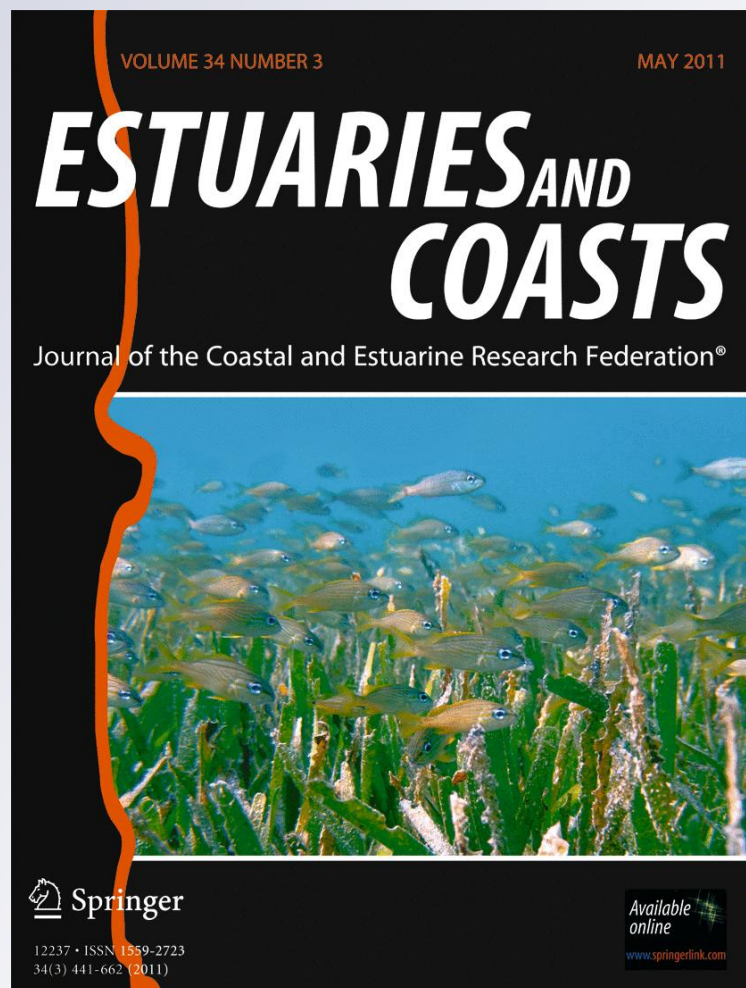


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# Sustainability of Mediterranean Deltaic and Lagoon Wetlands with Sea-Level Rise: The Importance of River Input

John Day · Carles Ibáñez · Francesco Scarton ·  
Didier Pont · Philippe Hensel · Jason Day · Robert Lane

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**Abstract** We report on a decadal trend of accretionary dynamics in the wetlands of several northwestern Mediterranean deltas and a lagoon system, all of them with high rates of wetland loss. Wetland vertical accretion and surface elevation change were measured at 55 riverine, marine, and impounded sites in four coastal systems: the Ebro delta, Spain; the Rhône delta, France; and the Po delta and Venice Lagoon, Italy. Vertical accretion and elevation change ranged between 0 and 25 mm year<sup>-1</sup> and were strongly correlated. The highest rates of elevation gain occurred at riverine sites where vertical accretion was highest. We conclude that areas with high sediment input, mainly riverine, are the only ones likely to survive

accelerated sea-level rise, especially if recent higher estimates of 1 m or more in the twenty-first century prove to be accurate. This is the first study where the importance of river input on wetland survival has been demonstrated at a decadal time scale over a broad geographical area.

**Keywords** Deltas · Sediment input · Flood pulse · Sea-level rise · Mediterranean wetlands

## Introduction

There are serious environmental problems in deltas and estuaries including salinity intrusion, water quality deterioration, decreased biological production, and wetland loss. These problems result from such factors as reduced freshwater and sediment input due to dams and levees; hydrological modifications from canals and impoundments; and increased subsidence from shallow subsurface extraction of oil, gas, and groundwater (Baumann et al. 1984; Pont et al. 2002; Day et al. 2007; FitzGerald et al. 2008; Syvitski et al. 2009; Vorosmarty et al. 2009). High subsidence is especially critical for deltaic wetlands as it results in rates of relative sea-level rise [RSLR=eustatic sea-level rise (ESLR)+subsidence] that are often much greater than ESLR alone (Stanley 1990; Vorosmarty et al. 2009). Because of this high rate of RSLR, deltas can serve as models for the impacts of accelerated ESLR in other coastal areas (Day and Templet 1989).

Deltaic wetlands survived and expanded over the past several thousand years in areas with high riverine input (Roberts 1997; Day et al. 2007; Syvitski et al. 2009). River water enhances wetland vertical accretion directly via sediment deposition and indirectly via enhanced root growth and organic soil formation (Mendelssohn and

J. Day (✉) · J. Day · R. Lane  
Department of Oceanography and Coastal Sciences,  
School of the Coast and Environment, Louisiana State University,  
Baton Rouge, LA 70803, USA  
e-mail: johnday@lsu.edu

C. Ibáñez  
Aquatic Ecosystems Program, IRTA,  
Carretera Poble Nou, km 5.5, 43540 Sant Carles de la Ràpita,  
Catalonia, Spain

F. Scarton  
SELC Società per l'Ecologia delle Lagune e delle Coste soc. coop.,  
Via dell'Elettricità 3/d,  
30175 Venezia, Italy

D. Pont  
Hydrosystems and Bioprocesses Research Unit (HBAN),  
CEMAGREF,  
Parc de Toruvoie, BP44,  
92163 Antony Cedex, France

P. Hensel  
NOAA National Geodetic Survey,  
1315 East-West Highway,  
Silver Spring, MD 20910, USA

Morris 2000). Sediment input to deltaic and estuarine areas is associated with episodic pulsing events, such as major storms events and river floods, which operate on different temporal and spatial scales (Day et al. 1997, 2007). A primary cause of wetland loss in deltas is from human activities that have reduced or eliminated these forcings (Stanley 1990; Day et al. 2000, 2007; Pont et al. 2002).

