Available online at www.sciencedirect.com
SCIENCE DIRECT®
Marine Pollution Bulletin 46 (2003) 1540–1548
MARINE POLLUTION BULLETIN
www.elsevier.com/locate/marpolbul

Restoration of the eutrophic Orbetello lagoon (Tyrrhenian Sea, Italy): water quality management

Mauro Lenzi [a,*], Roberto Palmieri [b], Salvatore Porrello [c]

[a] *Lagoon Ecology and Aquaculture Laboratory (LEALab), O.P.L. srl, Via Giacomo Leopardi 9, 58015 Orbetello, Italy*
[b] *ARPAT (Tuscany Regional Agency for the Environmental Protection), 50127 Florence, Italy*
[c] *ICRAM (Central Institute for Marine Research), 00166 Rome, Italy*

Abstract

The Orbetello lagoon (Tyrrhenian coast, Italy) receives treated urban and land based fishfarms wastewater. The development of severe eutrophication imposed the three main activity adoption focuses on (1) macroalgae harvesting; (2) pumping of water from the sea; (3) confining wastewater to phytotreatment ponds. The responses to these interventions were rapid and macroalgal reduction growth and seagrass return were recorded. Since 1999, a new macroalgal development was recorded. The aim of this research was to discover whether the recent macroalgal growth can be attributed to the continuing wastewater influx from the remaining persistent anthropic sources (PAS) or from the sediment nutrient release. A monitoring programme was carried out between August 1999 and July 2000 in order to measure dissolved inorganic nitrogen and phosphorus in the wastewaters entering into the lagoon and in central lagoon areas, seaweed and seagrass distribution and lagoon N, P annual budgets. The results showed higher N and P values close to PAS. The distribution of the macroalgal species confirms that the available P comes almost entirely from these remaining PAS. In conclusion, the environmental measures adopted produced a significant reduction in algal biomass development in the lagoon; the macroalgal harvesting activities produced a sediment disturbance with following oxidize conditions, which make P unavailable in the lagoon water, excepting close the PAS.

Keywords: Eutrophication management; Seaweed; Seagrass; Orbetello lagoon; Nitrogen; Phosphorus

1. Introduction

The Orbetello lagoon is located in southern Tuscany (Italian West Coast), between 42°25′ and 42°29′ lat. North and between 11°10′ and 11°17′ long. East, and covers a total area of 25.25 km^2. This lagoon has two communicating basins known as West and East with an area 15.25 and 10.00 km^2 respectively (Travaglia and Lorenzini, 1985), with an 1 m average depth (Fig. 1).

As many other coastal environments (Morand and Briand, 1996), the Orbetello lagoon, has developed a considerable seaweed (macroalgae) proliferation (Lenzi, 1992; Bombelli and Lenzi, 1996). The phenomenon has anthropic origins (urban, aquaculture and agriculture wastewatwer), increased from a developed tourist trade (Lenzi, 1992).

The increase in eutrophication has gradually led to a qualitative and quantitative change from seagrasses (phanerogams) to macroalgae. Various species of opportunistic macroalgae have dominated (Lenzi and Mattei, 1998); macroalgal blooms began to appear in the mid 1960s and have been periodically accompanied by microalgal blooms (Tolomio and Lenzi, 1996). The algal masses, produced almost uninterrupted throughout the year, are moved by the winds and accumulate at high densities (sometimes exceeding 20 kg m^{-2}; Lenzi, unpublished data). During the cold season, the seaweed biomass decomposition and subsequent sulphate reduction processes caused a drastic dissolved oxygen decrease and development of toxic reducing gases, which has led to mortalities of aquatic fauna (Izzo and Hull, 1991). These in turn led to a reduction in the quantity and quality of the fish caught from the lagoon in the 1980s (Lenzi, 1992), and the discoloured water outflow to the adjacent beach areas, causing tourist problem.

For these reasons a basin authority (Orbetello Lagoon Environmental Reclamation Authority, OLERA)

* Corresponding author.
E-mail address: lealab2@hotmail.com (M. Lenzi).

0025-326X/$ - see front matter © 2003 Elsevier Ltd. All rights reserved.
doi:10.1016/S0025-326X(03)00315-1

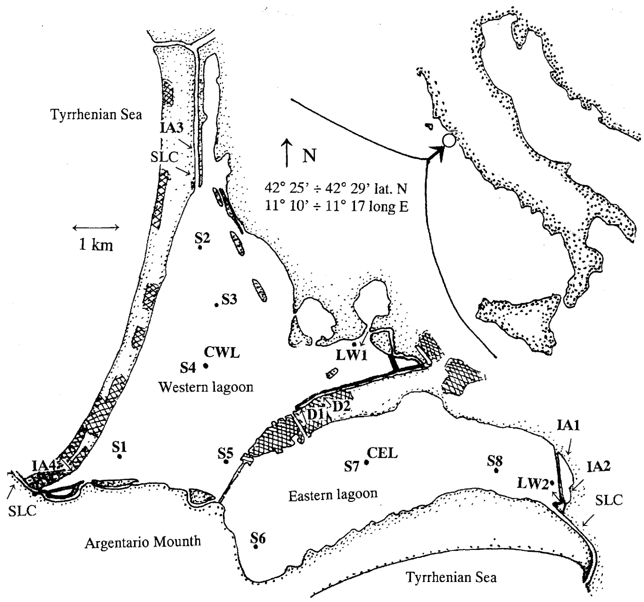

Fig. 1. Orbetello lagoon. Water and algae sampling stations: centre of west lagoon, CWL; centre of east lagoon, CEL; total urban wastewater treatment plant outlet (D1, D2); outlet of water from urban wastewater phytotreatment, LW1; aquaculture effluent outlet, IA1, IA2, IA3, IA4; outlet of IA1 and IA2 effluents phytotreatment, LW2. S1–S8, sediment sampling stations. Sea-lagoon canals (SLC), in which pumping stations are located. Urban centres (graph-paper).

was set up to implement action strategies and solve the environmental crisis. OLERA action followed three main strategies: (1) macroalgal masses removal from the lagoon; (2) increase of clean sea water amount into the lagoon; (3) reduction of anthropic origin nutrient input.

