Inflow and outflow water quality control in coastal aquaculture systems: a case study

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Abstract

Coastal water bodies are a particularly heterogeneous resource, typified by high spatial and temporal variability that could influence the aquaculture in coastal zones. However, the development of coastal aquaculture may produce negative impacts on the coastal area by the potential release of nutrients and organic matter that can be a source of pollution in receiving waters. The aim of this paper was to evaluate the performance of constructed wetland in controlling the dynamics of deoxygenating matter (organic matter and ammonia) and eutrophicating matter [organic matter and soluble reactive phosphorus (SRP)] in the waters entering (inflow) and flowing out (outflow) from a coastal aquaculture fish farm. We observed that constructed wetland systems are effective in removing fractions of total suspended solids, COD, total ammonia nitrogen and SRP contained in the inflow water with higher efficiency in the spring period (60.37%, 14.89%, 65.38% and 17.6% respectively) than in the summer period (45.10%, 8.06%, 32.43% and 8.00% respectively). Similar pattern was recorded for the treatment of the outflow waters, showing that the wetland system reduced most of the deoxygenating and eutrophicating matter produced as a consequence of feeding and fish metabolic activity. During the summer season, high algae mortality can reduce the performance of the wetland system in the outflow water control; this lower efficiency could be improved by controlling the biomass of algae by vegetation harvesting.

Keywords: coastal aquaculture, constructed wetland, water quality management at the intake and at the outfall

Introduction

Aquaculture is recognized as a key factor influencing the quality of water environment; such influences are normally addressed by considering two farm typologies: off-shore aquaculture (see Karacassiss, Tsapakis, Hatziyanni, Papadopoulou & Plaiti 2000; Aguado-Giménéz & García-García 2004; Pergent-martini, Boudouresque, Pasqualini & Pergent 2006) and land-based aquaculture (see Cripps & Bergheim 2000; Hussenot, 2003; Piedrahita 2003). The present paper reports the study results from a case study belonging to an in-between category in Mediterranean area.

Coastal water bodies represent a heterogeneous resource, typified by a high variation in their chemical and physical properties in terms of space and time. These water bodies vary in size, geomorphology and water dynamics following tidal exchange and exhibit a wide salinity gradient, from freshwater ($<0.5\ g\ L^{-1}$) to hypersalinity ($>40\ g\ L^{-1}$) (Greenwood & Wood 2003). As they are subjected to extreme physical and chemical conditions, these environments are more exposed to unfavourable conditions such as low oxygen content than marine or estuarine water (Joyce, Vina-Herbon & Metcalfe 2005). This implies that, in coastal aquaculture farms, some key variables, such as temperature, salinity, dissolved oxygen and concentrations of phosphorus and nitrogen compounds, may create stressful conditions for the fish being cultivated (Boyd & Tucker 1998) and control actions are therefore required.

After feeding and fish metabolic activity, the outflow waters from intensive aquaculture systems could contain a variety of constituents that can negatively impact the natural resources of the surrounding environment, which, in turn, can produce a negative feedback effect on the aquaculture system.

Key constituents include deoxygenating and eutrophicating matter from uneaten feed and excreta. To fulfil environmental protection requirements, great improvements in feed and feeding technologies have been made in the past few years to enhance the food quality by increasing nutrient retention. Nowadays, nitrogen and phosphorous retention ranges between 10–49% and 17–40% respectively. Similarly, nitrogen and phosphorus release in faeces ranges between 3.6–35% and 15–70%, respectively, and dissolved nitrogen and phosphorus excretions between 37–72% and 1–62% respectively (Piedrahita 2003).

In addition to reducing the emission of pollutants, outflow waters from aquaculture can be controlled by applying water treatment technologies that are based on important mechanical and biological processes for removing solids, organic matter, ammonia, nitrite, nitrate and phosphorus. Solids are usually removed by physical processes, including sand and mechanical filters (Kristiansen & Cripps 1996). Biological processes, such as submerged biofilters, trickling filters, rotating biological contactors and fluidized bed reactors, are employed for the oxidation of organic matter and in nitrification and denitrification processes (Van Rijn 1996). The disadvantages of these treatment methods are that they produce sludge and require frequent maintenance. Passage of the outflow waters through constructed wetland systems represents an alternative to this since various biotic and abiotic processes regulate pollutant removal in the wetland (Kadlec & Knight 1996; Reddy & D'Angelo 1997). Microbial mineralization, transformation (e.g., nitrification–denitrification), and phosphorus and nitrogen uptake by the vegetation are the main biotic processes, whereas abiotic processes include chemical precipitation, sedimentation and substrate adsorption. The cost of constructed wetlands is moderate; furthermore, they consume only low amounts of energy and require little maintenance, and provide additional wildlife habitats at the same time (International Water Association 2000). However, there is concern regarding the feasibility of wetlands to become a cost-effective method because wetland typically requires a low hydraulic loading rate (HLR) and a long hydraulic retention time (HRT) to achieve efficient pollutant removal; then, constructed wetlands systems may need a large area when a large amount of aquaculture wastewaters needs to be treated. Previous studies (Schwartz & Boyd 1995; Lin, Jing, Lee & Wang 2002; Tilley, Badrinarayanan, Rosati & Son 2002; Lin, Jing & Lee 2003; Schulz, Gelbrecht & Rennert 2004; Lin, Jing, Lee, Chang, Chen & Shih 2005) have demonstrated that constructed wetland can efficiently remove the pollutants contained in aquaculture wastewaters, including suspended solids, organic matter, nitrogen and phosphorus under an HLR and HRT ranging between 0.017–1.95 m day^{-1} and 0.06–12.8 day respectively.

