Contribution of Doñana Wetlands to Carbon Sequestration

Edward P. Morris1*, Susana Flecha3, Jordi Figuerola2, Eduardo Costas3, Gabriel Navarro1, Javier Ruiz1, Pablo Rodriguez4, Emma Huertas1

1 Department of Ecology and Coastal Management, Instituto de Ciencias Marinas de Andalucía (ICMAN-CSIC), Consejo Superior de Investigaciones Científicas, Puerto Real, Spain, 2 Estación Biológica de Doñana (CSIC), Sevilla, Spain, 3 Universidad Complutense de Madrid, Madrid, Spain, 4 Universidad de Murcia, Murcia, Spain

Abstract

Inland and transitional aquatic systems play an important role in global carbon (C) cycling. Yet, the C dynamics of wetlands and floodplains are poorly defined and field data is scarce. Air-water CO₂ fluxes in the wetlands of Doñana Natural Area (SW Spain) were examined by measuring alkalinity, pH and other physicochemical parameters in a range of water bodies during 2010–2011. Areal fluxes were calculated and, using remote sensing, an estimate of the contribution of aquatic habitats to gaseous CO₂ transport was derived. Semi-permanent ponds adjacent to the large Guadalquivir estuary acted as mild sinks, whilst temporal wetlands were strong sources of CO₂ (−0.8 and 36.3 mmolCO₂ m⁻² d⁻¹). Fluxes in semi-permanent streams and ponds changed seasonally; acting as sources in spring-winter and mild sinks in autumn (16.7 and −1.2 mmolCO₂ m⁻² d⁻¹). Overall, Donana’s water bodies were a net annual source of CO₂ (5.2 molC m⁻² y⁻¹). Up-scaling clarified the overwhelming contribution of seasonal flooding and allochthonous organic matter inputs in determining regional air-water gaseous CO₂ transport (13.1 GgC y⁻¹). Nevertheless, this estimate is about 6 times < local marsh net primary production, suggesting the system acts as an annual net CO₂ sink. Initial indications suggest longer hydroperiods may favour autochthonous C capture by phytoplankton. Direct anthropogenic impacts have reduced the hydroperiod in Doñana and this maybe exacerbated by climate change (less rainfall and more evaporation), suggesting potential for the modification of C sequestration.

Introduction

Inland and transitional (i.e., neither fully open coastal nor enclosed or flowing freshwater) aquatic systems are hotspots for biogeochemical transformations, and in particular play an important, previously under appreciated, role in global carbon (C) cycling [1–4]. Current estimates suggest that lakes, reservoirs, rivers, estuaries, ponds, streams and wetlands make a substantial contribution to global air-water C fluxes, long-term C accumulation in sediments and may play a role in regulating the climate [5–7].

Inland waters are estimated to emit 1.4 PgC y⁻¹ to the atmosphere, are responsible for the burial of 0.6 PgC y⁻¹ in sediments and transport 0.9 PgC y⁻¹ to the sea [5,6]. Hence, the total terrestrial organic carbon (OC) imported to inland waters is in the order of 2.9 Pg y⁻¹, which is comparable in magnitude to the terrestrial C sink for anthropogenic emissions [1] and terrestrial net ecosystem production [3]. Lakes [8–10], reservoirs [11], rivers [12], estuaries [13,14] and streams [15] make up the majority of these estimates, with each aquatic component contributing between 10 to 30% of the present total inland-transitional air-water CO₂ flux [5,6]. Nevertheless, whilst it is clear that inland-transitional waters are a vital component of global C cycling, the magnitude of fluxes are still relatively poorly known in terms of the global estimated area of water bodies and the diversity of aquatic systems with good data coverage [2,16,17]. The potential role of very small streams, lakes and ponds, as well as wetlands and anthropogenic water bodies (such as rice paddies, farm ponds, reservoirs and drainage-irrigation networks) is still to be adequately understood [1,2,6]. Indeed recent studies suggest strong scale-dependence of fluxes [4,18], reinforcing the need for detailed data coverage.

Globally smaller water bodies probably account for the majority of inland waters [lakes <1 km² may account for >50% of the total area of all lakes [16]] and tend to have very high rates of areal CO₂ fluxes and OC burial. This is particularly the case for small agriculturally eutrophic impoundments’, which alone have recently been estimated to bury more OC each year than the oceans [2,19]. The global area of wetlands (i.e., land surface that regularly has inundated, or saturated, conditions [20]) is about 5 times that of lakes, rivers and streams [2,21,22] and these diverse aquatic ecosystems also tend to have high rates of OC burial and
large fluxes of green house gases (GHG, such as CO₂, CH₄ and N₂O) [23–25]. Most wetlands are only temporarily flooded and contain a large (and variable) abundance of plant biomass in contact with the atmosphere, making the distinction between terrestrial and aquatic inputs of CO₂ and estimation of global budgets rather difficult [5].

Getting a good grip on how these shallow aquatic systems function and their potential role in carbon cycling is particularly important considering global wetland habitat losses and the increasingly apparent effects of climate change [21,26]. Indeed, regional differences, combined with the local effects of eutrophication and landscape changes, rather complicate predictions about how inland-transitional waters will respond to the future climate (although for some general indications see [6]). For the Mediterranean climatic region, higher minimum temperatures, more extreme high temperature events in summer and less precipitation is predicted [27]. This suggests competition for water between natural wetlands and anthropogenic activities (such as agriculture and tourism) will be exacerbated. Furthermore, as near a large rivers are already dammed and are under strong pressure from numerous stakeholders, ensuring the maintenance of environmental flows into Spanish wetlands may become increasingly difficult in the future [28].

As Mediterranean wetlands are characterised by a dry phase in summer to autumn, reduced runoff is likely to result in a shorter hydroperiod, as well as lower water levels and increased retention times in permanent water bodies. In consequence, smaller water bodies increasingly maintained by groundwater flows, with higher conductivity and recalitrant DOC maybe expected [6]. These will receive similar or even larger inputs of terrestrial organic matter (OM) and nutrients from anthropogenic activities, suggesting potential for increasing eutrophication. Higher temperatures will also tend to intensify the symptoms of eutrophication, on the one hand, potentially increasing autochthonous production, CO₂ influxes [29] and OC burial [6]. On the other hand, higher rates of respiration [30] combined with alterations to community structure and increasing anoxia, may actually enhance the release of GHGs to the atmosphere [6,31]. OM priming (or bacterial priming), were labile OC enhances the mineralization rates of more refractory OM [32–34] may also be an important feedback mechanism that potentially results in reduced C storage efficiency in wetlands. This maybe particularly relevant in the Mediterranean region where, because of the predictable dry phase, inputs of terrestrial and autochthonous OM have a strong temporal separation.