1. Macroalgae harvesting, the first emergency intervention was accomplished by increasing the number and tonnage of the algae harvesting boats.

2. An hydraulic model proposed by Bucci et al. (1989) was adopted to pump water from two sea-lagoon canals into the lagoon, and determine water exited through the third canal to the sea (Fig. 1). In fact, the natural renewal of the lagoon water is extremely low (tidal range around 0.45 m). The pumping was boosted from 8000 to 20,000 $l\,s^{-1}$ and was concentrated in the warmer months. Between 1993 and 2000, 39 complete water lagoon changes renewal was effectuated.

3. All domestic waste waters were collected and pumped into water treatment plant D1 (Fig. 1) and subsequently into a phytotreatment pond (marginal lagoon area) of about 12 ha (Fig. 1). From January 1999, a new treatment plant D2 effluent was discharged into the same phytotreatment area (Fig. 1). Phytotreatment pond wastewater was discharged into the open lagoon. The two largest intensive land based fish farms discharged their effluents into a semi-closed phytotreatment pond (9 ha marginal lagoon area), delimited by an embankment (IA1 and IA2, Fig. 1). During 1998 a leak caused the partial communication with lagoon. The two smallest land based fishfarms (IA3, IA4) instead discharged directly into the lagoon.

OLERA activities produced a significant reduction in algal biomass development in the lagoon. Since 1999 a new macroalgal development has been recorded. The aim of this research was to define whether the recent macroalgal growth can be attributed to the continuing wastewater influx from the persistent anthropic sources (PAS) or nutrient release from the sediment. For this purpose a water, algae and sediments monitoring programme was carried out during the 1999–2000 in the central lagoon areas and near the anthropic sources.

2. Materials and methods

2.1. Water quality

2.1.1. Physical and chemical parameters

Between August 1999 and July 2000, pH, temperature (°C) and dissolved oxygen (expressed as percent of saturation, DO) were determined hourly in two central lagoon stations, one in the west side, CWL, the other in the east side, CEL (Fig. 1), using two multi-parameter probes (HYDROLAB data sonde 3).

2.1.2. Nutrient determinations

Between August 1999 and July 2000, water samples were taken monthly in 9 stations (Fig. 1): central lagoon areas (CEL, CWL); domestic treatment plant outflow (TD = D1 + D2); urban phytotreatment outflow (LW1); land based fishfarms outflow (IA1, IA2, IA3, IA4); eastern fishfarms phytotreatment pond (LW2).

Each sample was filtered through a 0.45 μm (Millipore) membrane, after pre-filtering through a fibreglass filter (Sartorius). The sample was then frozen and maintained at −20 °C until analysed. Nutrients were assayed (three replicates) by an automated multiparameter autoanalyzer (MICROMAC-LAB, Systea srl) according to APHA (1998). Ammonia nitrogen ($N\text{–}NH_4^+$) was determined via Berthelot's reaction. Nitrous nitrogen ($N\text{–}NO_2^-$) was assayed spectrophotometrically after diazotation. Nitric nitrogen ($N\text{–}NO_3^-$) was first reduced to nitrite by passing the sample through an activated cadmium column and then was followed by the diazotation as previously described; the nitrous nitrogen was subtracted from this value. Dissolved inorganic nitrogen (DIN) was obtained as the sum of $N\text{–}NH_4^+$, $N\text{–}NO_2^-$ and $N\text{–}NO_3^-$. Soluble reactive phosphorus (SRP) was determined according to Murphy and Riley (1962). Nitrogen and phosphorus in dissolved and particulated organic matter were not examined.

2.2. Sediment

On a monthly basis throughout the same period, three sediment cores were collected in the west basin

(Stations 1–5) and in the east basin (Stations 6–8) (Fig. 1). The sediment was sampled using a 5 cm diameter core tube. The top layer (0–3 cm) was dried at 75 °C and the dry sediment was passed through a 2 mm sieve before the N and P determinations. The organic nitrogen content was determined by Kjeldhal's method, with a selenium catalyst (APHA, 1975). The phosphorus was analysed according to Aspila et al. (1976).

2.3. Seaweed and seagrass

Macrophyte biomasses were determined during the maximum growth phase. Using a square enclosure (50×50 cm^2), 70 samples were taken at fixed intervals along transects. Constant dry weight for each macroalgal species was obtained by drying in an oven at 75 °C. The total biomass for each species (kg d.w. m^{-2}) was determined according to Elliott (1971). Each species macroalgal standing crops (MSCi; tonnes d.w.) were calculated from biomass considering the total lagoon area (kg d.w. m$^{-2}\times25.25\times10^6$ m^2). The macroalgal total standing crop (MTSC $=\sum$ MSCi) and the weight species dominance percentage ($D\% = \text{MSC}i\times\text{MTSC}^{-1}\times100$) were computed. To determine phanerogams standing crop (FSC), the phanerogams were treated as "photosynthetic biomass" (according to Ott, 1980) and as a single entity. The data of the macroalgal biomass removed referring to the period 1999–2000, expressed as tonnes d.w., were provided by OLERA. Monthly, N and P tissue were determined in red alga *Gracilaria verrucosa* (Huds.) Papenf. (harvested in the CEL, CWL and LW2 stations); in green algae *Ulva rigida* C. Ag. (harvested in the IA2, LW1 and LW2 stations); in *Cladophora vagabunda* C. van D. Hoek (harvested in CWL and in LW1 and LW2 stations); in *Chatomorpha linum* Kutz. (harvested in the CWL, CEL and in the LW1 stations); in seagrasses *Zostera noltii* Hornem., *Ruppia cirrhosa* (Petagna) Grande and *Cymodocea nodosa* (Ucria) Asch. (harvested as a single entity in wide meadow). The samples were washed, cleaned and dried to constant weight in an oven at 75 °C. N and P were determined as already described for the sediment methods.

