dry fields (electronic supplementary material, figure S4).

By analysing the long-term International Waterbird Censuses (IWC) database, Pernollet *et al.* [18] also showed a positive effect of the post-harvest flooding AEPs at increasing the attractiveness of European rice fields for wintering



waterbirds. A recent re-analysis of this database for the Ebro Delta has reopened the debate, as they did not find evidence on a positive effect of post-harvest flooding on waterbird population trends neither a negative effect after its cessation [19]. An explanation for this result could be that even a small amount of remaining flooded rice fields plus the natural wetlands are enough to buffer the temporal loss of habitat, at least for some waterbird species. Other large-scale factors such as climate change [55,56] or habitat disturbances at the breeding grounds [57,58] may be also shaping waterbird communities beyond the Ebro Delta. However, another potential explanation might rely in the fact that the above-mentioned study [19] did not consider interannual variation in the proportion of flooded surface at the regional scale, which can largely vary across years (figure 1). In addition, they included in the analysis several species that use rice fields infrequently (e.g. pied avocet, Kentish plover, etc.), whereas they did not include other species that are intimately linked to rice fields (e.g. glossy ibis, European golden plover, etc.) [22]. Therefore, ignoring interannual variation in flooding patterns and potential differences among species with contrasting dependence on rice fields could be masking the negative effect of the cessation of the post-harvest flooding on overall waterbird population trends. Our results may be even more relevant from a waterbird conservation perspective if we note that the reduction of flooded habitats is also happening in other important wintering areas in southern Europe, such as the Doñana National Park [59]. Finally, the reduction of flooded surface is expected to affect to different groups of organisms that are intimately linked to the water layer of rice fields (from microorganisms to vertebrates), especially those with low tolerance to drying periods [60] and/or low dispersal capacity [61] (e.g. tadpoles, larval stages of macroinvertebrates, fishes or crayfishes).

Globalization of human activities is entailing a continuous flux of invasive species worldwide, promoting changes in AEP to buffer the economic and environmental impacts in the invaded regions [62]. Here we show that these changes in AEPs may shift the contribution of the invaded agrosystems to other global environmental issues such as climate change and biodiversity conservation. Our results highlight the need for gathering basic information on the ecology of worst invasive species (e.g. colonization patterns, reproductive biology) to better anticipate their response to habitat management and the effectiveness of large-scale management practices to control or eradicate their populations. In the Ebro Delta, after a decade

of post-harvest field drying, the apple snail has colonized the whole northern side of the delta and is colonizing the southern side in a scattered way [30], questioning the effectiveness of this practice to control the snail populations. Alternative practices such as washing fields with marine water or the use of molluscicides have also been used to control snail populations, however they are not exempt from problems due to the collateral effects they may entail on crop yield (soil salinization) or in communities of non-target organisms, respectively [30].

Our study emphasizes the importance to account for potential collateral effects during the policy-making process to design efficient agricultural management plans that enable to minimize undesirable agri-environmental outcomes. For example, regional plans against invasive species should incorporate the landscape scale and consider to sectorize and/or fallow the application of agricultural practices (e.g. post-harvest drying of rice fields) according to different criteria such as pressure of invasive species on agriculture, conservation importance for birds or mitigation potential of greenhouse gas emissions. We acknowledge this is a more knowledge- and resource-demanding strategy than homogenizing and simplifying management plans, yet it is also expected to better reduce potential trade-offs among different agri-environmental issues. This especially important in complex agroecosystems such as the rice systems of the Ebro Delta, which are especially vulnerable to invasive species, while playing an important role as food producers, greenhouse gas sources and biodiversity hotspots.

**Data accessibility.** The electronic supplementary material, containing datasets generated and analysed during the current study, is available in the Figshare repository [63].

**Authors' contributions.** N.P.-M.: conceptualization, data curation, formal analysis, investigation, methodology, visualization, writing—original draft, writing—review and editing; C.A.: conceptualization, investigation, writing—review and editing; A.B.: investigation, writing—review and editing; M.C.-F.: writing—review and editing; L.A.G.: writing—review and editing; J.P.G.-V.: formal analysis, writing—review and editing; S.R.: investigation, writing—review and editing; M.M.-E.: conceptualization, data curation, funding acquisition, investigation, methodology, project administration, writing—review and editing.

All authors gave final approval for publication and agreed to be held accountable for the work performed therein.