Climate change predictions in the Mediterranean region essentially represent a strengthening of the present seasonal trends, hence observing the seasonality of C cycling within wetlands may allow inferences to be derived about the potential effects of the future climate. Furthermore, examining water bodies with different degrees of anthropogenic influence may help understand potential interactions with eutrophication. Here we examine the spatio-temporal variation in air-water CO₂ fluxes (F_CO₂) within aquatic habitats of Doñana Natural Area. By collecting a suite of physio-chemical parameters from a representative range of different water bodies, we provide insights into the mechanisms that control F_CO₂ within these semi-permanent ponds, streams and temporary wetlands. We provide the first tentative estimates of air-water CO₂ transport for the region and valuable indications about the potential role of Mediterranean, transitional wetland ecosystems in regional carbon cycling.

**Methods**

**Study area**

Doñana is situated on the Atlantic coast of southwestern Spain (Fig. 1, Long: −6.373, Lat: 36.932, Datum: WSG84). Covering an area of 3560 km² the region includes a rich variety of landforms and vegetation types representative of Mediterranean lowlands [35]. The climate is Mediterranean sub-humid with well defined seasonality; mild (average daily temperature of 9.3°C) and wet winters and dry and hot summers (25.8°C). Mean annual precipitation is about 550 mm with rainfall mostly occurring between October and March (80%) and almost absent between June and August.

The wetlands within the region have the highest degree of environmental protection in Spain (National Park status) and are one of the most emblematic protected areas in Europe with a rich biotic diversity and unique importance for wildfowl in Western Europe. Doñana National Park (ca. 54.000 ha) was designated a Biosphere Reserve by UNESCO in 1980 and a Wetland Site of International Importance by the Ramsar Convention in 1982 (entering into the Montreux Record of Ramsar sites under threat in 1990), and was declared a World Heritage Site by UNESCO in 1994 [36]. Much of the surrounding region is designated Natural (i.e., Regional) Park status with the aim of buffering human impacts on the National Park (together they are now known as the Doñana Natural Area). This status allows a larger array of traditional activities including forestry, cattle ranching, hunting, aquaculture and agriculture [35].

A diversity of aquatic systems can be found within the region including the large Guadalquivir estuary, smaller rivers/streams, such as the Guadiamar, La Rocina and El Partido, semi-permanent and temporary ponds and marshes (Fig. 1). Human interventions throughout the 20th century have drastically modified the hydrological functioning of the region, in particular reducing the total input and distribution of water within the large temporal wetlands of the National Park [28]. Water inputs from the Guadalquivir and Guadiamar rivers are very low, although restoration programs have been implemented, the main inputs of water are rainfall, the smaller streams situated in the west and ground-water.

Outside of the National Park adjacent to the Guadalquivir estuary is a privately owned wetland system with Natural Park status (Veta la Palma, ca. 11.300 ha). The area has a long history of traditional human activities and is roughly divided into 3200 ha of permanent ponds used for extensive and semi-extensive aquaculture, 3500 ha dedicated to agriculture (of which 400 ha is seasonally inundated for rice production) and 4600 ha of preserved natural marshland [37].

**Meteorological data**

Rainfall (mm), air temperature (°C), atmospheric partial pressure of CO₂ (pCO₂air, µatm) and wind speed at height (u₂, m s⁻¹) measured hourly were provided by ICTS (http://ictsb.escal.csic.es) from a number of stations situated throughout the park (Fig. 1).

**Sampling design**

Samples were collected from 11 water bodies (n = 6 to 12 per water body), selected to represent the diversity of aquatic systems within the region (Fig. 1), as part of an on going monitoring program between 2009 and 2011. Permission for field sampling was given by the director (Juan Carlos Rubio Garcia) of the Doñana Natural Area. Field studies did not involve endangered or protected species. Sites from Veta la Palma (M1:M4) are outside of
the National Park boundaries and close to the Guadalquivir estuary. M1 (Cuquero Grande) and M2 (Cuquero Chico) are unmanaged temporal ponds, whereas M3 (Gaveta 4) and M4 (Gaveta 1) are used for aquaculture, with water pumped in from the estuary, and hence influenced by saline waters. In 2010–2011 water levels were maintained all season in M3, but not M4. M7 is situated at the entrance of La Rocina stream into the wetlands. M8 (Laguna Dulce) is one of the larger semi-permanent oligohaline ponds situated within a large sand dune system (the Dune ponds region). M5 (La Fao) and M6 (Marisma Galega) are semi-permanent ponds connected to the marshes and the other sites (M9:M11) are situated in temporal marshes, and thus only

Figure 1. Maps showing Spain within Europe (A) and the autonomous region of Andalucı´ a within Spain (B). Doñana Natural Area is highlighted by a black square. The Guadalquivir River is highlighted by a solid blue line. Datum: ETRS89, Projection: ETRS-LAEA. Detailed maps of the Doñana region: National and natural park limits (dark green lines), urban areas (grey polygon), positions of ICTS meteorological stations (black circles) and wetland regions (coloured polygons) (C), and position of sampling sites (black text, local names given as legend), artificial structures restricting water exchange (red line) and remote sensing-derived water coverage during the flood season 2010–2011 (blue shading, darker blue represents more permanent water bodies) (D). Datum: WSG84, Projection: UTM29N. doi:10.1371/journal.pone.0071456.g001
sampled when sufficient water was present (a minimum water depth of 0.1 m). Sampling was carried out on foot from the edge of the water body with the exact position (recorded using GPS) changing slightly depending on the distribution of water (i.e., the water level).

On each sampling occasion (approximately every 30 d) in situ water conductivity, temperature and pH were measured and water samples were collected for laboratory analysis of suspended particulate matter (SPM), chlorophyll (Chl), nutrients, dissolved organic matter, oxygen, pH and total alkalinity (AT). Sampling occurred during daylight hours and took 2 days, with the processing of water samples carried out in a field laboratory each evening.

Analytical techniques

In situ water measurements of salinity, temperature and pH (National Bureau of Standards, NBS, scale) were collected with a multi-probe (YSI-6920V2, YSI Incorporated, Yellow Springs, Ohio, USA) at approximately mid-depth. Determination of SPM as well as particulate organic matter (POM) and particulate inorganic matter (PIM) was carried out by filtering a known volume of water (pre-combusted 450 °C Whatman GF/F glass fiber, diameter 47 mm). Filters were dried at 60 °C for 48 h and weighed to derive SPM [g L\(^{-1}\)], further combusted at 450 °C for 5 h and weighed to derive PIM and POM by difference.