2.4. Annual nutrient budget

The N and P budget of the lagoon were estimated taking into account the (PAS) and the water, algae and sediment nutrient contents. Nutrient exchange with the atmosphere was not examined. DIN and SRP input, from fish farms (IA1 + IA2 + IA3 + IA4 = TIA), urban treatment plant (TD) and phytotreatment ponds (LW1 and LW2) were computed as annual means (August 1999–July 2000), multiplied by the annual water mass of each source. TIA + TD and LW1 + LW2 + IA3 + IA4 were considered as total anthropic sources (TAS) and persistent anthropic sources (PAS), respectively. The DIN and SRP content in lagoon water mass (BWM) and in yearly renewal water mass (YRWM) were computed from CEL and CWL annual means; the lagoon water mass and the water exchanged annually were respectively 25.25×10^6 and 110×10^6 m^3 (as 30×10^6 m^3, by tidal flow, according to Bucci et al. (1989) and 80×10^6 m^3, by clean sea water pumping, according to OLERA). N and P present in the total lagoon sediment (TLS) were obtained from the annual mean sediment values, reported to total lagoon sediment surface (25.25×10^6 m^2). N and P total contents of annual dry weight biomasses (MTSC, FSC, and seaweed harvested and removed from the system, HM) were estimated from N and P tissue content percentage.

These estimates must be considered imperfect, since only N and P dissolved inorganic compounds (DIN, SRP) were considered. However, DIN and SRP are the components of immediate use from part of the aquatic vegetation.

3. Results

3.1. Water quality

3.1.1. Physical and chemical parameters

Temperature ranged between 6 and 28 °C (17.16 ± 6.91), pH between 7.6 and 9.5 (8.27 ± 0.58), and DO% ranged between 40 and 230 (92.72 ± 44.00). Intense photosynthesis produces marked increases in pH and DO parameters in spring, and decreases in summer.

3.1.2. Nutrient determinations

Figs. 2 and 3 showed the N–NH_4^+, N–NO_3^-, DIN, SRP and DIN:SRP values for the central lagoon areas (CEL, CWL). These stations showed fairly similar trends, i.e. high dissolved inorganic nitrogen content (between 12.0 and 85.1 μM), mainly due to ammonium, which on average represents 86.4% of the DIN for CWL and 82.6% for CEL, while SRP was between 0.1 and 0.9 μM, with average values of 0.38 ± 0.24 (CEL) and 0.3 ± 0.19 (CWL). Overall in May, the CEL station showed SRP values higher respect to CWL (respectively 0.9 ± 0.03 and 0.1 ± 0.03). DIN:SRP atomic ratio, ranging between 40 and 851, was very variable and always very high, both in CEL and CWL, with wide difference between hot and cold period. In CEL, during April–November, this ratio was 71.4 ± 29.5 and during December–March 447.5 ± 273.9; in CWL, during June–November 54.6 ± 11.4 and during December–May 554.5 ± 263.1.

In Figs. 4 and 5, the DIN and SRP trends were reported for the direct and phytotreated sources (TD, LW1, IA1, IA2, IA3, IA4, LW2).

While nitrogen was always available at all stations, phosphorus was abundant only near the PAS stations

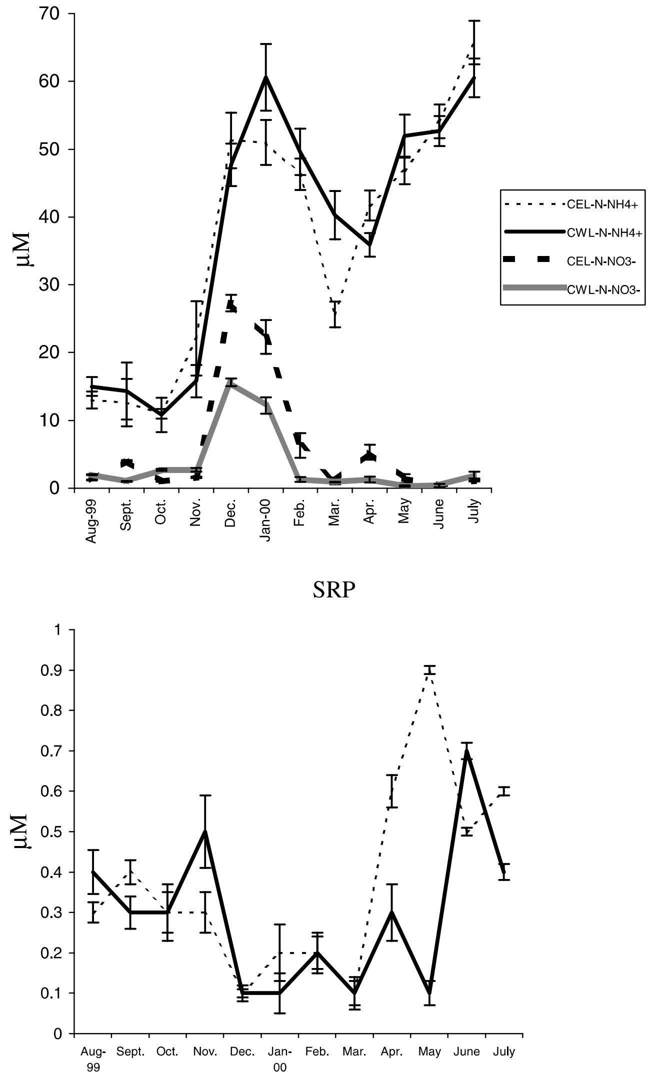

Fig. 2. Means (±SD) of ammonium nitrogen ($N–NH_4^+$), nitric nitrogen ($N–NO_3^-$) and soluble reactive phosphorus (SRP) μM in the central west (— CWL) and central east (--- CEL) basins of the Orbetello lagoon, for 1999–2000.