Current ESLR is on the order of 3 mm year<sup>-1</sup> (FitzGerald et al. 2008). There is scientific consensus that the rate of ESLR will accelerate in association with global warming, and the Intergovernmental Panel on Climate Change (IPCC 2007) predicts sea-level rise will be about 40 cm by 2100, with a range of uncertainty of 10–54 cm (IPCC 2007). Recent evidence suggests that this prediction may be an underestimation, as ESLR may be up to a meter or more (Rahmstorf 2007; Pfeffer 2008; Mitrovica et al. 2009; Vermeer and Rahmstorf 2009). For a review of issues relative to climate change and the Mediterranean and European coastal lowlands in general, see Jeftic et al. (1992) and Tooley and Jelgersma (1992); for the Mississippi delta, see Day and Templet (1989) and Day et al. (2005, 2007). To survive, coastal wetlands will have to increase rates of accretion and vertical elevation gain in pace with accelerated sea-level rise.

The objective of this study was to measure vertical accretion and surface elevation changes in a variety of coastal systems associated with deltas and lagoons in the northwestern Mediterranean with different human and natural forcings. We hypothesized that areas with strong riverine input would have higher rates of accretion and elevation gain than non-riverine sites, and that these sites would be the only ones likely to survive accelerated sea-level rise.

### Study Locations

The study sites represent a broad range of coastal wetland habitats in the northwestern Mediterranean (Fig. 1). The habitats include freshwater and low salinity tidal *Phragmites australis* marshes at the mouths of the three large rivers and the Dese River that discharges into Venice Lagoon; marine sites with low freshwater influence, estuarine tidal marshes, and impounded freshwater and saltwater marshes. In all, there were five sites in the Ebro delta, nine sites in the Rhône delta, three sites in the Po delta, and 11 sites in Venice Lagoon. Results are reported for 55 individual stations at these sites where surface elevation change and vertical accretion were measured. Detailed descriptions of the sites are provided elsewhere (Rhône—Hensel et al. 1998, 1999; Pont et al. 2002; Ebro—Ibáñez et al. 1997, 2010; Curcó et al. 2002; Po and Venice Lagoon—Scarton et al. 1998, 2002; Day et al. 1999).

Since the beginning of this research effort in the early 1990s, study sites have been categorized as riverine, marine, and impounded based on the different characteristics of the areas (Table 1). *P. australis* is the dominant species at riverine sites that receive large volumes of river water input. The dominant vegetation at marine sites is *Sarcocornia fruticosa*, which is characteristic of coastal lagoons and bay edges where there is regular tidal inundation, and where soils have a moderate variation of moisture and salinity during the year. *Arthrocnemum macrostachyum* dominates the impounded sites, and generally occurs at somewhat higher elevations that are seasonally flooded and that have a strong hypersaline period during summer. *P. australis* at riverine sites is generally 3–4 m in height, while *S. fruticosa* ranges from about 30–60 cm, and *A. macrostachyum* is generally less than 20 cm tall. Vegetation diversity is higher at Venice Lagoon where there is higher rainfall (about 800 mm year<sup>-1</sup> compared to 548 mm year<sup>-1</sup> at the Ebro and 590 mm year<sup>-1</sup> at the Rhône) and tidal range (about 60 cm vs. about 16 cm at the Rhône and Ebro deltas). In addition to *S. fruticosa*, *Limonium serotinum*, *Puccinellia palustris*, and *Spartina maritima* also occur at the marine marshes. Above- and belowground biomass in the Mediterranean wetlands is highly variable, but riverine and marine sites have much higher biomass than impounded sites. The highest belowground biomass occurs at riverine sites while the highest aboveground biomass is at marine sites. Mineral sediment input is much higher at riverine sites with values as high as 6,000 gm<sup>-2</sup> year<sup>-1</sup>. Sediment input ranges from 900 to 1,200 gm<sup>-2</sup> year<sup>-1</sup> at the marine sites, while input to the impounded sites is near zero. Pore water conductivity is low at riverine sites, reflecting high freshwater input, generally less than full seawater at marine sites, and most often hypersaline at impounded sites. There are no consistent differences in soil organic matter or bulk density among the sites, reflecting the complex interaction among factors such as species composition, productivity and decomposition, mineral sediment input, and hydrology.

### Materials and Methods

Accretion and wetland surface elevation change were measured at the different sites between 1992 and 2002. Two replicate stations were randomly established at each site. In the Ebro delta, one station was damaged early in the study and was not used in this analysis, which is why there are 28 sites and only 55 stations. Accretion was measured as the accumulation of material over feldspar marker horizons placed on the soil surface as described by Cahoon et al. (1995) and Day et al. (1999). Three replicate 0.25-m<sup>2</sup> marker horizons were randomly placed at each station.