Constructed wetlands contain macroalgae and emerging macrophytes and can be classified as either free water surface (FWS) or subsurface-flow (SSF) wetlands, according to their HLR and hydraulic residence time (Crites 1994). The time required to stabilize the nutrient removal processes is different for FWS and SSF, the latter being less resilient, since these systems present a greater diversity of nutrient uptake pathways (Brown & Glenn 1999; Brown, Glenn, Fitzsimmons & Smith 1999; Schulz *et al.* 2004) by providing more specific areas for biofilm growth (Lin *et al.* 2002).

To investigate the potential use of a constructed wetland, this paper evaluates the performance of the FWS system in controlling deoxygenating and eutrophicating matter concentrations in the inflow and outflow waters from a coastal aquaculture fish farm.

Materials and methods

Aquaculture system

The study was carried out at a fish farm located in a Ramsar Site near the Orbetello Lagoon (Tuscany, western coast of Italy). The farm yield consists of about 400 t year^{-1} of large-size (1.2–2.0 kg) European sea bass (*Dicentrarchus labrax* L.) in brackish water, obtained by mixing marine waters with waters coming from the surrounding marsh and inflow rivers. For the entire production cycle at this site, four summer seasons are required. The farm comprises two head lagoon ponds, 15 on-growing fish ponds and 11 final-discharge lagoon ponds (Fig. 1). The head lagoon system (HLS) consists of two lagoon ponds, 1.5 m deep, with surfaces of 5 and 10 ha, respectively, receiving the water from three pumps, with a maximum total flow of 3 m^3 s^{-1}. The fish pond volume varied, increasing from 4500 m^3 to 27 500 m^3, the water supply ranged from 0.1 to 0.2 m^3 s^{-1} and the retention time of the wider pond was estimated to be 1.6–3.2 day^{-1}. Stocked fish varied between 2.6 and 4.8 kg fish m^{-3} and were fed with commercially produced pellets (43–47% d.m. protein, 18% d.m. crude fat, 8–9.3% ash, 1.6–1.8% d.m. crude fibre and 1.05–1.25% d.m. phosphorus)

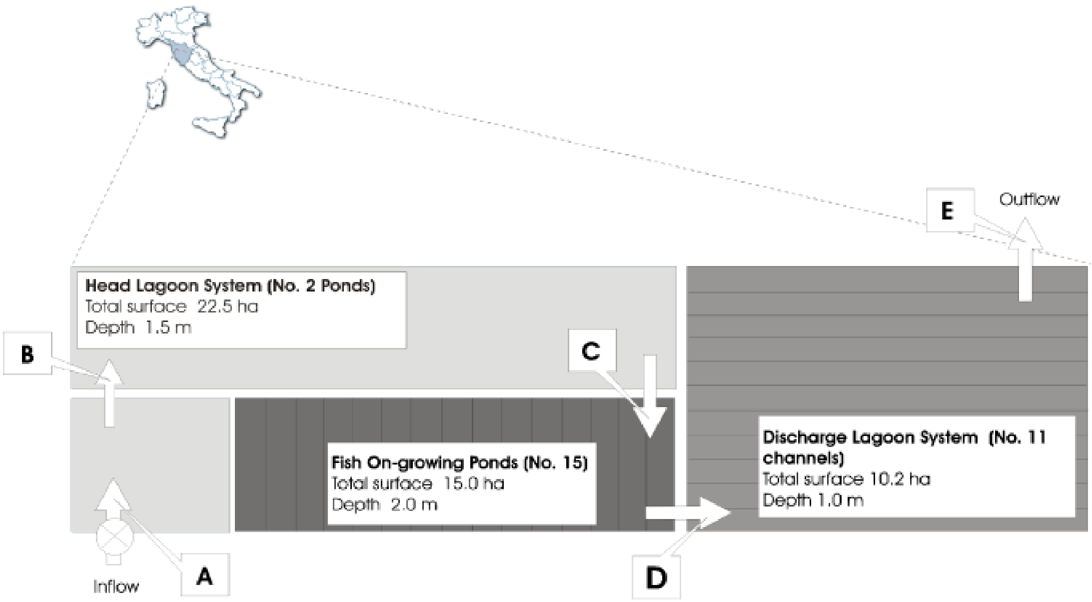

Figure 1 Schematic diagram of the aquaculture system, Tuscany (Italy). (a–e): water sampling points.