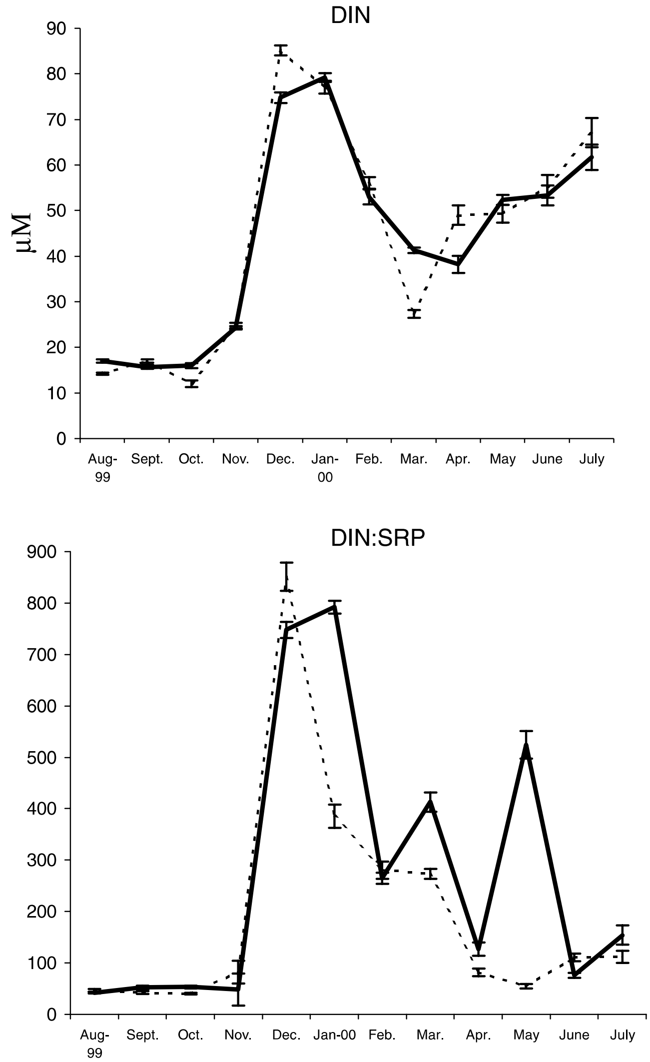

Fig. 3. Means (±SD) of dissolved inorganic nitrogen (DIN, in μM) and DIN:SRP atomic ratios in the central west (— CWL) and east (--- CEL) basins of the Orbetello lagoon, for 1999–2000.

(Figs. 4 and 5). Respect to central areas, DIN:SRP values was found to be substantially lower respect to anthropic sources stations, both direct and phytotreated: 29 ± 21 for LW2, 23 ± 10 for TIA, 16 ± 4 for TD, 13 ± 9 for LW1. The pond of treated urban wastewater (Fig. 4) showed an average nutrient reduction of 85% for DIN and 79% for SRP. Phytotreatment of east basin fishery effluent led to a reduction of nutrient loads of the order of 40% for DIN and 20% for SRP (Fig. 5). Despite the system's efficiency, the residual nutrients in the phytotreated effluents were still highly concentrated respect to central area values (Figs. 2 and 3).

3.2. Sediment

In East basin nitrogen was found to be 0.13 ± 0.02% d.w. and phosphorus 0.08 ± 0.07% d.w. In West basin N and P were 0.45 ± 0.12% and 0.06 ± 0.05% d.w., respectively. With respect to the whole lagoon, N and P mean values (%) and N:P atomic ratio in Table 1 were reported.

3.3. Seaweed and seagrass

Seaweed and seagrass distribution between August 1999 and July 2000 in Figs. 6e and 7d, were reported. Respect to two phytotreatment ponds outflow, *Cladophora* beds appeared close proximity, *C. linum* around here and *G. verrucosa* far-away. *U. rigida* developed mainly inside the confined phytotreatment pond area. Seagrass was distributed over a large part of the lagoon, but did not met near the LW1 and LW2 discharges. In Tables 1–3, respectively the mean N and P tissue contents, the estimated standing crops and the algal biomass harvested were reported. The algal biomass harvested during the study represented 34% of the maximum vegetation growth.

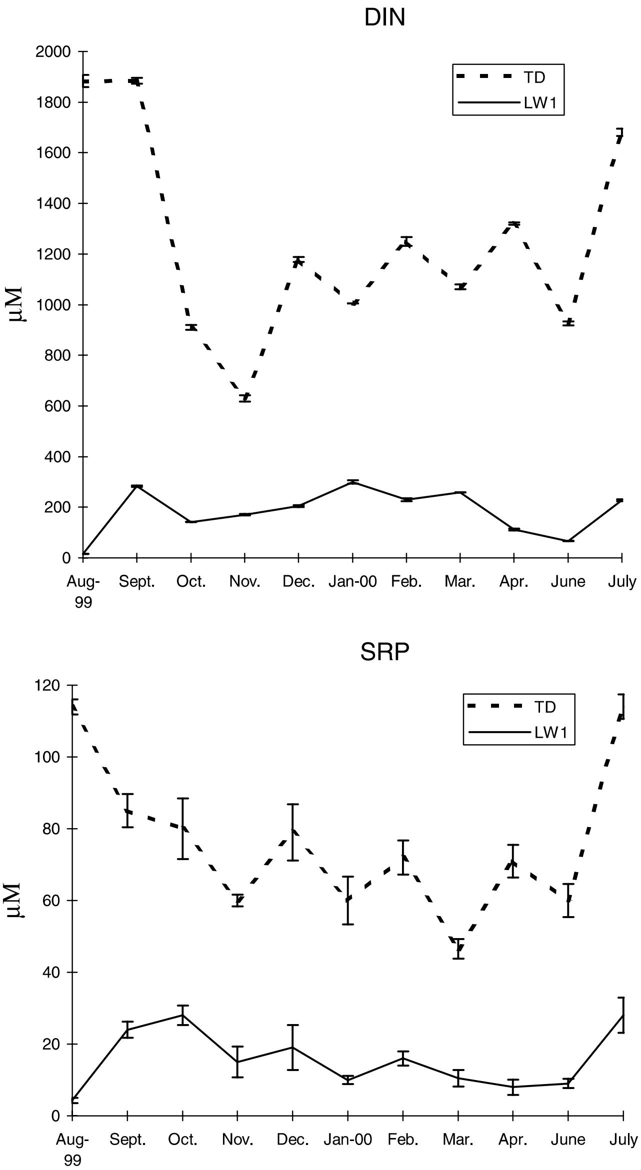

Fig. 4. Means (±SD) of dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP, P–PO_4^{--}) μM, for urban wastewater treatment effluent (TD) and for effluent from the relative phytotreatment area (LW1), for 1999–2000.