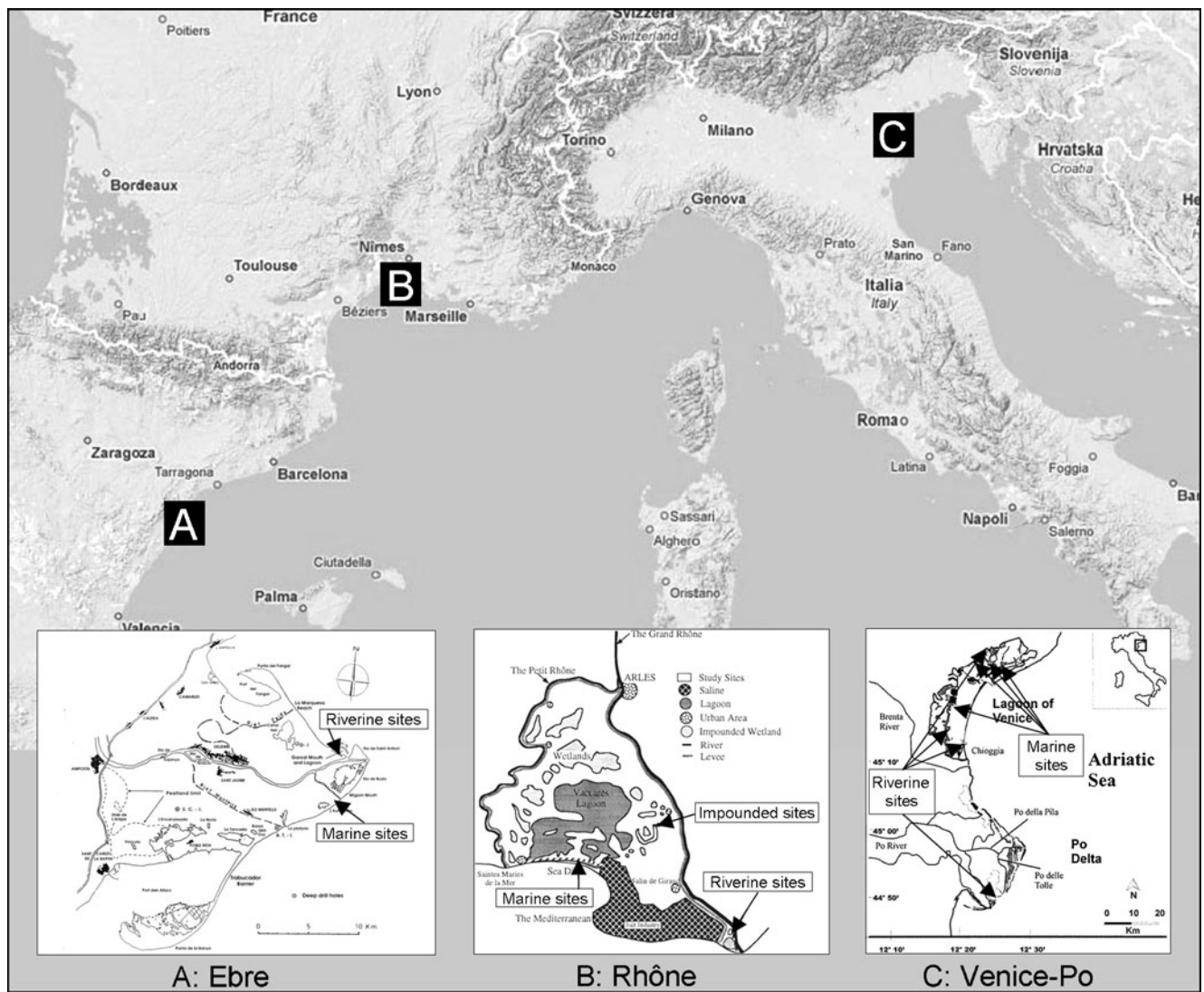


Fig. 1 Study areas and location of the study sites in the northwestern Mediterranean

**Table 1** Characteristics of riverine, marine, and impounded sites in northwestern Mediterranean wetland sites

Parameter	Site		
	Riverine	Marine	Impounded
Water level variation	High	Moderate	Low
Water level forcing	Rain, tide, river	Rain, tide	Rain
Dominant vegetation	<i>Phragmites australis</i>	<i>Sarcocornia fruticosa</i>	<i>Arthrocnemum macrostachyum</i>
Aboveground biomass (g/m <sup>2</sup> )	780–2,824	767–3,561	254–732
Belowground biomass (g/m <sup>2</sup> )	1,590–5,142	500–2,829	100–435
Vegetation height (cm)	300–400	33–43	4.0–6.5
Bulk density, 0–10 cm (g/cm <sup>2</sup> )	0.9–1.22	0.9–1.3	0.74–1.44
Soil organic matter (%)	4.5–12.4	6.9–9.8	0.8–17.6
Soil conductivity (mS/cm)	6.9–14.9	32.8–57.4	47.8–126
Mineral input (g m <sup>-2</sup> year <sup>-1</sup> )	1,611–6,000	905–1,202	13–34

Vegetation biomass is grams of dry matter. Parameters are representative values for the different sites. Data from Hensel et al. 1998, 1999; Scarton et al. 1998, 2002; Day et al. 1999; Ibáñez et al. 1999, 2010; Curcò et al. 2002; Pont et al. 2002

Wetland surface elevation changes were measured with a surface elevation table (SET) as described by Boumans and Day (1994) and Cahoon et al. (2000), with two replicate SET stations established at each site. We also compiled additional data on above- and belowground biomass, tidal range, and conductivity from the literature for the different sites (Tables 2, 3, and 4).