produced by Hendrix Skretting®. All the rearing ponds were supplied with pure oxygen by SON® via a distribution network normally regulated to prevent the PO_2 from dropping below 6 mg L^{-1}. To ensure this, up to 10 AquaEco® Forza 7 oxygen delivery machines (1.5 HP each) were in operation in each rearing pond. Waters from the rearing ponds flow into a lagoon system [discharge lagoon system (DLS)] consisting of 11 constructed FWS placed in parallel. Each wetland pond was 9200 m^2 (20 × 460 m) in size and 1 m deep. The HRT for the entire final wetland system were 0.30 and 0.25 (day) for the spring and summer periods respectively. The vegetation (*Potamogeton pectinatus* L., *Chaetomorpha linum* O. F. Müll. and *Gracilaria verrucosa* H.) in the wetlands was established naturally (1.14 kg m^{-2} ww) and resulted in being dominated by *C. linum*. After passing through the final wetland system, the waters flow into the nearby marsh canal.

Sampling and analysis

Key physical and chemical parameters were monitored daily and during 24-h observation in spring and summer 2004. Water flow rate, dissolved oxygen, temperature, pH, total suspended solids (TSS) and salinity were measured three times a day, namely at 19:00, 13:00 p.m. and 17:00 hours, at the monitoring points shown in Fig. 1 for a total of 18 and 11 days in each season respectively. During the 24-h observation monitoring period, which took place in May and July, the samples were taken at 2 h intervals for a 24 h period at the same monitoring points. The following parameters were measured: dissolved oxygen, temperature, pH, TSS, COD, total ammonia nitrogen (TAN), nitrite nitrogen (NO_2-N), nitrate nitrogen (NO_3-N) and soluble reactive phosphorus (SRP).

The nitrogen and phosphorus forms and COD were analysed using photometric methods (Nanocolor, Macherey–Nagel, Düren, Germany) after 0.45 μm (pore size) filtration (Schleicher & Schuell) and dilution (1:1 or 1:2) with deionized water, according to APHA Standard Methods (Eaton, Clesceri, Rice & Grenberg 2005). Dissolved oxygen, pH, temperature and salinity were measured with the Handy Gamma Oxiguard (Aquatrade S.r.l., Macerata, Italy), the pH-meter 250 Aplus (Orion, S.r.l., Padova, Italy) and a manual refractometer (Mod. 106 ACT, PCE S.r.l., Lucca, Italy) respectively; TSS concentrations were optically measured (880 nm) with a portable suspended solid analyzer (Insite Instrument Group, Slidell, LA, USA). The water flow rate (Q) was calculated using the following equation:

$$Q = AV$$

where A is the area across-section and V is the average water velocity measured by a current meter (Mod. ME 4001, Siap, Bologna, Italy).

Mean concentrations and relative standard deviations were calculated for all physical and chemical parameters. Tests for significant differences in water quality between influent and effluent of the wetland units were performed by means of ANOVA. Scheffè's test was applied for *post hoc* comparisons. All analyses were performed by STATISTICA'98 (StatSoft®).

Results

Daily and 24-h sampling

Table 1 shows the results obtained from daily (temperature, oxygen, % oxygen saturation, pH, salinity, TSS) and 24-h sampling (COD, TAN, NO_2-N, NO_3-N and SRP). Table 2 shows Scheffè's test results of comparing the data measured at sites A, B, C, D and E. In spring 2004, the values varied within the following ranges: temperature 16.7–17.2 °C, pH 7.35–7.79 U, oxygen 9.3–11.8 mg L^{-1}, oxygen saturation 98.4–126.6% and salinity 10.0–10.9 g L^{-1}. The highest TSS values were recorded at site A (79.35 ± 23.17 mg L^{-1}), a site directly influenced by water inflow. After the water flowed through the HLS, the TSS mean value decreased to 31.45 ± 9.23 mg L^{-1} (A > C; $P < 0.05$), then increased again after passing through the fish on-growing ponds, with a mean value of 55.00 ± 8.39 mg L^{-1} (C < D; $P < 0.05$). At the end of the DLS, the mean value recorded was 45.90 ± 14.66 mg L^{-1} (D > E and A > E; $P < 0.05$). A similar pattern was recorded for the nitrogen forms and COD. The level of SRP decreased after passage

