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Environmental Characterization of a Eutrophicated Semi-Enclosed System: Nutrient Budget (Encerrada Bay, Tierra del Fuego Island, Patagonia, Argentina)

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Abstract Encerrada Bay (EB) is located in the far south of Argentina, on the north coast of the Beagle Channel and is artificially connected with Ushuaia Bay (UB). This study was carried out between 2004 and 2005; and assesses the impact of Ushuaia city to the nutrient dynamics in EB. It was focused on physical and chemical characterization of discharges, water and sediment quality, nutrient benthic fluxes, and water exchange with UB. The average ammonium, nitrate, phosphate and silicate concentrations in the water bay were 99.5±30.7; 10.0±4.2; 2.0±0.7; 23.5±2.9 μM, respectively. Benthic fluxes showed a consumption of oxygen (50–450 mg $m^{-2}h^{-1}$) and nitrate (20–416 μmol m^{-2} h^{-1}) by sediment and release of ammonium (79−4,772 μmol m^{-2} h^{-1}) and phosphate (27–36 μmol m^{-2} h^{-1}) into the water column. The daily contributions of nitrogen and phosphate from the effluents to EB were between 102 and 517 kg day^{-1} and between 4 and 22 kg day^{-1} respectively, while the net average export fluxes to UB were 41.7 kg day^{-1} of nitrogen and 15.7 kg day^{-1} of phosphate. The difference between received and exported nutrients is consumed in EB by primary producers, partially buffering the impact of wastewater in UB at its own eutrophication risk.

Keywords Coastal pollution · Eutrophication · Nutrients · Pore water · Sediment–water exchanges · Sewage

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1 Introduction

The coastal zones constitute ecosystems which are generally subjected to antrophic pressure, being estuaries and shallow environments with low water renewal, the most susceptible to be impacted. Sewage outflow can generate nutrient enrichment and promote cultural eutrophication processes (GESAMP. 1990; Vollenweider 1992), leading to a deterioration of water and sediment quality, together with biodiversity decrease. Water turbidity and micro and macroalgae production are increased, while the superficial sediment may become anoxic due to excess organic matter. Its degradation consumes dissolved oxygen (and other electron acceptors when it is absent), liberating dissolved nutrients. Due to that, the benthos plays an essential role in the control of the trophic state (Clavero et al. 2000) as well as in the biogeochemical cycles of coastal marine environments (Christensen et al. 2000; Niencheski and Jahnke 2002; Sakamaki et al. 2006).

Considering the particular cases of Ushuaia Bay (UB), Golondrina Bay (GB), and Encerrada Bay (EB), these three systems receive the impact of Ushuaia City through urban and industrial effluent that reaches the waterline of the bays by runoff, streams or rivers (Fig. 1a). The purpose of this study was to assess the potential impact of sewage from Ushuaia city on nutrients dynamics of EB.

2 Study Area

Ushuaia City, located at 54°49′S and 68°19′W, extends along the EB–UB–GB system. Its thermal amplitude is estimated in nearly 21°C, with variations from -7°C (June and July) to 14°C (January and February). Even if annual precipitation is near 550 mm (distributed along the year), daily peaks can reach 40 mm (Iturraspe et al. 1989), generating sudden contributions of freshwater and suspended material in coastal waters. It may snow between the months of May and August, with a monthly average precipitation estimated in 23 mm (Iturraspe et al. 1989). January and February are the months with the greatest southwest wind speeds, with an annual average speed of 14 km h^{-1}, and a maximum of 60 km h^{-1}. The observed semi-diurnal tide amplitude is of 1.20 m (SHN 2008).

According to the 2001 National Census (INDEC. 2008), the population of Ushuaia is about 46000 inhabitants; however it had an explosive growth during the last few years. Tourists during the summer months (January and February) can duplicate the permanent population. The domestic, industrial, and harbor activities have caused significant alterations to the natural environment for over 20 years. Installed sewage network service is insufficient to meet the demand, especially concerning pluvial outflow channels and urban and industrial effluent treatment. The sewage network collects domestic water in only one part of the city and it is discharged to the sea through underwater pipeline in GB. The remaining sewage arrives at EB and UB through pluvials, streams, and rivers.