The rates of surface elevation change and vertical accretion were compared with estimates of ESLR and subsidence. We used 1.8–2.0 mm year<sup>-1</sup> as an average rate of ESLR for the twentieth century (Gornitz et al. 1982; Douglas et al. 2001), and estimates of subsidence for the different locations were taken from the literature. For the Rhône delta, Pont et al. (2002) reported that subsidence in the eastern delta where our sites were located ranged from 1.0 to 4.0 mm year<sup>-1</sup>. The background rate of geologic subsidence in Venice is presently 0.4 mm yr<sup>-1</sup> (Carbognin et al. 2004), but between 1930 and 1970, RSLR was 24 cm (i.e., 6 mm year<sup>-1</sup>), about half of which was due to groundwater withdrawals (Carbognin et al. 2004). For the Po delta, long-term geologic subsidence is reported to be between 1.0 and 5.0 mm year<sup>-1</sup> (Bondesan et al. 1995). In the Ebro delta, sediment depths suggest average rates of subsidence since the Quaternary ranging from 0.1 to 0.3 mm year<sup>-1</sup>. Ibáñez et al. (1997) estimated recent subsidence rates to be 1.0–3.2 mm year<sup>-1</sup> based on the elevation differences of old river channels. The IPCC (2007) predictions for ESLR during the twenty-first century described in the “Introduction” correspond to an average of 4 mm year<sup>-1</sup> with a range of uncertainty of 1 to 5 mm year<sup>-1</sup>. As stated in the “Introduction”, given recent predictions of much higher rates of ESLR, these estimates can be considered conservative. Estimates of current and future rates of RSLR were calculated as the sum of the IPCC (2007) prediction for ESLR and the rate of subsidence for each study area. These estimates of RSLR were compared to the rates of surface elevation change measured for this study.

Mean annual wetland surface elevation change and vertical accretion were regressed using simple linear regression analysis. JMP statistical software (Sall et al. 2007) was used to test for significant differences between the slope of the regression line and the mean (alpha <0.05), and to provide summary statistics (e.g., regression equations and R<sup>2</sup> values). A one-way analysis of variance was carried out to detect differences in wetland surface elevation change among site types. Comparisons of means with significant ANOVA tests (alpha <0.05) were made using Tukey–Kramer Honestly Significant Difference (HSD) test (Sall et al. 2007).

Forward stepwise multiple regression analysis was used to detect relationships among elevation and the other variables (i.e., accretion, shallow subsidence, deep subsidence, aboveground biomass, belowground biomass, tidal

**Table 2** Site characteristics for wetlands at the Ebro delta

Dominant species	Station type	Sediment availability	Surface elevation change (mm year <sup>-1</sup> )	Vertical accretion (mm year <sup>-1</sup> )	Shallow subsidence (mm year <sup>-1</sup> )	Deep subsidence (mm year <sup>-1</sup> )	Aboveground live biomass (g dry m <sup>-2</sup> )	Belowground live biomass (g dry m <sup>-2</sup> )	Tidal range (m)	Mean conductivity (mS/cm)
<i>Arthrocnemum macrostachyum</i>	M	Medium	0.25	1.4	1.15	2	467	123	0.16	74.2
<i>Arthrocnemum macrostachyum</i>	M	Medium	9.53	1.24	-8.29	2	467	123	0.16	74.2
<i>Sarcocornia fruticosa</i>	M	Medium	3.18	1.4	-1.78	2	1,658	793	0.16	58.9
<i>Sarcocornia fruticosa</i>	M	Medium	4.86	2.03	-2.83	2	1,658	793	0.16	58.9
<i>Sarcocornia fruticosa</i>	M	Low	1.17	1.46	0.29	2	n.a.	n.a.	0	n.a.
<i>Sarcocornia fruticosa</i>	M	Low	1.56	0.32	-1.24	2	n.a.	n.a.	0	n.a.
<i>Cladium mariscus</i>	M	Low	-0.05	n.a.	n.a.	1	852	5,142	0	2.8
<i>Phragmites australis</i>	R	High	6.57	4.88	-1.69	3	467	2,372	0.16	6.9
<i>Phragmites australis</i>	R	High	6.65	5.19	-1.46	3	467	2,372	0.16	6.9