Chlorophyll analysis was conducted by filtering known volumes of water (Whatman GF/F glass fiber, 0.7 m pore size), extracting in 90% acetone overnight in the dark, and measuring chlorophyll a concentrations using standard fluorometric methods following JGOFS protocols with a Turner Design Model-10. The fluorometer was calibrated using a pure chlorophyll a standard from the cyanobacterium *Anacystis nidulans* (Sigma Chemical Company).

For inorganic nutrient analysis two 5 mL samples of filtered water (Whatman GF/F borosilicate glass fiber, 0.7 m pore size) were stored at −20 °C until analysis (∼4 weeks). Concentrations of NH\(_4\)\(^+\), NO\(_3\)\(^-\), NO\(_2\)\(^-\), PO\(_4\)\(^{3-}\) and Si(OH)\(_4\)\(^-\) were derived following the techniques described by [30] using a SkalarSan + 215 Continuous Flow Analyzer.

For the analysis of dissolved organic carbon (DOC) and total dissolved nitrogen (TDN) water samples were collected in situ within borosilicate vials (pre-acid-washed and-combusted, 450 °C). Known water volumes were filtered (pre-combusted 450 °C, Whatman GF/F borosilicate glass fiber, 0.7 μm pore size) and a 24 mL sub-sample acidified (50 μL 25% H\(_2\)PO\(_4\)) sealed and conserved at 4 °C in darkness until analysis (∼5 d). Concentrations of DOC and TDN were derived by catalytic oxidation at high temperature (720 °C) and chemiluminescence, respectively using a Shimadzu TOC-VCPH analyser.

Dissolved oxygen was determined following the Winkler method. Water was carefully collected so as to avoid headspace in glass flasks of known weight. The flasks were sealed and stored in darkness, for at least 24 h until analysis. Dissolved oxygen concentrations were derived by potentiometric determination using a Metrohm 794 Titroprocessor, with an estimated error of ±1 μmol L\(^{-1}\).

AT was measured with a Metrohm 794 Titroprocessor following the method described by [39]. Water samples were collected and stored in 500 mL borosilicate bottles treated with 100 μL of HgCl\(_2\) saturated aqueous solution until analysis. The accuracy of AT determinations was ±2 μmol L\(^{-1}\) as determined from regular measurements of 2 batches (batch 85 and 89) of certified reference material (CRM, supplied by Prof. Andrew Dickson, Scripps Institution of Oceanography, La Jolla, CA, USA). Water pH (NBS scale) measurements were carried out using a Metrohm 780 pH meter equipped with a crystal electrode combination.

Calculations

The speciation of carbon in water was calculated using CO2sys.xls [40] with the dissociation constants for C and sulphate of [41] and [42], respectively. The input parameters were the measured in situ salinity, temperature (°C), atmospheric pressure (bar), AT (μmol kg\(^{-1}\)), pH (NBS scale), dissolved inorganic phosphate (μmol kg\(^{-1}\)) and dissolved inorganic silica (μmol kg\(^{-1}\)). Where nutrient data was unavailable (site M3 in May 2010 and all sites in Feb 2011), the data set average of nutrient values (2.84 and 141.91 mol L\(^{-1}\) for PO\(_4\)\(^{-3}\) and Si(OH)\(_4\), respectively) was used in calculations.

Air-water CO\(_2\) fluxes \(F_{CO2}\) (μmol CO\(_2\) m\(^{-2}\) d\(^{-1}\) were calculated according to [43]:

\[
F_{CO2} = k_w \cdot K_w \cdot (p_{CO2w} - p_{CO2a})
\]

where, \(k_w\) (m s\(^{-1}\)) is the water–air gas transfer velocity and \(K_w\) (mol m\(^{-2}\) atm\(^{-1}\)) is the aqueous-phase solubility of CO\(_2\) [44,45]. \(k_w\) was calculated using the numerical scheme of [44]:

\[
k_w = k_{600} \cdot \left(\frac{S_{CO2}}{600}\right)^{-0.5}
\]

where, \(S_{CO2}\) is the Schmidt number at the in situ water temperature and salinity, calculated from the diffusivity of CO\(_2\), dynamic viscosity and density of water (see [45] for details) and \(k_{600}\) is \(k_w\) normalized to a \(S_{CO2}\) value of 600 (often quoted as typical of freshwater at 20°C). \(k_{600}\) was predicted from time–ensemble averaged (1 d) horizontal wind velocity at 10 m above the surface (\(u_{10}\), m s\(^{-1}\)) using the empirical relationship derived for lakes by [46].

\[
u_{10} = 2.07 + 0.215 \cdot u_{10}^{0.7}
\]

\(u_{10}\) was calculated from \(u_z\) measured at nearby meteorological stations (ICTS Doñana, Fig. 1) according to [47] and spatially–averaged to give a single value for the region. We acknowledge that the empirical relationship chosen to adjust \(k_{600}\) values in this study may not be ideal for representing wind–enhancement effects in this particular aquatic system (shallow water bodies with substantial variations in extent). To give an indication of the uncertainty this choice introduces, we also predicted \(k_{600}\) values using empirical relationships derived for small water bodies [48] (median \(k_{600}\) ~30% lower) and the global oceans [43] (~30% higher values). Daily values of water temperature, salinity, pCO\(_2\)air and pCO\(_2\)water, needed for calculation of \(S_{CO2}\) and \(F_{CO2}\), were estimated by linear interpolation of the measured monthly values. Annual areal \(F_{CO2}\) was estimated by summing daily values.

A first approximation of daily air-water CO\(_2\) transport (MMol CO\(_2\) d\(^{-1}\)) for Doñana National Park and Veta la Palma was calculated by averaging the areal \(F_{CO2}\) values of all sites within each wetland region (see Fig. 1) and multiplying by the surface area of water in each region respectively. Annual air-water CO\(_2\) transport was calculated by summing over the year. Using the
empirical relationships derived for small water bodies and global oceans to adjust $k_{w0}$ values would give annual air–water CO$_2$ transport estimates 35% lower and 29% higher, respectively.

Water coverage was derived by remote sensing; using cloud-free Landsat TM (30 m$^2$ pixel size) and DEIMOS1 (22 m$^2$) scenes collected between Nov. 2009 and March 2011. Briefly, satellite images were orthorectified against high precision airborne-photos available from the Andalucian geo-spatial infrastructure (REDIAM, http://www.juntadeandalucia.es/medioambiente/site/web/rediam, Ortofoto Digital Color de Andalucia 2008–2009). Images

Figure 2. Time-series plots of air and water temperature (A), atmospheric $p$CO$_2$ (B), monthly rainfall (C) and daily mean wind velocity at a height of 10 m, $u_{10}$ (D). Light and dark shading represents seasons. doi:10.1371/journal.pone.0071456.g002
were radially calibrated and atmospherically corrected using an image-based procedure. Finally, the multi-temporal image series was normalised using major-axis regression on a number of pseudo invariant features (such as sandy beaches, man-made structures and offshore water) selected for their very low temporal variation. Pixels were classified as water if their reflectance in the near infra-red band was less than 0.2. The Water surface area within each region was estimated by counting the number of water pixels with each region were classified as water if their reflectance in the near infra-red band.