EB in particular, is a semi-closed system (Fig. 1b), which receives three sewage (Onas, Beban, and Guaraní) and the discharge of Buena Esperanza Stream (BES; which transports thaw water from Martial glacier as well as runoff and wastewater from Ushuaia city). BES has a medium flow of ~0.3 m^3 sec^{-1} although it can reach up to 3.0 m^3 sec^{-1} during thaw seasons (Iturraspe pers. comm.). The flow of the other effluents was estimated in less than 0.01 m^3 sec^{-1}. BE has a total area of 0.27 km^2 and a medium depth of 0.80 m. It is separated from UB by an artificial pathway, and they communicate through two vents (P1 and P2 of 4.5 m long each), hence allowing water exchange in each tide cycle.

3 Materials and Methods

3.1 Characterization of External Contributions

External outflows samples: Buena Esperanza stream, Onas, Guaraní, and Beban (Fig. 1b) were collected in October 2004 (spring), December 2004 (spring–summer), March 2005 (summer–fall) and September 2005 (winter–spring). In situ temperature (T) and dissolved oxygen (DO) were measured using a multi-probe sensor (YSI model 58). Conductivity, pH, total suspended solids (TSS), total evaporable waste (TEW), biochemical oxygen demand to 5 days (BOD_5) and nutrients (ammonium: NH_4^+, nitrates+nitrite expressed as: NO_3^-, phosphate: PO_4^{3-}, and silicate: SiO_3^{2-}) were analyzed in laboratory by standard methods (APHA 1980). Bacteriological analyses were carried out using the Colilert® method (Yakub et al. 2002).

3.2 Water and Sediment Quality in Encerrada Bay

The water and sediment samples were collected at eight stations in EB at the same time as those corresponding to sewages samples. Water T, pH, salinity (S), and DO were measured on site using a multi-probe sensor (Horiba), while NH_4^+, NO_3^-, PO_4^{3-}, SiO_3^{2-}, chlorophyll-*a* (chl-a) and phaeopigments were determined in the laboratory (Strickland and Parsons 1972).

Sediment was collected with a Van-Been dredge. Redox potential (E_h) was measured immediately with a Pt- electrode (Altronix TPA-1). Physical and chemical parameters were determined in the laboratory: density (wet weight/sediment volume ratio), porosity (by drying the sediment samples at 90°C until reaching constant weight), organic matter (OM; by dry combustion at 450°C for 4 h), granulometry (dry sifting process), and total NH_4^+ (Kemp et al. 1990). NH_4^+ and