**Table 3** Site characteristics for wetlands at the Rhône delta

Dominant species	Station type	Sediment availability	Surface elevation change (mm year <sup>-1</sup> )	Vertical accretion (mm year <sup>-1</sup> )	Shallow subsidence (mm year <sup>-1</sup> )	Deep subsidence (mm year <sup>-1</sup> )	Aboveground live biomass (g dry m <sup>-2</sup> )	Belowground live biomass (g dry m <sup>-2</sup> )	Tidal range (m)	Mean conductivity (mS/cm)
<i>Sarcocornia fruticosa</i>	I	Low	-0.1	0.1	0.22	0.25	321±28	907	0	n.a.
<i>Sarcocornia fruticosa</i>	I	Low	0.0	0.2	0.19	0.25	321±28	907	0	n.a.
<i>Sarcocornia fruticosa</i>	I	Medium	1.0	1.4	0.32	0.6–3.7	2,094±321	1,590	0.16	57.4
<i>Sarcocornia fruticosa</i>	I	Medium	-0.4	0.9	1.26	0.6–3.7	2,094±321	1,590	0.16	57.4
<i>Sarcocornia fruticosa</i>	I	Low	0.5	0.8	0.31	0.7	278±60	n.a.	0.16	84.6
<i>Sarcocornia fruticosa</i>	I	Low	0.9	0.9	0.02	0.7	278±60	n.a.	0.16	84.6
<i>Sarcocornia fruticosa</i>	I	Low	6.8	0.5	-6.33	0.25	254±69	n.a.	0	47.8
<i>Sarcocornia fruticosa</i>	I	Low	0.1	0.2	0.08	0.25	254±69	n.a.	0	47.8
<i>Typha angustifolia</i>	I	Low	1.4	0.4	-1.02	0.25	862±244	n.a.	0	2.7
<i>Typha angustifolia</i>	I	Low	1.5	0.3	-1.16	0.25	862±244	n.a.	0	2.7
<i>Sarcocornia fruticosa</i>	M	Medium	1.4	1.2	-0.25	0.7	1,540±6260	1,161	0.16	34.6
<i>Sarcocornia fruticosa</i>	M	Medium	0.8	0.8	0.00	0.7	1,540±6,260	1,161	0.16	34.6
<i>Phragmites australis</i>	R	High	12.7	20.7	8.00	0.6–3.7	305±74	n.a.	0.16	14.9
<i>Phragmites australis</i>	R	High	21.0	24.4	3.43	0.6–3.7	305±74	n.a.	0.16	14.9
<i>Phragmites australis</i>	R	High	9.1	9.0	-0.10	0.6–3.7	n.a.	n.a.	0.16	14.9
<i>Phragmites australis</i>	R	High	9.4	12.1	2.72	0.6–3.7	n.a.	n.a.	0.16	14.9
<i>Sarcocornia fruticosa</i>	R	Medium	3.5	2.9	-0.52	0.6–3.7	n.a.	n.a.	0.16	34.6
<i>Sarcocornia fruticosa</i>	R	Medium	4.8	2.5	-2.27	0.6–3.7	n.a.	n.a.	0.16	34.6

**Table 4** Site characteristics for wetlands at Venice Lagoon and the Po delta

Dominant species	Station type	Sediment availability	Surface elevation change (mm year <sup>-1</sup> )	Vertical accretion (mm year <sup>-1</sup> )	Shallow subsidence (mm year <sup>-1</sup> )	Deep subsidence (mm year <sup>-1</sup> )	Aboveground live biomass (g dry m <sup>-2</sup> )	Belowground live biomass (g dry m <sup>-2</sup> )	Tidal range (m)	Mean conductivity (mS/cm)
<i>Juncus maritimus</i>	M	Medium	5.1	4.1	-1.00	1.5-2.5	350	500	0.6	32.8
<i>Juncus maritimus</i>	M	Medium	4.2	4.4	0.20	1.5-2.5	350	500	0.6	32.8
<i>Sarcocornia fruticosa</i>	M	Medium	3.0	2.8	-0.20	1.5-2.5	1,200	834	0.6	32.8
<i>Sarcocornia fruticosa</i>	M	Medium	2.7	3.0	0.30	1.5-2.5	1,200	834	0.6	32.8
<i>Spartina maritima</i>	M	Medium	4.8	5.5	0.70	1.5-2.5	500	600	0.6	32.8
<i>Spartina maritima</i>	M	Medium	4.9	3.6	-1.30	1.5-2.5	500	600	0.6	32.8
<i>Sarcocornia fruticosa</i>	M	Medium	0.4	3.0	2.60	1.5-2.5	n.a.	n.a.	0.6	32.8
<i>Sarcocornia fruticosa</i>	M	Medium	0.4	4.0	3.60	1.5-2.5	n.a.	n.a.	0.6	32.8
<i>Sarcocornia fruticosa</i>	M	Medium	0.2	5.0	4.80	1.5-2.5	n.a.	n.a.	0.6	32.8
<i>Sarcocornia fruticosa</i>	M	Medium	-0.7	4.5	5.20	1.5-2.5	n.a.	n.a.	0.6	32.8
<i>Sarcocornia fruticosa</i>	M	Medium	2.0	3.0	1.00	1.5-2.5	n.a.	n.a.	0.6	41.8
<i>Sarcocornia fruticosa</i>	M	Medium	0.5	3.1	2.60	1.5-2.5	n.a.	n.a.	0.6	41.8
<i>Phragmites australis</i>	M	Medium	3.8	2.1	-1.70	3.5-5	780	4,087	0.3	7.5
<i>Phragmites australis</i>	M	Medium	4.0	2.1	-1.90	3.5-5	780	4,087	0.3	7.5
<i>Spartina maritima</i>	M	Medium	1.8	2.7	0.90	1.5-2.5	470	520	0.6	32.8
<i>Spartina maritima</i>	M	Medium	4.3	4.4	0.10	1.5-2.5	470	520	0.6	32.8
<i>Phragmites australis</i>	R	Medium	6.2	6.9	0.70	1-2	800	3,500	0.6	7.5
<i>Phragmites australis</i>	R	Medium	5.7	6.2	0.50	1-2	800	3,500	0.6	7.5
<i>Phragmites australis</i>	R	Medium	4.1	7.6	3.50	1.5-2.5	n.a.	n.a.	0.6	7.5
<i>Phragmites australis</i>	R	Medium	5.4	5.4	0.00	1.5-2.5	n.a.	n.a.	0.6	7.5
<i>Sarcocornia fruticosa</i>	R	High	5.9	15.8	9.90	2.5-3.5	1,000	1,200	0.6	32.8
<i>Sarcocornia fruticosa</i>	R	High	7.8	17.4	9.60	2.5-3.5	1,000	1,200	0.6	32.8
<i>Sarcocornia fruticosa</i>	R	High	3.8	3.1	-0.70	2.5-3.5	1,200	350	0.6	41.8
<i>Sarcocornia fruticosa</i>	R	High	5.5	4.7	-0.80	2.5-3.5	1,200	350	0.6	41.8
<i>Limonium narbonense</i>	R	Medium	8.2	10.3	2.10	2-2.5	250	1,800	0.6	32.8
<i>Limonium narbonense</i>	R	Medium	8.8	13.7	4.90	2-2.5	250	1,800	0.6	32.8
<i>Puccinellia maritima</i>	R	High	10.2	12.0	1.80	2-2.5	220	360	0.6	41.8
<i>Puccinellia maritima</i>	R	High	10.2	12.0	1.80	2-2.5	220	360	0.6	41.8