All data files required to calculate air-water CO₂ fluxes and regional transport rates are available for download from Digital.C-SIC, the Institutional Repository of the Spanish National Research Council (CSIC) [http://digital.csic.es/handle/10261/77418].

Statistics
Statistics were performed with the statistical program language R 2.15. Probability distributions of variables were examined visually and in many cases were log-normal and highly skewed. Non-parametrical Kruskal-Wallis rank sum tests (KWRS, R function; kruskal.test) and non-parametrical multiple test procedures (KWMC, Package; pgirmess, function; kruskalmc) were used to examine differences between sites [50]. Significance levels were set at p < 0.05. Principle components analysis (R package; FactoMineR, function; PCA, [51]) of transformed, log(x+1), variables, with monthly mean wind speed, total rainfall and flooding extent as a supplementary quantitative variables, was used to explore correlations. Pearson’s product-moment correlation (PPMC) was used to test for significant correlations between variables (R function; cor.test).

Results
Meteorological conditions
Time-ensemble-averaged monthly air temperature ranged between a minimum of about 10.5 and a maximum of 27.7°C in January and August, respectively (Fig. 2a). Atmospheric carbon dioxide partial pressure (pCO₂air) ranged between 369 and 398 μatm with a median value of 380 μatm. The lowest and highest values were observed in February and August 2010, respectively. Monthly total rainfall ranged from a minimum of 0 mm in the summer of 2010 to a maximum of about 200 mm in winter 2009 and late- autumn 2010 (Fig. 2b). The wetlands of Doñana had the largest extent of water coverage reaching a maximum of 224 km² in Feb. 2011., minimum values of between 1 to 3 km² were observed in late summer–autumn (Fig. 2c). Veta la Palma had the second largest water coverage and changed relatively little throughout the study period (ranging between 14 and 37 km²). Variation in water cover in the dune ponds was more seasonal (0.1 to 4 km²), with the highest cover observed in the wet season. Because of the small size of these water bodies i.e., problems reliably detecting them with the satellite sensors used, these are probably underestimations. The same issue appeared to affect estimates of water cover in La Rocina stream, resulting in estimates of <0.1 km². Time-ensemble-averaged (daily) horizontal wind velocity at 10 m above the surface (u₁₀) ranged between 1.3 and 7.2 m s⁻¹, with a median value of 2.5 m s⁻¹ (Fig. 2d).

Physical and chemical characteristics
Water temperature ranged between 13.6 and 31.7°C with a clear seasonal pattern that was similar at all sites (Fig. 2a). Median values of a range of other water physiochemical parameters are summarised in Table 1. Most water bodies could be considered as meso–to–eutrophic based on [Chl a] [52]. Sites in Veta la Palma (M1:M4) generally had higher median salinities (4) compared to La Rocina stream (M7) and the Dune ponds site (M8), however because of seasonal variations, most sites formed a group that ranged between oligo and mesohaline (KWMC, p < 0.05). Significant differences between sites were also observed for most other parameters except dissolved N, Chl a and POM (Table 1).

Principle component analysis revealed that the data could be summarised into 4 components that accounted for a cumulative percentage variance of 70%. In terms of briefly characterising the different water bodies only principle components 1 (PC1, 26.4%...
Table 2. Median (M6:M8) fell between these two extremes. The remaining sites observed in March 2010 (Fig. 4).

Water bodies of Veta La Palma (M1:M4). The remaining sites appeared to mainly represent the differences between sites; with positive values associated with higher [NO3]−, wind speed, rainfall and the extent of flooding, all winter-spring phenomena. In contrast PC2 appeared to mainly represent seasonality; positive values were associated with higher concentrations of phytoplankton variables (POM, SPM, Chlα and DOM) and higher water temperatures observed in summer, whereas negative values were associated with higher [NO3]−. Wind speed, rainfall and the extent of flooding, all winter-spring phenomena. In contrast PC2 appeared to mainly represent the differences between sites; with positive values associated with higher $pCO_2$, PO4 3−, Si(OH)4 in the water bodies of the Doñana wetlands (M5, M9, M10 and M11) and negative values associated with higher Salinity, $pO_2$ and pH in the water bodies of Veta La Palma (M1:M4). The remaining sites (M6:M8) fell between these two extremes.

Spatio-temporal variation in dissolved CO2

Dissolved carbon dioxide partial pressure ($pCO_2_{water}$) ranged between 5 and 10980 µatm and was significantly different between sites (Kruskal-Wallis rank sum test, $\chi^2_{(10)} = 28.51, p < 0.01$) (Table 2). Median $pCO_2_{water}$ was higher in the temporal wetland site M9 (6322 µatm) compared to the Veta La Palma site M3 (115 µatm) (KWMC, $p < 0.05$).

$pCO_2_{water}$ values at site M3 were below $pCO_2_{sat}$ (380 µatm) throughout the year, except for a single value of 325 µatm observed in March 2010 (Fig. 4). $pCO_2_{water}$ values 4 times higher than $pCO_2_{sat}$ were also observed at site M1 in March, however not at the other nearby water bodies, M2 and M4, where high values were observed just before and after the drying out phase. Within the National Park, seasonal $pCO_2_{water}$ dynamics in the semi-permanent water bodies, sites M5, M6, M7 and M8 were relatively similar. All sites had very low values in early spring 2010 (about 100 atm) and a maximum (10983, 3843, 3975 and 5842 µatm at sites M5, M6, M7 and M8, respectively) in late spring–early summer. Water levels in M5 were too low for sampling in September, however in sites M6, M7 and M8 $pCO_2_{water}$ reached a seasonal minimum in autumn of 42, 59 and 5 µatm, respectively. In site M8 $pCO_2$ values were so low in Sept. and Oct. that $pCO_2_{water}$ was almost completely depleted. All sites returned to a state of super-saturation with respect to atmospheric equilibrium in late winter.

In the temporal wetlands within the National Park, sites M9, M10 and M11, $pCO_2_{water}$ was generally highly oversaturated throughout the wet season (>2000 µatm). Relative minima of 294, 335, and 757 were observed in April at sites M9, M10 and M11, respectively. A second minimum of 130, and 54 was observed in June at sites M10 and M11, respectively. After reflooding of the wetlands in the winter of 2011, all sites had very high $pCO_2_{sat}$ values in the order of 3000 to 10000 µatm.