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## References

1. Simberloff D *et al.* 2013 Impacts of biological invasions: what's what and the way forward. *Trends Ecol. Evol.* **28**, 58–66. (doi:10.1016/j.tree.2012.07.013)
2. Sala OE *et al.* 2000 Global biodiversity scenarios for the year 2100. *Science* **287**, 1770–1774. (doi:10.1126/science.287.5459.1770)
3. Mooney HA, Hobbs RJ. 2000 *Invasive species in a changing world*. Washington, DC: Island Press.
4. Haubrock PJ *et al.* 2021 Economic costs of invasive alien species across Europe. *NeoBiota* **67**, 153–190. (doi:10.3897/neobiota.67.58196)
5. Simberloff D. 2013 *Invasive species: what everyone needs to know*. Oxford, UK: Oxford University Press.
6. Zavaleta ES, Hobbs RJ, Mooney HA. 2001 Viewing invasive species removal in a whole-ecosystem context. *Trends Ecol. Evol.* **16**, 454–459. (doi:10.1016/S0169-5347(01)02194-2)
7. Peltzer DA, Allen RB, Lovett GM, Whitehead D, Wardle DA. 2010 Effects of biological invasions on forest carbon sequestration. *Glob. Chang. Biol.* **16**, 732–746. (doi:10.1111/j.1365-2486.2009.02038.x)
8. Greenland DJ. 1997 *The sustainability of rice farming*. Wallingford, UK: CAB International.
9. FAO. 2021 FAOStat database. See <https://www.fao.org/faostat>.
10. Islam SF, van Groenigen JW, Jensen LS, Sander BO, de Neergaard A. 2018 The effective mitigation of greenhouse gas emissions from rice paddies without compromising yield by early-season drainage. *Sci. Total Environ.* **612**, 1329–1339. (doi:10.1016/j.scitotenv.2017.09.022)
11. Martínez-Eixarch M *et al.* 2018 Neglecting the fallow season can significantly underestimate annual methane emissions in Mediterranean rice