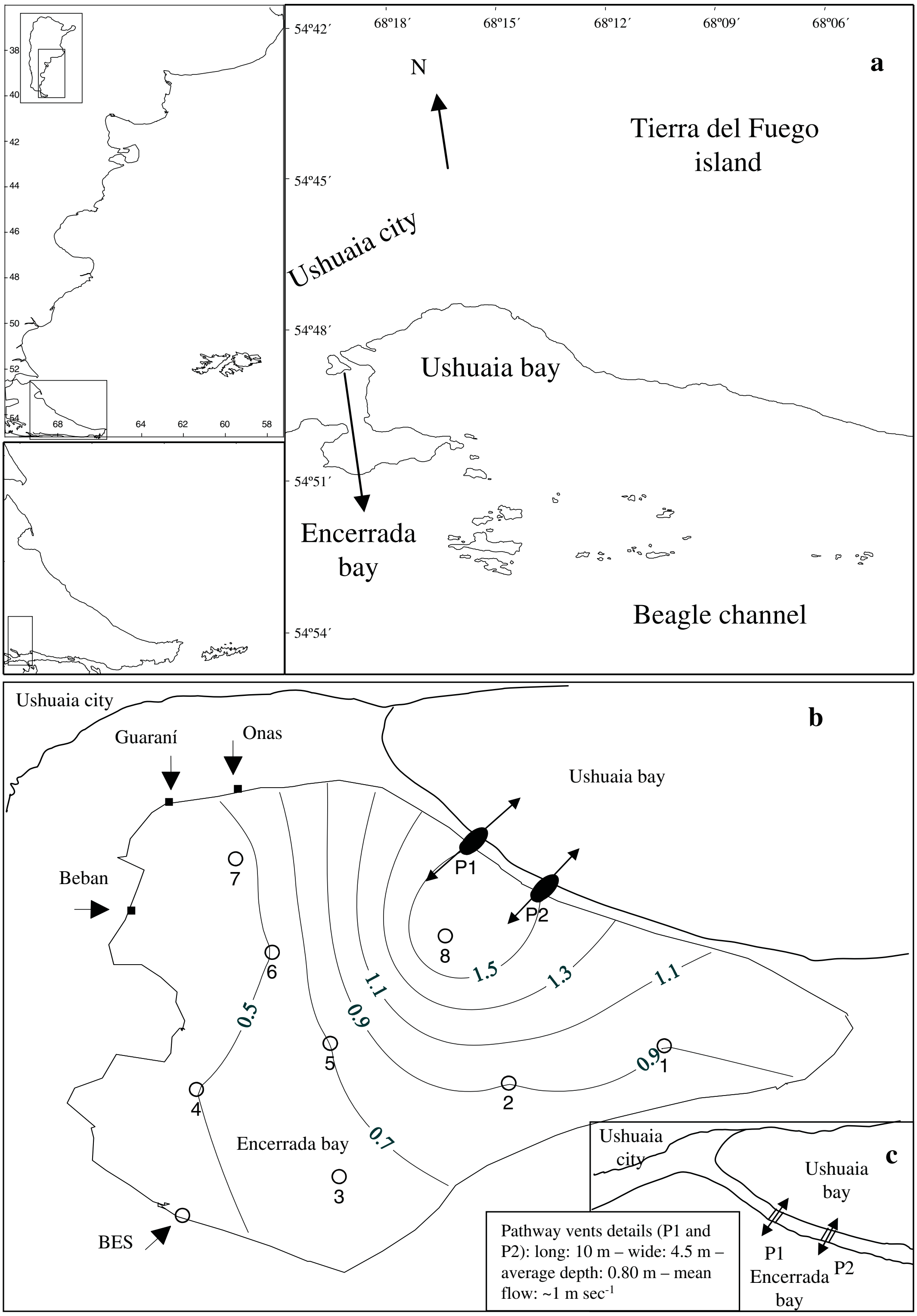

Fig. 1 **a** Map of the study area. **b** Position of stations and outfalls (Buena Esperanza Stream (*BES*) and runoff: *Onas*, *Guaraní*, and *Beban*) isobaths and vents (*P1* and *P2*) that communicate to Encerrada and Ushuaia Bays. **c** Pathway vents details

NO_3^- concentrations were determined in pore water after extraction with a high pressure filter (Sartorius®) and N_2 injection (Strickland and Parsons 1972). The adsorbed NH_4^+ concentration was obtained from the difference between the total and the pore water one.

3.3 Nutrient Flux Across the Sediment–Water Interface

Benthic fluxes were measured at stations 1 and 3 (Fig. 1b). An opaque PVC chamber was used (Esteves et al. 1986) with a closed volume of 9.70 l and a surface of 0.09 m^2. The water inside the chamber was homogenized by means of a magnetic stirring bar, in order to avoid the formation of concentration gradients. Temperature and DO were registered in the closed volume every 5 min (YSI model 58). In order to analyze dissolved nutrients, water samples (60 ml) were aspirated through a syringe from the inside of the chamber every 30 min, by means of 2-mm internal diameter tubing. The experiences lasted 2 h and were carried out by triplicate. After obtaining the concentrations, the magnitude and direction of benthic fluxes were calculated using the following equation:

$$F_b = (X - X_0) \times V_c / S_c \times t$$

Where F_b is the solute flux across the sediment–water interface (μmol m^{-2} h^{-1}); X_0 and X are the solute concentrations in the chamber at start and "t" time, respectively; V_c is the water volume in the chamber, and S_c is the sediment surface area covered by it. Positive and negative values of F_b correspond to fluxes out of sediments and into sediments, respectively.