Grouping the sites into similar types of water bodies, the mesohaline sites in Veta La Palma (M1:M4) had the lowest $pCO_2_{water}$ values of the region (224 µatm) and showed relatively little seasonal variation. The temporal wetlands (M9, M10 and M11) had relatively constant and high $pCO_2_{water}$ values (2358 µatm) during the flood period. In contrast the semi-permanent oligohaline sites (M5, M6, M7 and M8) had very large seasonal variations (the most extreme values observed) that could be generally characterised as low values (85 µatm) in early spring and autumn, and high values (3758 µatm) in early summer and winter.

Water–side gas transfer velocities

Calculated gas transfer velocities ($k_w$) ranged between 0.85 and 10.17 cm h−1 with a median value of 1.49 cm h−1. Differences in $k_w$ between sites were generally minor except for site M3, where high salinities (>30) resulted in substantially higher $Sc_w$ values.

Figure 3. Biplots of water physical and chemical characteristics with scaling highlighting variable correlations (A) and mapping of individual samples (B). Rainfall, wind velocity and total flooded surface area are plotted as supplementary variables (blue arrows and text) on A. Text representing the centroids for each site and month are shown on B. Colours of sites and months represent regions (see Fig. 1) and seasons, respectively. doi:10.1371/journal.pone.0071456.g003
Higher values of $k_w$ were more frequent in spring and autumn, the periods with maximum wind velocities.

**Figure 4. Seasonal variation in $pCO_{2\text{water}}$ at each of the sites.** The solid dark grey line is the measured $pCO_{2\text{air}}$. Gaps in the data represent periods when water levels were too low (<0.1 m) for sampling. Light and dark shading represents seasons.
doi:10.1371/journal.pone.0071456.g004

Areal air-water CO$_2$ fluxes

Daily air–water CO$_2$ fluxes calculated for the period March 2010–2011 ranged from $-19$ to 197 mmol$_{CO_2}$ m$^{-2}$ d$^{-1}$ and had different statistical distributions at each site (Table 3). At most sites...
the mean and median values were not equal and the distributions were often positively skewed, indicating a disproportionate number of higher values (i.e., highly positive fluxes). Significant differences in fluxes were found between sites (Kruskal-Wallis rank sum test, \(X^2\) (10) ~ 1435.55, \(p < 0.05\)). M9 had the highest median daily \(F_{CO2}\), followed by a group formed by M5, M8, M10 and M11 (19 mmolCO2 m\(^{-2}\) d\(^{-1}\)) (Table 3, KWMC, \(p < 0.05\)). The lowest median daily \(F_{CO2}\) was observed in M3, followed by a group formed by M6 and M2, and slightly higher values observed at site M4 and M1.

The dominance of large positive values (representing release of CO2 to the atmosphere) was clearly observable in the seasonal variations of \(F_{CO2}\) (Fig. 6, dark red represents a strong efflux of CO2). Periods with highly positive \(F_{CO2}\) (10 to 200 mmolCO2 m\(^{-2}\) d\(^{-1}\)), were observed in spring-early summer 2010 for all sites and at most sites (excluding M2 and M3) in the winter 2010–2011. Negative \(F_{CO2}\) were generally of a smaller magnitude (−5 to −10 mmolCO2 m\(^{-2}\) d\(^{-1}\)) and were observed, if only for very short periods, at all sites.

### Annual areal air-water CO2 fluxes

Summing daily \(F_{CO2}\) values over the year revealed that, despite having the shortest hydroperiod, site M9 was the largest source of CO2 to the atmosphere (Table 4). The other wetland sites (M5, M10 and M11) also had highly positive annual \(F_{CO2}\) as did the dune pond site (M8) and La Rocina stream (M7). In contrast wetland site M6 had a small positive annual \(F_{CO2}\), which considering the median \(F_{CO2}\) at this site was 2.2 mmolCO2 m\(^{-2}\) d\(^{-1}\), highlights the importance of short-term, but high magnitude efflux events in determining annual balances. The sites in Veta La Palma (M1:M4) were also close to equilibrium with the atmosphere and, consistent with the very low values of \(p_{CO2\text{water}}\) observed throughout the year, M3 had a small negative \(F_{CO2}\), making it an annual sink for CO2.

### Regional air–water CO2 transport rates

Upscaling daily areal fluxes considering the respective surface area of water within each region (Fig. 1 and 2c) clarified the overwhelming importance of the seasonal flooding of the temporal wetlands (Table 4). Because of the much larger surface area of water (4–7 times that of Veta la Palma during the flood period), the contribution of the temporal wetlands to daily air–water CO2

---

**Table 2.** Descriptive statistics for water CO2 partial pressure (\(p_{CO2\mu atm}\)) at each of the sites.

| ST.ID | M1  | M2  | M3  | M4  | M5  | M6  | M7  | M8  | M9  | M10 | M11 |
|-------|-----|-----|-----|-----|-----|-----|-----|-----|-----|-----|-----|
| Mean  | 758 | 250 | 201 | 458 | 3382| 729 | 1763| 2803| 5382| 2715| 1667|
| Median| 578\(^{ab}\) | 170\(^{ab}\) | 115\(^{a}\) | 302\(^{ab}\) | 695\(^{ab}\) | 115\(^{ab}\) | 847\(^{ab}\) | 222\(^{ab}\) | 632\(^{b}\) | 1871\(^{ab}\) | 1691\(^{b}\) |
| StDev | 525 | 209 | 222 | 469 | 4373| 1186| 2205| 3237| 2942| 3460| 1255|
| Minimum| 111 | 37  | 52  | 15  | 83  | 27  | 34  | 5   | 294 | 130 | 54  |
| Maximum| 1549| 486 | 825 | 1455| 10983| 3843| 7400| 10230| 8599| 10612| 3037|
| Range  | 1438| 449 | 773 | 1440| 10900| 3816| 7365| 10225| 8305| 10482| 2983|
| Interquartile Range | 545 | 388 | 144 | 344 | 6086 | 912 | 1604 | 4280 | 2571 | 2330 | 2211 |
| Skewness| 0.4 | 0.2 | 2.3 | 1.3 | 0.9 | 1.8 | 1.7 | 1.1 | −0.8 | 1.7 | −0.1 |
| Kurtosis| 2.0 | 1.2 | 7.0 | 3.6 | 2.0 | 5.4 | 4.9 | 3.4 | 2.5 | 4.6 | 1.2 |

Letters indicate significantly different groups (KWMC, \(p < 0.05\)).

doi:10.1371/journal.pone.0071456.t002

---

**Figure 5.** Calculated water–side gas transfer velocities (\(k_w\)) between Mar. 2010 and Feb. 2011 for each site.
doi:10.1371/journal.pone.0071456.g005
transport rates during the flood period (5–20 MmolCO₂ y⁻¹) was an order of magnitude larger than the other regions. Consistent with the seasonal patterns in flooding and areal fluxes, the highest effluxes were observed in winter–spring with near zero or negative values found in summer–autumn in all regions.