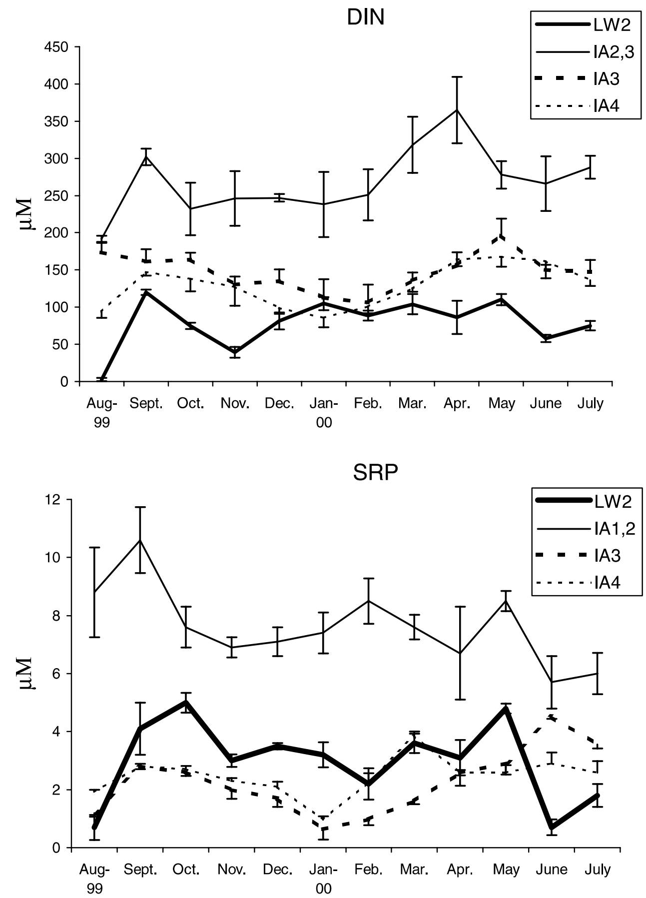

Fig. 5. Means (±SD) of dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP, P–PO_4^{--}) μM, for the effluent from the four intensive aquaculture plants, IA1, IA2, IA3, IA4, and for the effluent from the phytotreatment area of the first two (LW2).

3.4. Annual nutrient budget

In Table 4, nutrient values of each budget component were reported. Sediment (TLS) constituted the greater nutrient store, also just considering the top 3 cm. These values included also the relatively unbioavailable N and P fractions. The annual nutrient budget can be estimated as following:

(1) TAS was equivalent to 164.77 N tonnes and 11.01 P tonnes;
(2) PAS was equivalent to 108.06 N tonnes and 6.60 P tonnes;
(3) TLS stored 13.36 and 30.03 times the N and P PAS loads, respectively;
(4) the components of the ecosystem BMW + YRWM + MTSC + FSC were equivalent to 137.63 N tonnes and 4.13 P tonnes.

Differences between TAS and PAS (57.71 tonnes of N and 4.41 tonnes of P) were equivalent to the annual quantities detained by the two phytotreatments system. In fact, the phytotreatment ponds reduced TAS of 35.02% of N and 40.05% of P.

The renewal of water (YRWM) removed yearly 66.58% N and 27.88% P of the PAS. HM removed from the ecosystem 7.37% N and 3.64% P of the PAS.

4. Discussion

4.1. Water quality

4.1.1. Physical and chemical parameters

Temperature, pH and DO% displayed wide seasonal and daily fluctuations, as is typical of shallow water

Table 1
Percent nitrogen (N) and phosphorus (P) content, atomic ratio N:P and relative standard deviations of dry material from samples of *Chaetomorpha linum* (CH), *C. vagabunda* (C), *G. verrucosa* (G), *U. rigida* (U), phanerogams (SG) and sediment (top 3 cm; SED)

		N (%)	P (%)	N:P
CEL	CH	2.63 ± 1.58	0.061 ± 0.063	73 ± 41
	G	4.14 ± 1.26	0.090 ± 0.030	120 ± 80
CWL	CH	1.73 ± 0.51	0.055 ± 0.028	89 ± 70
	G	3.85 ± 1.07	0.110 ± 0.038	86 ± 41
	C	2.28 ± 0.73	0.096 ± 0.010	52 ± 11
IA2	U	3.47 ± 0.85	0.190 ± 0.050	42 ± 12
LW1	CH	4.01 ± 0.92	0.126 ± 0.040	123 ± 109
	C	3.48 ± 3.00	0.164 ± 0.001	115 ± 88
	U	4.26 ± 1.64	0.139 ± 0.093	71 ± 6
LW2	G	2.38 ± 0.33	0.088 ± 0.017	58 ± 22
	C	2.74 ± 1.04	0.155 ± 0.070	41 ± 15
IA2	U	2.87 ± 1.30	0.121 ± 0.066	61 ± 15
	SG	2.35 ± 0.74	0.107 ± 0.050	53 ± 13
	SED	0.51 ± 0.09	0.068 ± 0.010	26 ± 19

The algae were harvested in central areas CEL and CWL, in urban phytotreatment outflow, LW1, in eastern fishfarms phytotreatment outflow, LW2, inside East phytotreatment close IA2 wastewater. Seagrasses were harvested in central wide prairies.

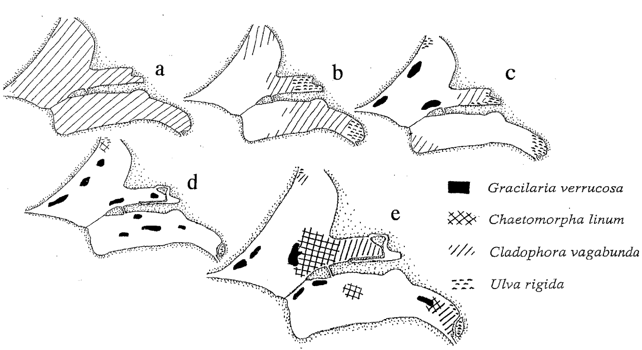

Fig. 6. Distribution of the principal macroalgae in the Orbetello lagoon, between 1993 and 2000: (a) distribution for 1993 (Bombelli and Lenzi, 1996; modified); (b–d) distributions for 1994, 1995, 1997, respectively (Lenzi and Mattei, 1998; modified); (e) distribution for 2000.