range, conductivity, sediment availability). Forward stepwise multiple regression analysis selects parameters that best described the response variable, choosing the most significant variable first, and adding additional variables until there was lack of a significant improvement in the fit of the model to the data (Sall et al. 2007). This type of data mining must be interpreted with caution for several reasons, including problems with collinearity, which was tested for and addressed in the analysis.

## Results and Discussion

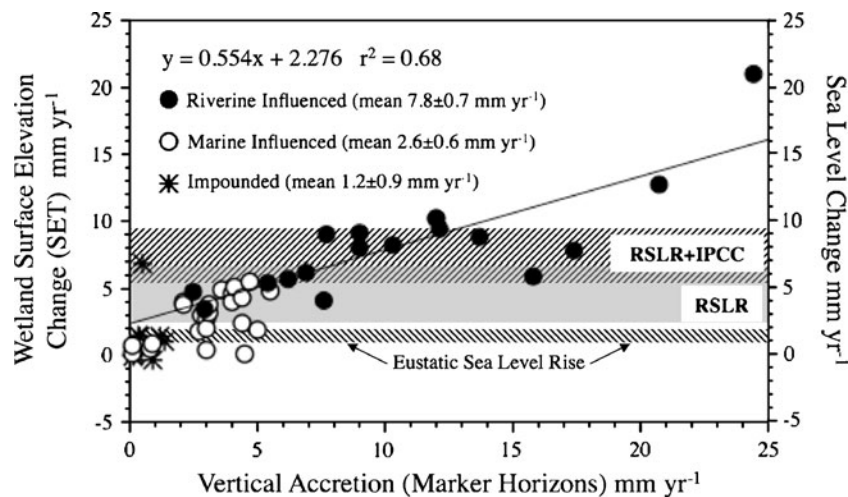
Results show that wetland surface elevation change was strongly correlated with vertical accretion ( $R^2=0.68$ ,  $p<0.01$ ), and that rates of vertical accretion and surface elevation gain were generally much higher at riverine sites (Fig. 2). Vertical accretion was generally greater than surface elevation gain (Fig. 2), with the difference due to shallow subsidence caused by compaction and consolidation of recently deposited sediments (Cahoon et al. 1995). The riverine sites were significantly higher than the marine or impounded sites, which were not significantly different from each other ( $p<0.001$ ). The rate of shallow subsidence in this study was as high as  $9.9 \text{ mm year}^{-1}$  at a site with high organic matter deposition located in Venice Lagoon (Day et al. 1999).

Stepwise regression analysis initially selected parameters accretion and shallow subsidence, but these parameters exhibited collinearity, and thus shallow subsidence data

were excluded from further analysis. Analysis of the abbreviated dataset indicated accretion, aboveground biomass, and sediment availability best describe elevation change ( $p<0.0001$ ,  $R^2=0.76$ ); however, accretion alone significantly described elevation change using simple linear regression ( $p<0.0001$ ;  $R^2=0.68$ ), though with slightly less ability.

Elevation gain generally is more strongly related to accretion at mineral-rich marshes (Cahoon et al. 1995; Hensel et al. 1998, 1999) than in more organic marshes (i.e., Lane et al. 2006). Multiple regression analysis indicated that accretion, aboveground biomass, and sediment availability best explained elevation change. We interpret this as sediment availability leading to accretion and thus to higher elevation gain. Vegetation biomass production is increased not only by sediment input but also by nutrients in river water (Morris and Bradley 1999), which further stimulates accretion and elevation gain.

Sites affected by river input of freshwater, sediments, and nutrients had high vertical accretion and surface elevation gain. For example, in the Rhône delta, the average rate of accretion and elevation change at riverine sites were  $12.5\pm 3.2$  and  $11.4\pm 3.2 \text{ mm year}^{-1}$ , respectively, compared to  $2.2\pm 0.9$  and  $1.6\pm 0.7 \text{ mm year}^{-1}$  at non-riverine sites. In the Ebro delta, accretion ( $7.7 \text{ mm year}^{-1}$ ) and elevation change ( $9.0 \text{ mm year}^{-1}$ ) were greater at the riverine sites compared to isolated marine sites ( $0.1$  vs.  $0.4 \text{ mm year}^{-1}$ ), but were less than at the Rhône delta. Others have reported instances where elevation change is greater than accretion. This results when some of the forces leading to elevation gain, such as swelling of the soil due to