Estimates of regional annual air–water C transport were 12.95, 0.21, 0.004 and −0.06 GgC y⁻¹ for the wetlands of Doñana National Park (M5, M6, M9:M11), Dune ponds (M8), La Rocina stream (M7) and the ponds of Veta La Palma (M1:M4), respectively. Combined together these give an estimate of total annual (2010–2011) air–water C transport of 13.1 GgC y⁻¹.

### Discussion

Air-water CO₂ transport in the region of Doñana National Park was dominated by seasonal flooding. Strong areal effluxes of CO₂ coincided with the seasonal maximums in water coverage (spring–early summer 2010 and winter 2011) resulting in high rates of net annual air-water CO₂ transport to the atmosphere. These fluxes were a direct consequence of the the CO₂ super-saturation of waters with respect to atmospheric equilibrium. This agrees with previous measurements of the ratio of gross primary production (GPP) to community respiration (CR) in the marshes of Doñana, which also suggested that flood waters were strongly hetertrophic (GPP) to community respiration (CR) in the marshes of Don˜ana, previous measurements of the ratio of gross primary production (GPP) to community respiration (CR) in the marshes of Don˜ana, agreeing with this general pattern was observed in most of the semi-permanent water bodies, however, influxes were rather small compared to the large efflux events. At a regional scale, the much larger area of water coverage within the marshes during the flood period dominated calculations of air-water CO₂ transport, overall resulting in an estimate of net CO₂ transfer from flood waters to the atmosphere.

In terms of their physicochemical properties and F_{CO₂} the sites in Veta la Palma, of which M3 was the only site with water all year, were clearly different compared to the National Park (Fig. 3, Table 1 and 2). Apart from being generally mesohaline, M3 also had extreme salinity variations (changing from 0 to 30 within a month) indicating management of the water levels within this pond. Indeed, because M3 is used for extensive aquaculture of commercially important fish and shrimp species, water levels are maintained by pumping in water from the Guadalquivir, which also has the side-effect of providing a suitable all year round habitat for numerous bird species [62]. These high salinity values resulted in calculated water-side gas transfer coefficients about 2 times higher than other sites in summer–autumn. Combined with the low water pCO₂ values observed throughout most of the year, this led to mild negative F_{CO₂}, resulting in the pond acting as a mild annual sink for CO₂.

Very high rates of GPP have been previously reported in the semi–permanent, hyper–trophic lake Sant Olalla, situated next to Laguna Dulce (M8) [61]. Similar to our study, the highest rates of GPP were observed in spring and late summer and were accompanied by periods of very high CR meaning that annually pelagic metabolism was essentially in balance. In our study this general pattern was observed in most of the semi-permanent water bodies, however, influxes were rather small compared to the large efflux events. At a regional scale, the much larger area of water coverage within the marshes during the flood period dominated calculations of air-water CO₂ transport, overall resulting in an estimate of net CO₂ transfer from flood waters to the atmosphere.

### Table 3. Descriptive statistics for daily areal air–water CO₂ fluxes (F_{CO₂}, mmolCO₂ m⁻² d⁻¹) at each of the sites.

| Site     | M1  | M2  | M3  | M4  | M5  | M6  | M7  | M8  | M9  | M10 | M11 |
|----------|-----|-----|-----|-----|-----|-----|-----|-----|-----|-----|-----|
| Mean     | 5.5 | −1.5| −4.1| 2.0 | 33.8| 3.7 | 13.8| 27.7| 73.9| 29.9| 19.5|
| Median   | 2.4b| −1.3b| −4.1c| −0.7b| 20.1d| −2.2b| 6.5a| 22.8d| 76.6| 14.7d| 17.8d|
| Standard Deviation | 8.1 | 2.3 | 4.1 | 6.6 | 39.7 | 11.6 | 19.3 | 28.9 | 35.4 | 39.1 | 14.5 |
| Minimum  | −6.9| −9.3| −19.1| −6.8 | −5.3 | −10.2| −4.3 | −7.7 | −0.8 | −3.3 | −2.8 |
| Maximum  | 36.8| 2.4 | 12.6| 29.3 | 193.4| 55.7 | 114.6| 172.7| 177.3| 197.0| 70.2 |
| Range    | 43.7| 11.7| 31.6| 36.1 | 198.7| 66.0 | 118.9| 180.4| 178.1| 200.3| 73.0 |
| Interquartile Range | 10.0 | 3.0 | 3.6 | 7.4 | 53.3 | 11.4 | 14.4 | 34.6 | 42.7 | 44.2 | 19.5 |
| Skewness | 1.3 | −0.8 | 0.5 | 1.5 | 1.4 | 1.7 | 2.3 | 1.7 | 0.2 | 1.8 | 0.7 |
| Kurtosis | 4.4 | 3.1 | 6.0 | 5.1 | 5.0 | 5.4 | 9.2 | 7.5 | 3.1 | 6.0 | 3.3 |

Letters indicate significantly different groups (KWMC, p<0.05).

doi:10.1371/journal.pone.0071456.t003
molC m$^{-2}$ y$^{-1}$) is similar to the global average for large lakes of the world (6.0 molC m$^{-2}$ y$^{-1}$, [9]) and the nearby Guadalacacin reservoir (4.6 molC m$^{-2}$ y$^{-1}$, M. Morales Pers. Comm). The highest flux (13.4 molC m$^{-2}$ y$^{-1}$), observed in the temporal wetlands, is similar to the average for European estuaries (15.8 molC m$^{-2}$ y$^{-1}$, [66]), and more locally, the nearby Rio San Pedro tidal creek (16.9 molC m$^{-2}$ y$^{-1}$, [67]) and Bornos reservoir (18.4 molC m$^{-2}$ y$^{-1}$, M. Morales Pers. Comm). However, it is about half the flux estimated for the Guadalquivir estuary (31.1 molC m$^{-2}$ y$^{-1}$, [68]) and an order of magnitude smaller than super-emitters such as the Amazon floodplain [25,69]. Reports of negative annual air-water fluxes (i.e., net uptake of CO$_2$) are relatively rare for inland and transitional waters, nevertheless, the lowest value observed in Veta La Palma (−1.6 molC m$^{-2}$ y$^{-1}$) is
similar to that reported for Aby Lagoon, Ivory Coast (−3.9 molC m⁻² y⁻¹ [70]) and near-shore waters of the Gulf of Cadiz (−0.4 molC m⁻² y⁻¹ [71]).