Table 1 Water qualities (mean ± SD) at sampling location for spring and summer period

Parameters	Spring A	Spring B	Spring C	Spring D	Spring E	Summer A	Summer B	Summer C	Summer D	Summer E
Temperature (°C)*										
Mean	16.7	16.7	16.7	16.9	17.2	25.8	25.9	26.3	26.7	27.6
SD	2.0	1.8	1.7	1.6	2.5	1.7	1.5	1.5	1.3	2.3
Oxygen (mg L^{-1})*										
Mean	9.3	10.1	11.4	10.0	11.8	6.0	6.9	8.8	6.1	7.2
SD	2.2	2.2	2.3	2.3	4.8	1.8	2.1	3.2	1.1	3.5
% Saturation O_2*										
Mean	98.4	107.2	121.4	104.9	126.6	77.0	88.6	115.1	79.0	96.5
SD	24.67	23.81	26.35	23.68	55.37	24.54	30.41	44.63	15.82	49.23
pH*										
Mean	7.49	7.65	7.79	7.35	7.67	7.51	7.55	7.72	7.03	7.40
SD	0.49	0.50	0.51	0.32	0.69	0.31	0.20	0.21	0.10	0.21
Salinity (g L^{-1})*										
Mean	10.4	10.0	10.4	10.8	10.9	35.2	35.3	36.0	36.9	37.2
SD	2.40	1.58	2.03	1.62	1.59	4.56	4.05	2.90	2.12	1.94
TSS (mg L^{-1})*										
Mean	79.35	56.93	31.45	52.59	45.90	47.36	33.91	26.00	39.65	61.56
SD	23.17	13.92	9.23	8.39	14.66	17.30	9.48	9.02	8.34	28.68
COD (mg L^{-1})†										
Mean	28.2	22.0	24.0	24.5	23.4	105.4	93.2	96.9	97.3	104.1
SD	12.5	13.0	14.9	9.5	6.3	23.2	32.3	21.2	41.6	20.2
TAN (mg L^{-1})†										
Mean	0.26	0.15	0.09	0.41	0.25	0.37	0.44	0.25	0.72	0.74
SD	0.19	0.04	0.04	0.05	0.06	0.19	0.11	0.11	0.12	0.11
NO_2–N (mg L^{-1})†										
Mean	0.09	0.15	0.15	0.12	0.12	0.15	0.19	0.16	0.33	0.28
SD	0.01	0.23	0.21	0.01	0.01	0.09	0.04	0.01	0.01	0.02
NO_3–N (mg L^{-1})†										
Mean	0.80	0.75	0.67	0.66	0.66	0.29	0.30	0.31	0.40	0.30
SD	0.68	0.48	0.62	0.60	0.49	0.01	0.03	0.06	0.19	0.03
SRP (mg L^{-1})†										
Mean	0.17	0.15	0.14	0.25	0.30	0.25	0.25	0.23	0.42	0.34
SD	0.07	0.07	0.05	0.08	0.10	0.13	0.05	0.04	0.02	0.05

*Data obtained from the sample taken during daily sampling in spring (n = 55) and summer (n = 33) periods.
†Data obtained from the sample taken during a 24-h observation in spring and summer periods (n = 13).
SRP, soluble reactive phosphorus; TAN, total ammonia nitrogen; TSS, total suspended solids.

Table 2 For evaluating the potential efficiencies of the analytical parameters in the sample points, single comparisons were made and their significance tested by Scheffè's test: site A vs. site C, site C vs. site D, site D vs. site E and site A vs. site E. Only comparisons significant at the 0.05 level are reported