3.4 Nutrient Exchange Between Encerrada and Ushuaia Bays

By using one of the pathway vents, water samples were collected as it was flowing out of EB at low tide, as well as that entering during high tide. Water height in the vent, pH, T, conductivity, and DO were measured every 15 min in a whole tide cycle in November 2005 and samples were collected for analysis of NH_4^+, NO_3^-, and PO_4^{3-}. Daily nutrients import and export mass fluxes from and to UB, respectively, were estimated in a tide cycle through the following equations:

$$F_i = \sum M_{x-i} \text{ and } F_e = \sum M_{x-e}$$

Where: $M_{x-i}(\mu\text{mol}) = C_x\left[\mu\text{mol} \cdot \text{l}^{-1}\right] \times \Delta v_i(\text{cm}^3) \times 0.001$

M_{x-i}	is the imported mass of nutrients from UB at time interval (Δt=15 min) at high tide.
C_x	is average concentration of the nutrient in the time interval Δt.
$\Delta v_i = \Delta H(\text{cm}) \times A(\text{cm}^2)$	is the volume of water imported from UB in each time interval Δt.
ΔH	is absolute value of height difference in the water column measured in the pathway vent during Δt.
A	EB area
M_{x-e}	is the exported mass of nutrient towards UB at time interval (Δt=15 min) at low tide.

$$M_{x-e}(\mu\text{mol}) = C_x\left[\mu\text{mol} \cdot \text{l}^{-1}\right] \cdot \Delta v_e\left(\text{cm}^3\right) \times 0.001$$

$\Delta v_e = \Delta H(\text{cm}) \times A(\text{cm}^2)$	is the volume of water exported towards UB in each time interval Δt.

Exchanged mass flux in a tide cycle: $F = F_e - F_i F>0$ implies a net contribution from EB to UB, $F<0$ implies a net contribution from UB to EB

3.5 Statistical Analysis

Comparative analyses of benthic fluxes were carried out by means of a non-parametric test (Kruskal–Wallis by ranking $p<0.05$) after verifying the absence of homoscedascity. By means of principal component analyses (PCA), sampling date and site in the Bay was evaluated. In all the cases, InfoStat (2008) statistical package was used.

4 Results and Discussions

4.1 Characterization of External Contributions

The physical, chemical, and microbiological characteristics of the four outflows are shown in Table 1.

Table 1 Characterization of the discharges in EB during studied periods

Effluent	BES	Onas	Guaraní	Beban	Metcalf and Eddy 1991 (weak-medium)
T(°C)	4.0–10.2	10.4–14.1	10.7–13.4	9.6–12.4	n/d
pH	6.9–7.2	7.3–8.3	7.4–8.4	7.1–8.3	n/d
DO (mg l^{-1})	7.8–11.2	9.1–11.0	7.1–8.1	8.4–12.6	n/d
TSS (g l^{-1})	0–2.3	0.1–2.5	0.1–2.7	0.04–1.95	0.1–.2
TEW (g l^{-1})	0.1–5.2	16.3–27.5	1.1–11.7	0.31–8.69	0.1–.2
Conductivity (μmhos)	150–832	2,976–30438	1,211–4,696	1,275–3,384	n/d
NH_4^+ (μM)	263.1–1,313.0	13.3–1,485.6	32.3–477.0	54.7–85.7	666.7–1388.9
NO_3^- (μM)	14.3–36.1	11.5–88.6	31.4–77.0	16.7–61.9	0
PO_4^{3-} (μM)	4.6–24.8	0.4–5.7	9.9–27.5	2.3–41.5	96.9–161.4
SiO_3^{2-} (μM)	22.3–156.0	8.4–34.3	44.6–183.6	21.7–199.0	n/d
BOD_5 (mg l^{-1})	5.4–80.0	<30	<30	<30	110–220
Total coliforms/100 ml (NMP)	6.59×10^5–2.01×10^6	1.00×10^4–2.00×10^6	1.00×10^6–5.30×10^6	2.00×10^6–8.70×10^6	10×10^6–10×10^8
Fecal coliforms/100 ml (NMP)	1.64×10^5–6.24×10^5	7.38×10^3–6.97×10^5	6.97×10^5–8.85×10^5	5.31×10^5–5.60×10^5	10×10^4–10×10^5
NH_4^+ (kg day^{-1})	122.8–612.6	0.2–23.1	0.5–7.4	0.8–1.3	n/d
NO_3^- (kg day^{-1})	23.0–58.0	0.6–4.8	1.7–4.2	0.9–3.3	n/d
PO_4^{3-} (kg day^{-1})	11.3–61.1	0–0.5	0.9–2.3	0.2–3.4	n/d