Air–water transport is only one component of the annual net ecosystem CO₂ exchange (NEE) of the marshes, uptake of CO₂ directly from the air by vascular plants and soil exchange during dry periods are not included here. Indeed, measurements of annual NEE in wetlands actually tend to suggest they are sinks for CO₂ (values ranging between −2 and −15 molC m⁻² y⁻¹, see references in [72]). Similar to our results, periods with large effluxes are often observed in winter when plant growth is minimal, however these are offset by the high rates of net primary production (NPP) of emergent macrophytes during the growing season [73].

Whilst specific data on NEE in the Doñana marshes is not presently available (eddy covariance measurements are planned, http://icts.ebd.csic.es), potential net primary production (NPP) of the surrounding forests, marshes, heath and scrubland is estimated to be about 41, 27, 14 and 3 molC m⁻² y⁻¹ [74–77]. Assuming a constant NPP of marshes (27 ± 18 molC m⁻² y⁻¹) for the whole Doñana wetland region (231 km²), provides an upper estimate of potential NPP of 75 ± 50 GgC y⁻¹, which is 2 to 10 times higher than annual aquatic air-water transport (13.1 GgC y⁻¹). Although, this initial estimate of marsh NPP needs improvement and methane is likely to be quantitatively important [78],

| Region              | Site | F_{CO2} (molCO₂ m⁻² y⁻¹) | Water extent (km²) | C transport (GgC y⁻¹) |
|---------------------|------|-------------------------|--------------------|-----------------------|
| Veta la Palma      | M1   | 1.1                     | 14–37              | −0.06                 |
|                     | M2   | −0.3                    |                    |                       |
|                     | M3   | −1.6                    |                    |                       |
|                     | M4   | 0.5                     |                    |                       |
| Doñana wetlands    | M5   | 11.9                    | 1–224              | 12.95                 |
|                     | M6   | 1.4                     |                    |                       |
|                     | M9   | 13.4                    |                    |                       |
|                     | M10  | 8.8                     |                    |                       |
|                     | M11  | 5.1                     |                    |                       |
| Dune ponds         | M8   | 11.1                    | 0.1–4              | 0.21                  |
| La Rocina          | M7   | 6.0                     | 0.0–0.1            | 0.004                 |

doi:10.1371/journal.pone.0071456.t004

Figure 7. Histograms of annual mean pCO₂water (A) and annual areal air-water F_{CO2} (B) of a range of inland, transitional and coastal–marine aquatic systems (citations in text). Bars represent data bins of 400 µatm (total n = 49) and 5 molC m⁻² y⁻¹ (total n = 58), respectively. Coloured lines and text represent the measured pCO₂water values for each site in this study. Labels highlight examples of annual areal air-water F_{CO2} for different aquatic systems mentioned in the discussion.

doi:10.1371/journal.pone.0071456.g007

Table 4. Summary of annual areal air–water CO₂ fluxes (F_{CO2}) at each of the sites, range of water coverage and annual air–water C transport in each region.
suggests that; 1) despite a large efflux of CO₂ to the atmosphere during the flood period, annual NEE of the Doñana wetlands is still likely to be negative i.e., the ecosystem acts as a CO₂ sink, and 2) the main source of C within the aquatic systems of Doñana is probably the primary production of plants within the local ecosystem. Ponds within the managed wetland (Veta La Palma) that retained water during the dry period tended to be weak annual net CO₂ sinks. Whilst a thorough investigation of C (and other GHGs) transport is still necessary, this maybe an indication of another valuable ecosystem service provided by this particular type of low intensity aquaculture activity. For the Natural wetlands, predictions about the affects of a shortened hydroperiod [79,80] on ecosystem CO₂ transport rates are likely to be complicated. However based on this limited dataset, we can anticipate that shorter hydroperiods may not necessarily mean less air-water transport (the site with the shortest period was the most intense emitter of CO₂) and at some point the NPP of the marshes will be reduced by water shortages. This suggests that the hydrological restoration program presently under way to restore the connectivity between the marsh and the Guadalquivir estuary, which should increase the hydroperiod and water renewal within the marshes, may enhance the C sequestration ecosystem service provided by the Doñana Natural Area.

Acknowledgments

Maria Ferrer-Marcó, Manuel Arjouilla and Antonio Moreno are thanked for their involvement in sample collection and analysis. We thank the staff of Doñana Natural Area for logistical assistance. We are grateful to Miguel Medialdea, the staff and owners of Veta la Palma for giving us information about and access to their aquaculture ponds. Meteorological data was provided by the ICTS Doñana Scientific Reserve (ICTS-2009-39), Spanish Ministry of Science and Innovation. Landsat data available from the U.S. Geological Survey. DEIMOS data provided by DEIMOS-Imaging.

Author Contributions

Conceived and designed the experiments: SF EH JF PR GN EC JR. Performed the experiments: SF EH JF PR GN JR EC. Analyzed the data: EPM SF EH PR JF GN JR EC. Contributed reagents/materials/analysis tools: SF EH JF GN JR EC. Wrote the paper: EPM SF EH JF GN JR EC.