	Spring				Summer			
Parameters	**A vs. C**	**C vs. D**	**D vs. E**	**A vs. E**	**A vs. C**	**C vs. D**	**D vs. E**	**A vs. E**
Temperature (°C)*	–	–	–	–	–	–	–	A < E
	–	–	–	–	–	–	–	$F_{[1, 128]} = 17.5$
	–	–	–	–	–	–	–	$P = 8.9 \times 10^{-4}$
Oxygen (mg L^{-1})*	A < C	–	D < E	A < E	A < C	C > D	–	–
	$F_{[1, 216]} = 12.1$	–	$F_{[1, 216]} = 9.7$	$F_{[1, 216]} = 17.6$	$F_{[1, 128]} = 19.2$	$F_{[1, 128]} = 18.6$	–	–
	$P = 0.008$	–	$P = 0.023$	$P = 7.1 \times 10^{-4}$	$P = 4.3 \times 10^{-4}$	$P = 5.6 \times 10^{-4}$	–	–
% Saturation O_2*	A < C	–	D < E	A < E	A < C	C > D	–	–
	$F_{[1, 216]} = 11.6$	–	$F_{[1, 216]} = 10.2$	$F_{[1, 216]} = 17.4$	$F_{[1, 128]} = 18.1$	$F_{[1, 128]} = 16.3$	–	–
	$P = 0.009$	–	$P = 0.018$	$P = 7.6 \times 10^{-4}$	$P = 7 \times 10^{-4}$	$P = 1.5 \times 10^{-3}$	–	–
pH*	A < C	C > D	D < E	–	A < C	C > D	D < E	–
	$F_{[1, 216]} = 9.1$	$F_{[1, 216]} = 19.9$	$F_{[1, 216]} = 10.5$	–	$F_{[1, 128]} = 15$	$F_{[1, 128]} = 162.9$	$F_{[1, 128]} = 45.7$	–
	$P = 0.029$	$P = 2.5 \times 10^{-4}$	$P = 0.015$	–	$P = 2.5 \times 10^{-3}$	$P = 1 \times 10^{-22}$	$P = 1.5 \times 10^{-8}$	–
Salinity (g L^{-1})*	–	–	–	–	–	–	–	–
TSS (mg L^{-1})*	A > C	C < D	D > E	A > E	A > C	C < D	D < E	A < E
	$F_{[1, 216]} = 278$	$F_{[1, 216]} = 54.1$	$F_{[1, 216]} = 5.4$	$F_{[1, 216]} = 135.5$	$F_{[1, 128]} = 23.6$	$F_{[1, 128]} = 9.6$	$F_{[1, 128]} = 24.8$	$F_{[1, 128]} = 10.4$
	$P = 0$	$P = 1.7 \times 10^{-10}$	$P = 0.020$	$P = 1 \times 10^{-22}$	$P = 7.1 \times 10^{-5}$	$P = 0.025$	$P = 4.3 \times 10^{-5}$	$P = 0.017$
COD (mg L^{-1})†	–	–	–	–	–	–	–	–
TAN (mg L^{-1})†	A > C	C < D	D > E	–	–	C < D	–	A < E
	$F_{[1, 48]} = 31.3$	$F_{[1, 48]} = 57.6$	$F_{[1, 48]} = 14.5$	–	–	$F_{[1, 48]} = 77.6$	–	$F_{[1, 48]} = 50.3$
	$P = 0.003$	$P = 2.5 \times 10^{-8}$	$P = 0.004$	–	–	$P = 4.1 \times 10^{-10}$	–	$P = 1.3 \times 10^{-7}$
NO_2–N (mg L^{-1})†	–	–	–	–	–	C < D	–	A < E
	–	–	–	–	–	$F_{[1, 48]} = 83.6$	–	$F_{[1, 48]} = 43.4$
	–	–	–	–	–	$P = 1.3 \times 10^{-10}$	–	$P = 7.5 \times 10^{-7}$
NO_3–N (mg L^{-1})†	–	–	–	–	–	–	–	–
SRP (mg L^{-1})†	–	C < D	–	A < E	–	C < D	–	A < E
	–	$F_{[1, 48]} = 14.2$	–	$F_{[1, 48]} = 21.8$	–	$F_{[1, 48]} = 42.3$	–	$F_{[1, 48]} = 9.9$
	–	$P = 0.005$	–	$P = 3.9 \times 10^{-4}$	–	$P = 1 \times 10^{-6}$	–	$P = 0.027$

*Data obtained from the sample taken during daily sampling in May ($n = 55$) and July ($n = 33$).

†Data obtained from the sample taken during a 24 h observation ($n = 13$).

SRP, soluble reactive phosphorus; TAN, total ammonia nitrogen; TSS, total suspended solids.

into the HLS, with mean values ranging from 0.17 ± 0.07 to $0.14 \pm 0.05\,mg\,L^{-1}$. In the waters flowing out of the fish on-growing ponds, the SRP concentrations increased again, with a mean value of $0.25 \pm 0.08\,mg\,L^{-1}$ (C < D, $P < 0.05$), but the mean concentrations at the end of the DLS were higher ($0.30 \pm 0.10\,mg\,L^{-1}$) than those measured in the water inflow (A < E; $P < 0.05$).