**Fig. 2** Wetland vertical accretion vs. surface elevation change for coastal Mediterranean riverine (black circle), marine (white circle), and impounded (black star) sites. The riverine sites were significantly higher than the marine or impounded sites, which were not significantly different from each other ( $P<0.001$ ). Mean surface elevation changes for the three types were compared to forcings causing water level rise: twentieth century global eustatic sea-level rise (ESLR); relative sea-level rise (RSLR) for the different sites due

to subsidence plus ESLR; and RSLR plus twenty-first century predicted ESLR from the IPCC (2007) (RSLR+IPCC; see text for references). The two vertical axes are identical and apply to both wetland surface elevation change and to sea-level change. To survive rising sea level, coastal wetlands must grow at a rate  $\geq$  water level increase, implying that only sites with high sediment input will survive predicted sea-level rise

groundwater inflow, occur below the marker horizon (a depth of centimeters) but above the bottom of the SET pipe (a depth of several meters; Cahoon et al. 1999, 2006; Cahoon 2006). A similar pattern emerged at the Venice Lagoon and the Po delta. High sediment input was also found at sites bordering large expanses of open water, where riverine derived sediments were resuspended and deposited on the marsh surface during storm events. Average accretion ( $10.7 \pm 4.2$  mm year<sup>-1</sup>) and elevation change ( $7.3 \pm 2.1$  mm year<sup>-1</sup>) at these riverine influenced sites were higher than sites without high sediment input ( $3.7 \pm 1.0$  vs.  $3.3 \pm 1.6$  mm year<sup>-1</sup>).

Other studies have shown the importance of fluvial sediments to long-term accretion and elevation change. An average elevation change of  $11.9 \pm 5.5$  mm year<sup>-1</sup> was recorded at a brackish coastal marsh in South Carolina (Childers et al. 1993). Accretion rates as high as  $18.3$  mm year<sup>-1</sup> were recorded at a fluvial tidal freshwater wetland in Maine. The average for 10 Maine fluvial marshes was  $3.8 \pm 1.7$  mm year<sup>-1</sup>, which is somewhat greater than local RSLR ( $2.5 \pm 0.1$  mm year<sup>-1</sup>; Wood et al. 1989). A mean accretion rate of 7.9 cm in less than a year was reported for wetlands formed by sediment deposition from the Yangtze River (Yang 1998). Baumann et al. (1984) reported that river water inputs led to high accretion in marshes near the mouth of the Atchafalaya River (a tributary of the Mississippi) in Louisiana.

The impounded habitats in the Rhône delta had both low average accretion ( $0.81 \pm 0.1$  mm year<sup>-1</sup>) and elevation change ( $1.9 \pm 0.7$  mm year<sup>-1</sup>). Other studies have shown that coastal wetland areas with restricted connection such as impoundments and areas with water control structures have reduced materials fluxes and accretion (Reed 1992; Boumans and Day 1994; Cahoon 1994). In the Mississippi delta, which has tidal amplitudes similar to the Rhône delta, there was significantly lower accretion at two brackish impoundments compared to adjacent unmanaged marshes (7 vs. 30 mm year<sup>-1</sup>; Cahoon 1994). These two impoundments also had lower short-term sedimentation over a period of one to several weeks, and lower materials fluxes between the impoundments and the adjacent estuary (Boumans and Day 1994). Short-term sedimentation was also lower at several other coastal marshes in the Mississippi delta that were impounded with water control structures as compared to nearby control areas (Reed 1992).

Our results clearly show that coastal wetlands in the northwestern Mediterranean with high mineral sediment input, mainly from rivers, can survive high rates of ESLR. This has implications for the sustainable management of coastal wetlands. Riverine input to these coastal areas, especially deltas, has been greatly altered. Dams and diversions have greatly reduced freshwater and sediment input to the coastal zone, and many rivers are diked to their

mouths, preventing riverine input into deltaic and other coastal wetlands (Day et al. 1995; Day and Giosan 2008; Vorosmarty et al. 2009). These human modifications have fundamentally altered the functioning of river basins and associated coastal ecosystems (Day et al. 2000; Mendelssohn and Morris 2000; Vorosmarty et al. 2009). Our results imply that inputs from rivers and other coastal processes to deltaic and other coastal wetlands are necessary on a variety of spatial and temporal scales if these wetlands are to survive accelerated sea-level rise. The time scales range from daily tides, to storm inputs such as during frontal passages and tropical storms, annual river floods, great river floods that occur a few times a century, to changes in deltaic lobe formation that occur over centuries (i.e., Day et al. 1997, 2007).

Sustainable management approaches to achieve the survival of coastal wetlands include remobilization of sediments trapped in reservoirs, controlled re-introduction of river water into coastal wetlands, restoration of natural hydrology in coastal areas, and use of locally generated, high nutrient runoff to stimulate wetland plant growth (Day et al. 1997, 2007, 2009; Rovira and Ibáñez 2007; Tornqvist et al. 2008; Vorosmarty et al. 2009).

Our finding that riverine input enhances accretion and elevation gain may be at variance with recent findings of the effects of nutrient input on coastal marshes. For example, Darby and Turner (2008a, b) and Turner et al. (2009) reported that additions of inorganic nutrients reduced belowground biomass in *Spartina alterniflora* marshes in the Mississippi delta and along the Atlantic coast of the USA and Canada. They suggested that this might decrease soil elevation and accelerate the conversion of emergent wetlands to open water.