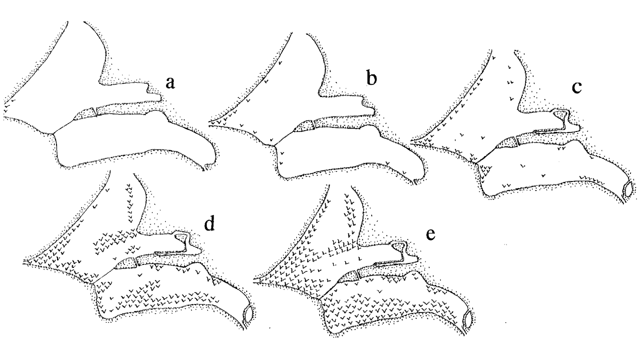

Fig. 7. Phanerogam distribution (vvvv) in the Orbetello lagoon, between 1993 and 2000: (a) 1993 (Bombelli and Lenzi, 1996; modified); (b, c) 1995, 1997, respectively (Lenzi, unpub. data); (d) 1999; (e) 2000.

Table 2
Higher standing crop in tonnes d.w. of seaweed (between 1990 and 2000) and seagrass (between 1998 and 2000)

	Species	Tonne d.w.	*D*%
1990	CH	1782	80
1991	CH	1290	80
1992	C	750	60
1993	C	3000	100
1994	C	480	93
1995	C	348	90
1996	G/U/C	330	54/23/23
1997	G/U/CH	1312	82/17/0,4
1998	G/U	694	98/2
1999	G/U/CH	1049	8/0.4/86
2000	G/U/CH/C	828	27/4/55/12
1998	Seagrass	737	
1999	Seagrass	814	
2000	Seagrass	1087	

For the seaweed (CH, *Chaetomorpha linum*; C, *Cladophora vagabunda*; G, *Gracilaria verrucosa*; U, *Ulva rigida*), the weighted percentage dominance values (*D*%) are given.

Table 3
Results of the algal biomass harvesting carried out from 1993 to 2000

	Species	Tonne d.w.
1993	C	66
1994	C	270
	U	30
	Total	300
1995	C	15
	U	15
	CH	15
	G	128
	Total	173
1996–1997	CH	0.33
	U	0.33
	G	149
	Total	150
1999	CH	209
	U	2
	G	6
	Total	217
2000	CH	183
	C	18
	U	18
	G	63
	Total	282

The quantities removed are expressed in tonnes d.w. CH, *Chaetomorpha linum*; C, *Cladophora vagabunda*; G, *Gracilaria verrucosa*; U, *Ulva rigida*.

environments. On the other hand, DO and pH presented a little better values in 1999–2000 with respect to the first emergency interventions years (1993–1995), when pH ranged between 7.2 and 8.8 (8.17 ± 0.57) and DO% ranged between 5 and 240 (71.5 ± 50.1) (OLERA data). The improving water quality was probably the result of the abundant seagrass growth in recent years, with

Table 4
Annual N and P in kg of each budget component

	N	P
TD	32,586	4352
LW1	5283	901
IA1,2	71,258	4458
IA3,4	60,927	2200
TIA	132,185	6658
LW2	41,854	3496
MSC	23,607	706
FSC	25,549	1163
HM	7961	238
TLS	1,444,212	198,225
BWM	16,516	423
YRWM	71,949	1841

TD, urban treatment plants; LW1, phytotreatment pond of urban treatment plant wastewater; IA1,2, land based fishfarms which discharged into phytotreatment pond 2; IA3,4, land based fish farms which discharged directly into the lagoon; TIA, total fish farms; LW2, phytotreatment pond of fishfarms IA1,2 wastewater; MSC, macroalgal total standing crop; FSC, seagrass standing crop; HM, harvesting macroalgae; TLS, total lagoon sediment (top 3 cm); BWM, lagoon basin water mass; YRWM, yearly renewal water mass.

respect to the past years when the respiratory and decomposition macroalgae processes were dominant.

4.1.2. Nutrient determinations

The nutrient values in Orbetello lagoon central areas, compared to other similar Mediterranean environments, such as Venice lagoon (Sfriso et al., 1992) and Thau lagoon (De Casabianca et al., 1997), were very high for the nitrogen values and very low for the phosphorus values. This result confirms the results reported in Innamorati (1998) and Lenzi et al. (1998). Considering the PAS distance, the nutrient trends recorded in CEL and in CWL were probably regulated by endogenous factors. One reason for the nutrient homogeneity is the pleustophytic algal masses dynamics which were distributed to most parts of the lagoon over the years by the wind (Lenzi et al., 1998).

In eutrophic lagoon shallow waters, ammonium is the major nitrogen compound produced in the anoxic layers (Marty et al., 1990). In the case of the Orbetello lagoon, ammonium formed 86% of the DIN. This high value was the consequence of diffusion from the sediment and was also probably due to ecosystem incapacity to complete nitrification steps.

Above all, in hot periods, the DIN:SRP values recorded in CEL and CWL showed conditions of extreme phosphorus limitation. These values were much higher than 16, considered as optimal for the growth of phytoplankton, and than 35, accepted value for unlimited growth of macroalgae (Atkinson and Smith, 1983).

4.2. Sediment

According to Glodman et al. (1979) and Atkinson and Smith (1983), N:P atomic ratio value (26 ± 19) expressed equilibrium condition. Nevertheless, N:P atomic ratio did not appear related to the high values of the same ratio in seaweed (Table 1), probably because P was more difficult to release from sediment than N. In Venice lagoon sediment, N:P atomic ratio ranged between 7 and 13, almost the same P values of Orbetello, versus thalli N:P of 25 ± 5. In this last case, during macroalgal growth, nitrogen was considered to be the limiting factor (Sfriso et al., 1992).