In summer 2004, the following values were measured at all the sampling sites: temperature 25.8–27.6 °C, pH 7.03–7.55 U, dissolved oxygen 6.0–8.8 $mg\,L^{-1}$, oxygen saturation 77.0–115.1% and salinity 35.2–37.2 $g\,L^{-1}$. The pattern of TSS concentration was similar to that observed in the spring season, with a mean concentration in $mg\,L^{-1}$ 47.36 ± 17.30 (site A), 33.91 ± 9.48 (site B), 26.00 ± 9.02 (site C) and 39.65 ± 8.34 (site D); in comparison A > C, C < D and D < E ($P < 0.05$), respectively, but the highest mean value was recorded at the end of DLS ($61.56 \pm 28.68\,mg\,L^{-1}$, A < E; $P < 0.05$). The levels of COD and TAN at the end of DLS (104.1 ± 20.2 and $0.74 \pm 0.11\,mg\,L^{-1}$) were higher than those at inflow in the same system (97.31 ± 41.56 and $0.72 \pm 0.12\,mg\,L^{-1}$). For TAN, NO_2-N and SRF, the values recorded in the fish farm effluent water in the summer season (0.74 ± 0.11, 0.28 ± 0.02 and $0.34 \pm 0.05\,mg\,L^{-1}$ respectively) were higher than the values recorded in the influent water (0.37 ± 0.19, 0.15 ± 0.09 and $0.25 \pm 0.13\,mg\,L^{-1}$, respectively; A < E; $P < 0.05$).

Wetland performances

The percentage of concentration reduction and the removal rate ($g\,m^{-2}\,day^{-1}$) for key parameters of water quality, in the HLS and DLS, are shown in Table 3. The removal rates were defined as HLR times the difference in concentration between the inflow and outflow waters. The HLR was calculated by dividing the average flow rate ($m^3\,day^{-1}$) by the surface area of the wetland(s). In the HLS, the HLR values were found to be 1.05 $m\,day^{-1}$ during the spring period and 1.30 $m\,day^{-1}$ during the summer period, while in the DLS, the HLR values were 2.49 and 2.98 ($m\,day^{-1}$) in the spring and summer period respectively.

For the HLS, the reduction percentages were higher during spring, with values up to 60.37%, 14.89%, 65.38% and 17.65% for TSS, COD, TAN and SRP respectively. Similarly, the removal rates were higher during the spring period, the highest values being for TSS (57.29 $g\,m^{-2}\,day^{-1}$). In summer the reductions in percentage of TSS, COD, TAN and SRP were 45.10%, 8.06%, 32.43% and 8.00%, respectively, and the highest removal rate was recorded for TSS (27.68 $g\,m^{-2}\,day^{-1}$).

In the DLS, the reduction percentages for TSS, COD and TAN during spring were 12.72%, 4.56% and 39.02%, respectively, whereas negative differences were measured between inflow and outflow SRP concentrations in DLS. A similar pattern was recorded during the summer period for TSS, COD and TAN, but the percentage of concentration reduction was higher for SRP, with a value of 19.05%. Higher removal rates were recorded for TSS and COD in the spring periods, with values of 16.65 and 2.78 $g\,m^{-2}\,day^{-1}$, respectively, whereas in the summer period higher values were recorded for NO_3-N (0.30 $g\,m^{-2}\,day^{-1}$).

Table 4 shows the removal rate constants for key pollutants determined for the constructed wetland system studied and compares them with reference data from the literature. These constant rates were determined using the basic equation of the first-order plug kinetic model:

$$C_o/C_i = \exp(-Kt)$$

where C_i is the influent pollutant concentration ($mg\,L^{-1}$), C_o the effluent pollutant concentration ($mg\,L^{-1}$), t the nominal HRT of the entire wetland unit (day), and K the first-order removal rate constant (day^{-1}). Nominal retention time (HRT) was computed as surface area times water depth times porosity of wetland(s) divided by average flow rate. The porosity or fraction of the space available for water to flow through the wetland in this study was assumed to be 0.75 in accordance with Lin *et al.* (2002). Because average temperature of the influent and effluent of the last wetland system in each season ranged from 14.69 to 17.15 and from 25.32 to 27.61 °C respectively (Table 1), we did not include the temperature effect on each seasonal K and considered the K determined as an apparent reaction rate constant.

All the values of the rate constant estimated in this manner were comparable with those reported in other studies (Table 4).

Discussion

Water temperature is one of the most important variables affecting aquaculture production since the rates of all biochemical processes are temperature dependent. Water temperature also affects the natural

Table 3 Percentage of concentration reduction and removal rate (g m^{-2} day^{-1}) for various parameters of water quality by head and discharge lagoon system throughout the study

	Head lagoon system				Discharge lagoon system			
	Concentration reduction (%)		Removal rate (g m^{-2} day^{-1})		Concentration reduction (%)		Removal rate (g m^{-2} day^{-1})	
	Spring	Summer	Spring	Summer	Spring	Summer	Spring	Summer
TSS*	60.37	45.10	57.29	27.68	12.72	*N*	16.65	*N*
COD†	14.89	8.06	4.41	11.02	4.56	*N*	2.78	*N*
TAN†	65.38	32.43	0.17	0.15	39.02	*N*	0.40	*N*
NO_2–N†	*N*	*N*	*N*	*N*	1.06	15.15	0.003	0.15
NO_3–N†	16.25	*N*	0.14	*N*	0.27	25.00	0.045	0.30
SRP†	17.65	8.00	0.03	0.02	*N*	19.05	*N*	0.24

*Data obtained from the sample taken during daily sampling in spring ($n = 55$) and summer ($n = 33$) periods.
†Data obtained from the sample taken during a 24-h observation in spring and summer periods ($n = 13$).
N, negative value; SRP, soluble reactive phosphorus; TAN, total ammonia nitrogen; TSS, total suspended solids.