- fields. *PLoS ONE* **13**, e0198081. (doi:10.1371/journal.pone.0198081)
12. Martínez-Eixarch M, Alcaraz C, Viñas M, Noguero J, Aranda X, Prenafeta-Boldú FX, Català-Fornier M, Fennessy MS, Ibáñez C. 2021 The main drivers of methane emissions differ in the growing and flooded fallow seasons in Mediterranean rice fields. *Plant Soil* **460**, 211–227. (doi:10.1007/s11104-020-04809-5)
  13. Amano T, Kusumoto Y, Tokuoka Y, Yamada S, Kim EY, Yamamoto S. 2008 Spatial and temporal variations in the use of rice-paddy dominated landscapes by birds in Japan. *Biol. Conserv.* **141**, 1704–1716. (doi:10.1016/j.biocon.2008.04.012)
  14. King S, Elphick CS, Guadagnin D, Taft O, Amano T. 2010 Effects of landscape features on waterbird use of rice fields. *Waterbirds* **33**, 151–159. (doi:10.1675/063.033.s111)
  15. Stafford JD, Kaminski RM, Reinecke KJ. 2010 Avian foods, foraging and habitat conservation in world rice fields. *Waterbirds* **33**, 133–150. (doi:10.1675/063.033.s110)
  16. Ibáñez C, Curcó A, Riera X, Ripoll I, Sánchez C. 2010 Influence on birds of rice field management practices during the growing season: a review and an experiment. *Waterbirds* **33**, 167–180. (doi:10.1675/063.033.s113)
  17. Elphick CS, Oring LW. 2003 Conservation implications of flooding rice fields on winter waterbird communities. *Agric. Ecosyst. Environ.* **94**, 17–29. (doi:10.1016/S0167-8809(02)00022-1)
  18. Pernollet CA *et al.* 2015 A comparison of wintering duck numbers among European rice production areas with contrasting flooding regimes. *Biol. Conserv.* **186**, 214–224. (doi:10.1016/j.biocon.2015.03.019)
  19. Bernardo-Madrid R, Vera P, Gallardo B, Vilà M. 2022 Stopping winter flooding of rice fields to control invasive snails has no effect on waterbird abundance at the landscape scale. *Front. Ecol. Evol.* **9**, 1–12. (doi:10.3389/fevo.2021.688325)
  20. Finlayson CM, Milton GR, Prentice RC, Davidson NC. 2018 *The wetland book*. Berlin, Germany: Springer Nature.
  21. Genua-Olmedo A, Temmerman S, Ibáñez C, Alcaraz C. 2022 Evaluating adaptation options to sea level rise and benefits to agriculture: the Ebro Delta showcase. *Sci. Total Environ.* **806**, 150624. (doi:10.1016/j.scitotenv.2021.150624)
  22. Curcó A, Bigas D. 2018 *Cens hivernal d'ocells aquàtics al delta de l'Ebre, 2015*. Deltebre, Spain: Parc Natural del Delta de l'Ebre.
  23. Tajiri H, Ohkawara K. 2013 The effects of flooding and plowing on foraging site selection by wintering dabbling ducks in rice fields. *Ornithol. Sci.* **12**, 127–136. (doi:10.2326/osj.12.127)
  24. Toral GM, Figuerola J. 2010 Unraveling the importance of rice fields for waterbird populations in Europe. *Biodivers. Conserv.* **19**, 3459–3469. (doi:10.1007/s10531-010-9907-9)
  25. Eadie JM, Elphick CS, Reinecke KJ, Miller MR. 2008 *Wildlife values of north American ricelands*. Stuttgart, AR: The Rice Foundation.
  26. Kurechi M. 2007 Restoring rice paddy wetland environments and the local sustainable society: project for achieving co-existence of rice paddy agriculture with waterbirds at Kabukuri-numa, Miyagi Prefecture, Japan. *Glob. Environ. Res.* **12**, 141–152.
  27. Cowie RH, Hayes KA. 2012 Apple snails. In *A handbook of global freshwater invasive species* (ed. RA Francis), pp. 216–232. London, UK: Routledge.
  28. López MA, Altaba CR, Andree KB, López V. 2010 First invasion of the apple snail *Pomacea insularum* in Europe. *Tentacle* **18**, 26–28.
  29. Naylor R. 1996 Invasions in agriculture: assessing the cost of the golden apple snail in Asia. *Ambio* **25**, 443–448.
  30. Gencat. 2018 *Caracol Manzana Pomacea maculata memoria de actuaciones período 2015–2018*. Barcelona, Spain: Generalitat de Catalunya.
  31. Belenguer-Manzanedo M, Alcaraz C, Camacho A, Ibáñez C, Català-Fornier M, Martínez-Eixarch M. 2022 Effect of post-harvest practices on greenhouse gas emissions in rice paddies: flooding regime and straw management. *Plant Soil*, 77–98. (doi:10.1007/s11104-021-05234-y)
  32. Altor AE, Mitsch WJ. 2008 Pulsing hydrology, methane emissions and carbon dioxide fluxes in created marshes: A 2-year ecosystem study. *Wetlands* **28**, 423–438. (doi:10.1672/07-98.1)
  33. Hatala JA, Detto M, Baldocchi DD. 2012 Gross ecosystem photosynthesis causes a diurnal pattern in methane emission from rice. *Geophys. Res. Lett.* **39**, 1–5. (doi:10.1029/2012GL051303)
  34. Billerman SM, Keeney BK, Rodewald PG, Schulenberg TS. 2022 *Birds of the world*. Ithaca, NY: Cornell Laboratory of Ornithology.
  35. Blondel J. 2003 Guilds or functional groups: does it matter? *Oikos* **100**, 223–231. (doi:10.1034/j.1600-0706.2003.12152.x)
  36. Roshier DA, Robertson AI, Kingsford RT. 2002 Responses of waterbirds to flooding in an arid region of Australia and implications for conservation. *Biol. Conserv.* **106**, 399–411. (doi:10.1016/S0006-3207(01)00268-3)
  37. Almeida BA, Sebastián-González E, dos Anjos L, Green AJ. 2020 Comparing the diversity and composition of waterbird functional traits between natural, restored, and artificial wetlands. *Freshw. Biol.* **65**, 2196–2210. (doi:10.1111/fwb.13618)
  38. Foster SD, Bravington M V. 2013 A Poisson-Gamma model for analysis of ecological non-negative continuous data. *Environ. Ecol. Stat.* **20**, 533–552. (doi:10.1007/s10651-012-0233-0)
  39. Chao A, Chiu C-H, Jost L. 2014 Unifying species diversity, phylogenetic diversity, functional diversity, and related similarity and differentiation measures through Hill numbers. *Annu. Rev. Ecol. Syst.* **45**, 297–324. (doi:10.1146/annurev-ecolsys-120213-091540)
  40. Hill MO. 1973 Diversity and evenness: a unifying notation and its consequences. *Ecology* **54**, 427–432. (doi:10.2307/1934352)
  41. Roswell M, Dushoff J, Winfree R. 2021 A conceptual guide to measuring species diversity. *Oikos* **130**, 321–338. (doi:10.1111/oik.07202)
  42. Tobias JA *et al.* 2022 AVONET: morphological, ecological and geographical data for all birds. *Ecol. Lett.* **25**, 581–597. (doi:10.1111/ele.13898)
  43. R Core Team. 2022 *R: a language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing.
  44. Brooks ME *et al.* 2017 glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. *R J.* **9**, 378. (doi:10.32614/RJ-2017-066)
  45. Hartig F. 2020 DHARMA: residual diagnostics for hierarchical (multi-level/mixed) regression models. See <https://cran.r-project.org/web/packages/DHARMA/vignettes/DHARMA.html>.
  46. Lenth R. 2020 emmeans: estimated marginal means, aka least-squares means. See <https://mran.microsoft.com/web/packages/emmeans/emmeans.pdf>.
  47. Wickham H *et al.* 2019 Welcome to the tidyverse. *J. Open Source Softw.* **4**, 1686. (doi:10.21105/joss.01686)
  48. Carlson KM *et al.* 2017 Greenhouse gas emissions intensity of global croplands. *Nat. Clim. Chang.* **7**, 63–68. (doi:10.1038/nclimate3158)
  49. Bouman BAM, Lampayan RM, Tuong TP. 2007 *Water management in irrigated rice: coping with water scarcity*. Los Baños, Philippines: International Rice Research Institute.
  50. Martínez-Eixarch M *et al.* 2021 Multiple environmental benefits of alternate wetting and drying irrigation system with limited yield impact on European rice cultivation: the Ebre Delta case. *Agric. Water Manag.* **258**, 107164. (doi:10.1016/j.agwat.2021.107164)
  51. Fey A, Conrad R. 2003 Effect of temperature on the rate limiting step in the methanogenic degradation pathway in rice field soil. *Soil Biol. Biochem.* **35**, 1–8. (doi:10.1016/S0038-0717(02)00175-X)
  52. Zhang ZS, Guo LJ, Liu TQ, Li CF, Cao CG. 2015 Effects of tillage practices and straw returning methods on greenhouse gas emissions and net ecosystem economic budget in rice-wheat cropping systems in central China. *Atmos. Environ.* **122**, 636–644. (doi:10.1016/j.atmosenv.2015.09.065)
  53. Devèvre OC, Horwath WR. 2000 Decomposition of rice straw and microbial carbon use efficiency under different soil temperatures and moistures. *Soil Biol. Biochem.* **32**, 1773–1785. (doi:10.1016/S0038-0717(00)00096-1)
  54. Gonzalez-Solis J, Bernadi X, Ruiz X. 1996 Seasonal variation of waterbird prey in the Ebro Delta rice fields. *Colon. Waterbirds* **19**, 135–142. (doi:10.2307/1521957)
  55. Lehikoinen A *et al.* 2013 Rapid climate driven shifts in wintering distributions of three common waterbird species. *Glob. Chang. Biol.* **19**, 2071–2081. (doi:10.1111/gcb.12200)
  56. Knudsen E *et al.* 2011 Challenging claims in the study of migratory birds and climate change. *Biol. Rev.* **86**, 928–946. (doi:10.1111/j.1469-185X.2011.00179.x)
  57. Sanderson FJ, Donald PF, Pain DJ, Burfield IJ, van Bommel FPJ. 2006 Long-term population declines in Afro-Palaearctic migrant birds. *Biol. Conserv.* **131**, 93–105. (doi:10.1016/j.biocon.2006.02.008)