4.3. Seaweed and seagrass

In the early 1990s only *C. vagabunda* developed to any great extent (Fig. 6a; Bombelli and Lenzi, 1996), and was accompanied by dystrophy, while the phanerogams were almost disappeared (Fig. 7a). In that period, the harvesting of macroalgal biomasses was realised only on a small scale and was not sufficient to control algal growth (Lenzi and Mattei, 1998). With the first OLERA interventions, macroalgal growth decreased (Lenzi and Mattei, 1998), beginning with *C. vagabunda* (Fig. 6b), then *G. verrucosa* in the central areas, and *C. vagabunda* and *U. rigida* in the marginal areas and near the PAS (Fig. 6c and d). Macroalgal biomass harvesting decrease from 1994 and 1997 was described in Table 3 (Lenzi and Mattei, 1998). During the same period seagrass reappeared, first in small clusters (Fig. 7b), and then over a large area (Fig. 7c and d). In July 2000, seagrass covered more than 50% of the lagoon area, including the Orbetello urban area, but remaining distant from the PAS (Fig. 7e). Since 1998, after the embankment leak (LW2) and increase of nutrient discharges (D2 in LW1), the *Cladophora* beds reappeared, together with a large bed of *Chaetomorpha* (Fig. 6e). Consequently, starting in 1999 increased efforts were again made to harvest the algal biomasses (Table 3). These observations were also supported by standing crops estimated from 1990–2000 (Table 2). Table 2 showed the succession in time of the different stages of dominance of the four main algal species. The tendency towards monoxenia of the algal populations confirmed the highly eutrophic nature of the lagoon and the succession of dominance by the different species was probably the result of unstable trophic conditions.

A macroalgal tissue atomic ratio of $N:P < 10$ and $N:P > 30$ expressed an N and P limitation, respectively (Lapointe et al., 1992). An 35 N:P average ratio among the various seaweed species was reported from Atkinson and Smith (1983). In the Orbetello lagoon, N:P ranged between 41 and 123 (Table 1), showing a strong P limitation and presence of nitrophilous species. The large bed of *C. linum* was limited near the urban centre in the CWL (Fig. 6e) tended towards both N and P limitation (Table 1), respect to critical concentrations in the thalli (N 1.2% and P 0.05%), reported in Lavery and McComb (1991). *C. vagabunda* displayed the lowest N:P atomic ratio value recorded. This species was considered to be highly P demanding (critical concentration 0.33%, according to Gordon et al., 1981) and it became more abundant in the zone where P was more readily available (0.164% in LW1 and 0.155% in LW2; Table 1). *C. vagabunda* did not occur in other zones, except in a small quantity in the macroalgal mats of CWL, where P tissue content was 0.096% (Table 1; Fig. 6e). The nitrophilous species *G. verrucosa* was observed in all areas with sufficient N in the water column, in small beds at a considerable PAS distance (Fig. 6e). *U. rigida* grew as a monospecific crop inside the East treatment area. In this station, characterized by very shallow water, intense illumination and high concentrations of DIN and SRP, *U. rigida* accumulated P in amounts up to eight times its critical value and grew in hydrogen sulphide environment and ammonium concentration always higher than 50 μM. The seagrass did not seem to be affected by limitation problems (Table 1), because of its ability to capture the nutrients directly from the sediment through the root system.

In summary, the four opportunistic macroalgae distribution was a function of the nutrient input diffusion from the two main PAS (Fig. 6e) and different DIN:SRP ratios along the distribution areas. The rapid alternation between *C. linum* and *C. vagabunda* in 1991–1992 (Table 2), and, again, between *C. vagabunda* and *G. verrucosa* in 1996–1997 (Fig. 6c and d; Table 2) can be explained in the same terms: the sediment was enriched with organic matter until 1993 and the vegetation dominance tended toward species with a high phosphorus demand; the improvements introduced led to the oxidation of the sediments and, in 1996, an abrupt shift toward a phosphorus limitation tolerant species.

4.4. Annual nutrient budget

Respect to PAS, the HM and YRWM removed as a whole 73.95% of N and 31.52% of P. As SRP and TTP values was low, these results were obtained as consequence of water pump activity and harvesting high effort. Considering only water and vegetation components (points 2 and 4, in Section 3), with respect to PAS contributions, the ecosystem requested 29.57 N tonnes in addition and had an excess of 2.47 P tonnes. Because seagrass is able to capture the nutrients directly from the sediment, 25.55 N tonnes stored in seagrass (FSC, Table 4) was able to balanced the above-said budget for the nitrogen, while the P excess became 3.63 tonnes.

5. Conclusion

Assessment of ecosystem trophic state focused on inorganic nutrient concentration can result inadequate (Cloern, 2001); but ecosystems with oligotrophic water column, can result in eutrophic if the sediment is considered (Dell'Anno et al., 2002). Thus, the Orbetello lagoon can be considered oligotrophic with respect to water column orthophosphate and to macroalgal tissue P content, while showing an high sediment nutrient storage. These conditions were observed in spite of budget results, for which input of PAS were in excess for P respect to the ecosystem demand.

The restoration strategy adopted to eutrophication management of Orbetello lagoon had determined optimal conditions on the sediment: (1) the increased seawater volume pumped in the lagoon ensured faster water flow on rich nutrient sediment layer and on inland lagoon areas, reducing the stagnation processes; (2) since the persistent anthropic sources were confined to the phytotreatment areas, organic matter and nutrient inputs were not spread out in the lagoon; so the macroalgal growth reappeared near persistent anthropic sources when phytotreatment areas were inefficient (break collapsed or sewage excess); (3) the macroalgal harvesting boat activities take away organic matter from the system, constantly disturbing the sediment in less than one meter deep water (Lenzi and Mattei, 1998; Lenzi et al., 1998). This action contributed to the organic matter oxidation and, consequently, to the reduction of organic matter load into the sediment. In fact, the input of organic matter to coastal system is considered the triggering mechanism leading to biodiversity loss and dystrophy (Izzo and Hull, 1991; Cloern, 2001).

All these conditions favoured the onset of new processes of orthophosphate adsorption by the carbonate detritus and clays (Dodge et al., 1984; De Jonge and Villerius, 1989), making them unavailable to the vegetation and favouring the onset of conditions of phosphorus limitation during several years in a large lagoon area (Fig. 2; Lenzi et al., 1998). According to Hines and Lyons (1982) adsorption can be an important cause of P unavailability when the redox potential is high. Conversely, orthophosphate release from the sediment takes place in conditions of organic matter accumulation, when the redox potential is lowered (Fenchel and Riedl, 1970).