Table 4 A summary of average first-order removal rate constant (*K*) from this study and literature for treating aquaculture water and wastewater

	Wetland type	*t* (day)	HLR (m day^{-1})	*K* for TSS (day^{-1})*	*K* for TAN (day^{-1})†	*K* for NO_2–N (day^{-1})†	*K* for SRP (day^{-1})†
This study Spring	FWS	0.300	2.490	0.453	1.648	0.035	*N*
This study Summer	FWS	0.250	2.980	*N*	*N*	0.657	0.845
Lin *et al.* (2005)	FWS–SSF	0.090	1.540	5.950	7.580	17.320	–
Lin *et al.* (2005)	FWS–SSF	0.060	1.950	7.050	11.370	19.600	–
Lin *et al.* (2003)	FWS–SSF	1.31	0.300	1.685	1.115	3.323	0.470
Lin *et al.* (2002)	FWS–SSF	6.50	0.023	0.195	0.372	0.468	–
Lin *et al.* (2002)	FWS–SSF	4.40	0.034	0.291	0.410	0.790	–
Lin *et al.* (2002)	FWS–SSF	2.20	0.068	0.294	0.893	1.489	–
Lin *et al.* (2002)	FWS–SSF	1.10	0.135	0.382	1.167	2.911	–
Tilley *et al.* (2002)	FWS	1.00	0.177	1.050	–	–	–

*Data obtained from the sample taken during daily sampling in spring ($n = 55$) and summer ($n = 33$) periods.
†Data obtained from the sample taken during a 24-h observation in spring and summer periods ($n = 13$).
N, negative value; FWS, free water surface; HLR, hydraulic loading rate; SSF, subsurface flow; SRP, soluble reactive phosphorus; TAN, total ammonia nitrogen; TSS, total suspended solids.

productivity of aquatic ecosystems and directly or indirectly regulates other water quality variables. In this study the temperature was lower in the spring than in the summer period, when the values recorded were similar to the temperature considered to be the 'optimum' for the growth and health of *D. labrax*, which is about 20–22 °C (Cataudella & Bronzi 2001).

The salinity of surface water is influenced primarily by climate and also by the topography and geology of the draining area; in coastal zones, this depends on the relative amounts of fresh river water and seawater that are mixed together. In areas with pronounced wet and dry seasons, salinity shows a great seasonal variation, mainly due to high river discharge rates during the wet season that decrease the salinity. On a shorter scale, salinity at a given point in the coastal zone decreases during ebb tide. In our study the salinity of inflow water in the spring period was 25 g L^{-1} lower than the value recorded in the summer period (Table 1). This variation in salinity is important for osmoregulatory processes. Each species has an optimum salinity range; outside of this range, the fish must expend a considerable amount of energy for osmoregulation at the expense of growth. If salinity deviates too much from the optimum range, the fish will die because it cannot maintain homeostasis. In this study, the highest mean values of salinity were recorded during the summer period (37.2 ± 1.94 g L^{-1}); this mean values were higher than the optimal salinity level and were lower than the tolerated salinity values for *D. labrax* in

Mediterranean aquaculture, reputed to be about 28 g L^{-1} (Conides & Glamuzina 2006) and 45 g L^{-1} (Saroglia & Ingle 1992) respectively.

As for dissolved oxygen, it is to be noted that after the waters flow through the fish on-growing ponds, the oxygen concentration decreases and mechanical aeration is therefore required to maintain values above those stressful or lethal for fish culture.

Coastal water bodies are a particularly heterogeneous resource, typified by high spatial and temporal variability in most parameters of water quality (Joyce *et al.* 2005). In both of the examined seasons, the HLS are effective in removing fractions of TSS, COD, TAN and SRP contained in the inflow waters. This was more efficient in the spring period than in the summer period (60.37%, 14.89%, 65.38%, 17.6% and 45.10%, 8.06%, 32.43% and 8.00% respectively). These results suggest that high chemical precipitation, sedimentation and sufficient nitrification and denitrification processes were concurrently in operation in the HLS wetlands. It is important to control the daily and seasonal variations of the most important water quality parameters to maintain an adequate and satisfactory environment for fish to grow.