References

1. Battin TJ, Layssaert S, Kaplan LA, Auldenkampe AK, Richter A, et al. (2009) The boundless carbon cycle. Nature Geosci 2: 606–610.
2. Downing JA (2009) Plenary lecture global limnology: Up-scaling aquatic services and processes to planet Earth. Verh Internat Verein Limnol 30: 1149–1166.
3. Borges AV (2005) Do we have enough pieces of the jigsaw to integrate CO₂ fluxes in the coastal ocean? Estuaries 28: 3–27.
4. Borges A, Abril G (2011) Carbon dioxide and methane dynamics in estuaries. In: Editors-in-Chief: Eric Wolanski, Donald McLusky, editors, Treatise on Estuarine and Coastal Science, Waltham: Academic Press. 119–161. Available: http://www.sciencedirect.com/science/article/pii/B9780123477411200540.
5. Cole JJ, Prairie YT, Caraco NF, McDowell WH, Travisik LJ, et al. (2007) Plumbing the global carbon cycle: Integrating inland waters into the terrestrial carbon budget. Ecosystems 10: 172–185.
6. Travisik LJ, Downing JA, Cotter JB, Louisel SA, Striegl RG, et al. (2009) Lakes and reservoirs as regulators of carbon cycling and climate. Limnol Oceanogr 54: 2298–2314.
7. Downing JA (2010) Emerging global role of small lakes and ponds: Little things mean a lot. Limnetica 1: 9–24.
8. Dean WE, Gorham E (1998) Magnitude and significance of carbon burial in lakes, reservoirs, and peatlands. Geology 26: 535–538.
9. Alin SR, Johnson TC (2007) Carbon cycling in large lakes of the world: A synthesis of production, burial, and lake-atmosphere exchange estimates. Global Biogeochem Cycles 21: 12 P.
10. Duarte CM, Prairie YT, Montes C, Cole JJ, Striegl R, et al. (2008) CO₂ emissions from saline lakes: A global estimate of a surprisingly large flux. J Geophys Res 103: 7 P.
11. St Louis VL, Kelly CA, Duchemin E, Rudd JWM, Rosenberg DM (2000) Reservoir surfaces as sources of greenhouse gases to the atmosphere: A global estimate. BioScience 50: 766–773.
12. Cole JJ, Caraco NF (2001) Carbon in catchments: Connecting terrestrial carbon losses with aquatic metabolism. Mar Fresh Res 52: 101–110.
13. Frankignoulle M, Abril G, Borges A, Bourge I, Canon C, et al. (1998) Carbon dioxide emission from European estuaries. Science 202: 434–436.
14. Borges A, Delille B, Frankignoulle M (2005) Budgeting sinks and sources of CO₂ in the coastal ocean: Diversity of ecosystems counts. Geophys Res Lett 32: 4 P.
15. Battin TJ, Kaplan LA, Findlay S, Hopkinson CS, Marti E, et al. (2008) Biophysical controls on organic carbon fluxes in fluvial networks. Nature Geosci 1: 95–100.
16. Downing JA, Prairie YT, Cole JJ, Duarte CM, Travisik LJ, et al. (2006) The global distribution and size distribution of lakes, ponds, and impoundments. Limnol Oceanogr 2: 2301–2317.
17. Seekell DA, Pace ML (2011) Does the Pareto distribution adequately describe the size distribution of lakes? Limnol Oceanogr 56: 350–356.
18. Borges AV, Abril G, Deilie B, Desy JP, Darchambeau F (2011) Diffusive methane emissions to the atmosphere from Lake Kivu (Eastern Africa). J Geophys Res 116: 03030.
19. Downing JA, Cole JJ, Middelburg JJ, Striegl RG, Duarte CM, et al. (2008) Sediment organic carbon burial in agriculturally eutrophic impoundments over the last global Biogeochem Cycles 22: 10 P.
20. Melton JR, Wania R, Hudson EL, Poulet B, Ringeval B, et al. (2013) Present state of global wetland extent and wetland methane modelling: Conclusions from a model inter-comparison project (WETCHIMP). Biogeosciences 10: 753–781.

21. Sahagian D, Melack J (1998) Global wetland distribution and functional characterization: Trace gases and a hydrologic cycle. UNIT Digital Library.
22. Mita S, Wassmann R, Vlek PLG (2005) An appraisal of global wetland area and its organic carbon stock.Curr Sci 88(1): 25–35.
23. Cao M, Marshall S, Gregory K (1996) Global carbon exchange and methane emissions from natural wetlands: Application of a process-based model. J Geophys Res 101: 14399–14414.
24. Kayrani B, Scholz M, Mutafa A, Hedmark MA (2010) Carbon storage and fluxes within freshwater wetlands: A critical review. Wetlands 30: 111–124.
25. Richley JE, Melack JM, Auldenkampe AK, Ballester VM, Hess LL, et al. (2002) Outgassing from Amazonian rivers and wetlands as a large tropical source of atmospheric CO₂. Nature 416: 617–620.
26. Hooijer A, Page S, Canadell JG, Silvius M, Koedijk J, et al. (2010) Current and future CO₂ emissions from drained peatlands in Southeast Asia. Biogeosciences 7: 1565–1574.
27. Giorgi F, Lionello P (2008) Climate change projection changes for the Mediterranean region. Global Planet Change 63: 90–104.
28. Aldaya MM, Garcia-Novoa F, Llamas MR (2010) Incorporating the water footprint and environmental water requirements into policy: Reflections from the Doñana Region (Spain). In: Papeles de Agua Virtual, Santander: Fundación Marcelino Botin, number 5 in Observatorio del Agua. p. 193. Available: http://www.fundacionbotin.org/.
29. Balmer MB, Downing JA (2011) Carbon dioxide concentrations in eutrophic lakes: Underestimation implies atmospheric uptake. Inland Wat 1: 125–132.
30. Yvon-Durocher G, Jones JI, Trimmer M, Woodward G, Montoya JM (2010) Warming alters the metabolic balance of ecosystems. Phil Trans R Soc B 365: 1127–1126.
31. Mos B, Kosten S, Meerkof M, Battarbee R, Jeppeš E, et al. (2011) Allied attack: Climate change and eutrophication. Inland Wat 1: 101–105.
32. Nuttегen P, Moodley I, Brummer G, Heip CHR, Herman PMJ, et al. (2009) Seafloor ecosystem functioning: The importance of organic matter priming, Mar Biol 156: 2227–2237.
33. Kuyakov Y (2010) Priming effects: Interactions between living and dead organic matter. Soil Biol Biochem 42: 1363–1371.
34. Bianchi T (2011) The role of terrestrially derived organic carbon in the coastal ocean: A changing paradigm and the priming effect. Proc Natl Acad Sci 108: 19473–19481.
35. Fernández N, Panuelo JM, Delibes M (2010) Ecosystem functioning of protected and altered Mediterranean environments: A remote sensing classification in Doñana, Spain. Remote Sens Environ 114: 211–220.
36. Espinar JL, Serrano L (2009) A quantitative hydrogeomorphic approach to the classification of temporary wetlands in the Doñana National Park (SW Spain). Aquat Ecol 45: 325–334.
37. Rendon MA, Green AJ, Aguilara E, Almaraza P (2008) Status, distribution and long-term changes in the waterbird community wintering in Doñana, southwest Spain. Bird Conserv 114: 1371–1388.
38. Grasshoff K, Ehrhardt M, Kremling K, Almgren T (1983) Methods of seawater analysis, volume 2. Wiley Online Library. Available: http://onlinelibrary.wiley.com/doi/10.1002/9783527671394.fnatter/summary.
39. Minnport L, Perez FF, Gonzalez-Davila M, Santana-Casiano JM, Kortzinger A (2000) Alkalinity determination by potentiometry: Intercalibration using three different methods. Cienca Mar 26:23–27.

PLOS ONE | www.plosone.org 13 August 2013 | Volume 8 | Issue 8 | e71456

Doñana Wetland Carbon Sequestration