58. Gauthier G, Giroux JF, Reed A, Béchet A, Bélanger L. 2005 Interactions between land use, habitat use, and population increase in greater snow geese: what are the consequences for natural wetlands? *Glob. Chang. Biol.* **11**, 856–868. (doi:10.1111/j.1365-2486.2005.00944.x)
59. Camacho C, Negro JJ, Elmberg J, Fox AD, Nagy S, Pain DJ, Green AJ. 2022 Groundwater extraction poses extreme threat to Doñana World Heritage Site. *Nat. Ecol. Evol.* **6**, 654–655. (doi:10.1038/s41559-022-01763-6)
60. Bonada N, Rieradevall M, Prat N. 2007 Macroinvertebrate community structure and biological traits related to flow permanence in a Mediterranean river network. *Hydrobiologia* **589**, 91–106. (doi:10.1007/s10750-007-0723-5)
61. Cañedo-Argüelles M, Boersma KS, Bogan MT, Olden JD, Phillipsen I, Schriever TA, Lytle DA. 2015 Dispersal strength determines meta-community structure in a dendritic riverine network. *J. Biogeogr.* **42**, 778–790. (doi:10.1111/jbi.12457)
62. Hulme PE. 2009 Trade, transport and trouble: managing invasive species pathways in an era of globalization. *J. Appl. Ecol.* **46**, 10–18. (doi:10.1111/j.1365-2664.2008.01600.x)
63. Pérez-Méndez N, Alcaraz C, Bertolero A, Catala-Forner M, Garibaldi LA, González-Varo JP, Rivaes S, Martínez-Eixarch M. 2022 Data from: Agricultural policies against invasive species generate contrasting outcomes for climate change mitigation and biodiversity conservation. Figshare. (doi:10.6084/m9.figshare.c.6238502)