The TAN and SRP concentrations recorded in the waters flowing out of the fish on-growing ponds were higher than those measured in the inflow waters in both seasons. The spring levels of these environment compounds were lower than those in the summer because in this season the metabolic activity and the food rations were higher. During the spring month, the increases in TAN and SRP were about 0.32 and 0.11 mg L^{-1}, respectively, whereas a higher increase was recorded in the summer period, 0.47 and 0.19 mg L^{-1} respectively. It has been shown that the nutritional composition of aquaculture effluents depends on various factors related to hydraulic management, oxygen and feeding management (Summerfelt, Adler, Glenn & Kretschmann 1999; Cripps & Bergheim 2000). Reports from different investigations on the nutritional composition of untreated aquaculture effluents showed a wide range: TSS 5–50 mg L^{-1}, SRP 0.06–0.15 mg L^{-1}, TN 0.2–3.3 mg L^{-1} and TAN 0.5–1.1 mg L^{-1} (Bergheim, Sanni, Indrevik & Holland 1993; Ackefors & Enell 1994; Kelly, Bergheim & Hennessy 1994; Dumas, Laliberte, Lessard & De La Noüe 1998; Bergheim & Brinker 2003). In the present study, the level of quality variables examined in untreated aquaculture outflow waters was similar to those quoted in the literature, except for SRP, which showed higher values in both seasons (0.25 and 0.42 mg L^{-1} in the spring and summer periods respectively).

It is essential that nutrients are removed from the waters flowing out of the aquaculture fish farms to protect the receiving water body from eutrophication and therefore for potential reuse of the waters (Lin *et al.* 2002). The nutrient concentrations in the waters flowing out of the DLS indicate that this system can control the aquaculture effluent quality, although a seasonal variability was recorded. In the spring period, the process occurred in DLS: sedimentation, filtration, adsorption, chemical reaction, vegetation uptake, nitrification, denitrification and other microbial processes (Boyd & Tucker, 1998) reduced the levels of TSS, COD, TAN, NO_2-N and NO_3-N. In the summer period, the DLS reduced NO_2-N and NO_3-N levels although increases in TSS, COD and TAN were recorded in the outflow waters from DLS with respect to the inflow site. A possible explanation of these increments could be attributed to the mortality of the vegetation present in the DLS, being dominated by *C. linum*. Taylor, Fletcher and Raven (2001) showed that vegetative growth processes of eight 'green tide' algae, including *C. linum*, were significantly affected by temperature, all species growing at temperatures within the range of 10–20 °C, while temperatures above 25 °C promoted rapid growth over short time periods, but prolonged exposure damaged algae tissue. Taylor *et al.* (2001) also demonstrated that growth of all the examined algae significantly correlated with water salinity from 23.8 up to 27.2 g L^{-1} (70–80% full seawater). Moreover, all the algae tested showed a wide tolerance to salinity: at 3.4–34 g L^{-1} (10–95% full seawater). This result showed that the most limiting factor for algae growth in DLS is temperature since the values measured in the summer period (27.6 ± 2.3 °C) adversely affected the growth of *C. linum*, which is the dominant species found in DLS.

Panella, Cignini, Battilotti, Falcucci, Hull, Milone, Monfrinotti, Pipornetti, Pancioni and Cataudella (1999) used a wetland-pond system, operating at HRT between 2.8 and 3.0 day, to treat and recycle the outflow waters from intensive aquaculture systems. They reported a purifying performance of the wetland-pond system, with removal percentages of 33% (0.69 g m^{-2} day^{-1}) for BOD_5, 14% (0.46 g m^{-2} day^{-1}) for suspended organic solids, 41% (0.015 g m^{-2} day^{-1}) for TAN, 27% (0.419 g m^{-2} day^{-1}) for NO_3-N and 58% (0.015 g m^{-2} day^{-1}) for PO_4-P. This study demonstrated similar removal rates of TSS and TAN in the spring period, while in the summer period removal rates of NO_3-N were similar but those for SRP were lower. In addition, comparison of the *K* values defined in this study with those from the literature showed, in some cases, similar *K* values. These differences in performance may be due to the difference in the diverse operating conditions (including HLR and influent concentration) or pollutant loading rates (defined as HLR multiplied by influent concentrations).

Conclusions

During spring and summer periods, wetland HLS can effectively reduce some important water quality parameters, such as TSS, TAN and SRP, that may influence aquaculture production. The use of a wetland unit does not require any mechanical facilities or energy input.

Furthermore, this study showed that the wetland system reduced most of the deoxygenating and eutrophicating matter contained in the outflow waters. In spite of this positive result, it should be noted that during summer high algae mortality can reduce the performance of the DLS. This lower efficiency could be improved by controlling the biomass of algae by vegetation harvesting.

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