There are several possible reasons why we found enhanced accretion in Mediterranean coastal marshes. First is that the Mediterranean marshes are much more mineral rich. Morris and Bradley (1999) reported that accumulation of inorganic sediments led to an increase in soil elevation in fertilized plots. Impounded sites with low riverine and nutrient input had very low rates of accretion and elevation gain. A second factor that may affect the net accretion balance is the loading rate. For example, studies in the Mississippi delta show that river inputs have total nitrogen and total phosphorus loading rates of less than 20 and 5 g m<sup>-2</sup> year<sup>-1</sup>, respectively (Lane et al. 2002; Hyfield et al. 2008), while Darby and Turner (2008a) used much higher nitrogen loadings before detecting a response. In addition, no sediment or carbon sources were added to the Darby and Turner (2008a, b) plots in contrast to natural nutrient inputs to coastal systems that are often combined with freshwater, sediments, iron, and a variety of organic compounds. Experimental inputs of high levels of inorganic nutrients alone may stimulate organic matter decomposition to a greater degree than inputs that include other compounds.

However, decomposition was not significantly greater at the riverine sites of our study (Ibáñez et al. 1999; Curcó et al. 2002; Pont et al. 2002; Scarton et al. 2002). These questions remain to be fully investigated, but there is no doubt that sites with high riverine inputs in our study had much higher accretion rates. This is in spite of high inorganic nutrient concentrations in the river water. For example, dissolved inorganic phosphorus (DIP) and nitrate ( $\text{NO}_3$ ) levels ranged from 0.08 to 0.12 and 0.4–1.8  $\text{mg l}^{-1}$ , respectively, in the Rhône, and in the Ebro River DIP ranged from 0.2 to 0.7  $\text{mg l}^{-1}$  and  $\text{NO}_3$  averaged about 10  $\text{mg l}^{-1}$  (Hensel 1998; Ibáñez et al. 2008). The sites analyzed in our study of Mediterranean coastal marshes have experienced ambient environmental forcings (impounded, tidal, high riverine input) for decades. Thus, if there were to be impacts of riverine input on the ability to maintain and gain elevation, it would have been evident. In support of our hypothesis, the impounded marshes with no riverine input had very low rates of accretion and elevation gain, and are not keeping pace with local RSLR.

Morris et al. (2002) (see also Morris 2006) developed a model that identified feedbacks among marsh plant production, relative sea level, and sedimentation that establish an equilibrium between relative marsh elevation and the rate of sea-level rise. They predict that there is an optimal rate of sea-level rise that maximizes sediment accretion and productivity, but that the optimal rate is very close to a tipping point where a further increase in rate of sea-level rise decreases productivity and sedimentation, leading to marsh collapse. Thus, the relative elevation of the salt marsh surface is an important variable that affects the productivity of the salt marsh plant community, and that the productivity or biomass density has a positive feedback on the rate of accretion of the marsh surface (Morris et al. 2002). The elevation of the marsh platform relative to mean high water and mean sea level determines inundation frequency, duration and, consequently, wetland productivity. There is a species-specific, optimum elevation and hydroperiod for marsh vegetation and, at superoptimal elevations, an increase in mean sea level relative to the marsh surface will stimulate growth (Morris and Haskin 1990). Sedimentation rate is affected positively by the biomass density of marsh vegetation due to the drag exerted by the plant canopy (Leonard and Luther 1995; Christiansen et al. 2000; Morris et al. 2002). Hence, in response to rising sea level, an increase in biomass density will increase sedimentation rate, thereby raising the elevation of the marsh. Also, with increased plant growth, belowground productivity, which also contributes to elevation change, may increase (McKee et al. 2007). The equilibrium elevation is inversely related to the rate of sea-level rise and subsidence (Morris et al. 2002). That is, when rates of sea-level rise and/or subsidence increase, the equilibrium elevation will

decrease. One implication of this feedback is that a vegetated marsh is only stable when its elevation is superoptimal, because at suboptimal elevations an increase in sea level will depress growth and, thus, sedimentation. Thus, the optimal elevation is also a tipping point.

The model of Morris et al. (2002) provides insight into our findings for the Mediterranean coastal marshes. The highest biomass occurred in marine and riverine marshes, with riverine marshes having the highest belowground biomass. These wetlands receive high mineral sediment inputs as well as high nutrients and are relatively high in the tidal plain. Riverine marshes have long duration flooding during high river discharge but drain well during low discharge. Salinity is low during high discharge resulting in less salt stress. The high mineral sediment input and high productivity result in riverine marshes maintaining a high elevation, and thus these marshes should be stable with expected sea-level rise, as shown by our results. Impounded marshes are isolated from both tidal and river influence and have low biomass and mineral sediment input. Thus, these marshes are losing elevation relative to local water levels. Hypersaline soil salinity in the Rhône and Ebro deltas is an additional stress negatively impacting impounded marshes, resulting in the inability of sufficient plant production to offset elevation loss.

Management of sediment flows to the coastal zone is especially important given recent reports of significant reductions of riverine sediment flux (Syvitski et al. 2005). Day et al. (2008) reported that many coastal wetlands could adjust to climate change, and specifically predicted sea-level rise, but that human impacts in combination with climate change will lower the ability of coastal wetlands to adjust to sea-level rise. This is what we found for the Mediterranean sites. Only those sites that still have riverine input will likely survive predicted acceleration of sea-level rise. More generally, this implies that coastal wetland systems that function naturally will have a greater chance of persisting. This is the idea that system functioning should form the basis for sustainable management of these coastal wetlands (Day et al. 1997). These concepts are reflected in recent European legislation (Integrated River Basin Management and the Water Framework Directive, see Day et al. 2006) that stresses the need for maintaining an ecological balance both in rivers and the coastal areas into which they drain.

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