The restoration of the Orbetello lagoon showed that organic pollution management in shallow eutrophic areas was possible in a relatively short period of time. Under these conditions macroalgal growth, particularly as far as phosphorus, depended on nutrient amounts originated from persistent anthropic sources.

References

APHA, 1975. Standard Methods for the Examination of Water and Wastewater, 14th ed. APHA, AWWA, WPCF, Washington.

APHA, 1998. Standard Methods for the Examination of Water and Wastewater, 20th ed. APHA, AWWA, WPCF, Washington.

Aspila, K.I., Agemiam, H., Chau, A.S.Y., 1976. A semiautomated method for the determination of inorganic, organic and total phosphate in sediments. Analyst 101, 187–197.

Atkinson, M.J., Smith, S.V., 1983. C:N:P ratios of benthic marine plants. Limnology and Oceanography 28, 568–574.

Bombelli, V., Lenzi, M., 1996. Italy. The Orbetello lagoon and the Tuscany coast. In: Schramm, Nienhuis, (Eds.), Marine Benthic Vegetation. Ecological Studies 123, 331–337.

Bucci, M., Gorelli, V., Vincenzi, D., 1989. Metodologia di intervento per la caratterizzazione idrodinamica della laguna di Orbetello. Attività 1987–1988. RT/PAS/89/18, 10-60, ENEA, Casaccia, Roma.

De Casabianca, M.L., Laugier, T., Collart, D., 1997. Impact of shellfish farming eutrophication on benthic macrophyte communities in the Thau lagoon, France. Aquaculture International 5, 301–314.

Cloern, J.E., 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210, 223–253.

De Jonge, V.N., Villerius, L.A., 1989. Possible role of carbonate dissolution in estuarine phosphate dynamics. Limnology and Oceanography 34, 332–340.

Dell'Anno, A., Mei, M.L., Pusceddu, A., Danovaro, R., 2002. Assessing the trophic state and eutrophication of coastal marine system: a new approach based on the biochemical composition of sediment organic matter. Marine Pollution Bulletin 44, 611–622.

Dodge, R.E., Jickells, T.D., Knap, A.H., Boyd, S., Bak, R.P.M., 1984. Reef-building coral skeletons as chemical pollution (phosphorus) indicator. Marine Pollution Bulletin 15, 178–187.

Elliott, J.M., 1971. Some Methods for the Statistical Analysis of Samples of Benthic Invertebrates, Science Publication No. 15 Freshwater Biological Association, Windermere, UK. 144 pp.

Fenchel, T.M., Riedl, R.J., 1970. The sulfide system: a new biotic community underneath the oxidized layer of marine sand bottoms. Marine Biology 7, 255–268.

Glodman, J.C., McCarthy, J.J., Peavey, D.G., 1979. Growth rate influence on the chemical composition of phytoplankton in oceanic waters. Nature (Lond.) 279, 210–215.

Gordon, D.M., Birch, P.B., McComb, A.J., 1981. Effects of inorganic phosphorus and nitrogen on the growth of an estuarine *Cladophora* in culture. Botanica Marina 24, 93–106.

Hines, M., Lyons, W.B., 1982. Biogeochemistry of nearshore Bermuda sediments. I. Sulphate reduction rates and nutrient generation. Marine Ecology Progress Series 8, 87–95.

Innamorati, M., 1998. La laguna di Orbetello: risanamento e gestione conservativa. Biologia Marina Mediterranea 5 (3), 1352–1361.

Izzo, G., Hull, V., 1991. The anoxic crises in dystrophic processes of coastal lagoon: an energetic explanation. In: Rossi, C., Tiezzi, E. (Eds.), Ecological Physical Chemistry. Elsevier, Amsterdam, pp. 559–572.

Lapointe, B.E., Little, M.M., Little, D.S., 1992. Nutrient availability to marine macroalgae in siliciclastic versus carbonate-rich coastal waters. Estuaries 15 (1), 75–82.

Lavery, P.S., McComb, A.J., 1991. The nutritional ecophysiology of *Chaetomorpha linum* and *Ulva rigida* in Peel Inlet, Western Australia. Botanica Marina 34, 251–260.

Lenzi, M., 1992. Experiences for the management of Orbetello lagoon: eutrophication and fishing. Science of the Total Environment 5 (suppl.), 1189–1198.

Lenzi, M., Mattei, N., 1998. Risultati di quattro anni di gestione dell'ecosistema lagunare di Orbetello attraverso la raccolta delle biomasse algali. Biologi Italiani XXVIII (2), 7–12.

Lenzi, M., Costa, A., Giannerini, S., 1998. Variazioni di dominanza nei popolamenti algali della laguna di Orbetello. Un'ipotesi che ne individua le cause. Atti della Società toscana di Scienze Natuali, Memorie, Serie B 105, 35–43.

Marty, D., Esnault, S., Caumette, P., Ranaivoson-Rambeloarisoa, E., Bertrand, J.C., 1990. Denitrification, sulfato-reduction et methanogenese dans les sediments superficiels d'un étang saumatre méditeranéem. Oceanologica Acta 13, 199–210.

Morand, P., Briand, X., 1996. Excessive growth of macroalgae: a symptom of environmental disturbance. Botanica Marina 39, 491–516.

Murphy, J., Riley, J.P., 1962. A modified simple solution method for the determination of phosphate in natural waters. Analytica Chimica Acta 27, 31–36.

Ott, J.A., 1980. Growth and production in *Posidonia oceanica* (L.) Delile. P.S.Z.N.I. Marine Ecology 1, 47–64.

Sfriso, A., Pavoni, B., Marcomini, A., Orio, A., 1992. Macroalgae, nutrient cycles, and pollutants in the lagoon of Venice. Estuaries 15 (4), 517–528.

Tolomio, C., Lenzi, M., 1996. ''Eaux colorées'' dans les lagunes d'Orrbetello et de Burano (Mer Tyrrhénienne du Nord) de 1986–1989. Vie Milieu 46 (1), 25–37.

Travaglia, C., Lorenzini, M., 1985. Monitoring algae growth by digital analysis of LANDSAT data: the Orbetello lagoon case study. In: RSC Series. FAO, Rome. 19 pp.