Obtained range (n=4) by each parameter is showed. In the last column, typical values of non treatment sewage according to Metcalf and Eddy (1991) are indicated. Range of variation of dissolved inorganic nitrogen and phosphate discharged in Encerrada bay. In the last rows the load contributed by the sewage are indicated. TSS is total suspended solids. TEW is total evaporable waste

n/d no date

The highest thermal amplitude and the lowest *T* were observed in the BES (range, 4–10.2°C), while in the others, *T* varied from 9.6 to 14.1°C. The pH was neutral or slightly alkaline in all the cases and DO remained close to saturation. The presence of TSS and TEW could be associated mainly to the transport of dissolved and particulated solids from the continent. Conductivity presented typical levels of freshwater and was correlated positively with the TEW (R^2=0.98). BOD_5 was not detected or low (for the three minor effluents, but values of up to 80 mg l^{-1} were measured in the BES); NH_4^+ presented maximum values typical of weak or medium sewage (Metcalf and Eddy 1991) in BES and Onas. The simultaneous presence of low BOD_5 and high NH_4^+ and DO concentrations suggests the development of aerobic degradation processes of the OM being transported. Even when the presence of NO_3^- indicates the existence of nitrification processes, the average nitrate concentration was around two orders of magnitude lower than those of NH_4^+. The minimum nutrient concentration in BES was measured in December (coincident with the thaw period) and the maximum concentration was measured in September. The mass fluxes of each nutrient were higher in BES than in the other effluents (Table 1) due to the difference between flows (0.03 m^3 seg^{-1} vs. 0.01 m^3 seg^{-1}, respectively). Total and fecal coliforms were detected in all the effluents with typical values of weak sewage. The highest fecal coliforms level was found in the Guaraní effluent and the lowest in the Onas.

4.2 Water Quality in Encerrada Bay

Table 2 summarizes the results obtained for the environmental parameters measured in EB waters. Temperature followed a seasonal pattern and ranged from 4.6°C (September) to 12.9°C (December). Salinity ranged from 0.4 to 26.0 (psu), with a weak horizontal gradient towards the vents. The pH was neutral or slightly alkaline and the DO remained close to saturation. Nutrients, chl-a, and phaeopigments concentration were found within the range of values cited for eutrophicated coastal environments (Kontas et al. 2004; Saunders et al. 2007). The phaeopigments/chl-a ratio was always greater than 1 with the exception of December, typical of more degraded systems (Pinto et al. 2001). Figure 2a shows the

Table 2 Water characterization of Encerrada bay during studied period (2004–2005)

Sampling date	T (°C)	S (psu)	pH	D.O. (mg l^{-1})	NH_4^+ (μM)	NO_3^- (μM)	PO_4^{3-} (μM)	SiO_3^{2-} (μM)	Chlor.-*a* (mg m^{-3})	Phaeopigments (mg m^{-3})
Oct. 2004	7.3	17.6	7.4	9.3	97.7	13.4	1.5	14.8	22.1	54.5
	7.3	17.6	7.4	9.3	99.5	11.5	1.5	13.3	20.3	40.7
	(7.2–7.4)	(11.4–22.8)	(6.9–7.8)	(9.1–9.5)	(12.0–179.9)	(9.2–21.1)	(0.3–2.9)	(96–25.7)	(7.3–43.8)	(12.1–114.3)
Dec. 2004	12.3	17.6	7.9	10.5	69.0	4.0	3.2	20.3	3.0	1.9
	12.4	16.6	7.9	10.4	47.0	3.8	2.3	23.5	2.5	1.9
	(11.7–12.9)	(10.4–24.1)	(7.7–8.1)	(9.7–11.6)	(23.6–183.6)	(2.3–6.2)	(1.6–7.7)	(6.4–33.2)	(1.0–5.4)	(0.9–3.5)
Mar. 2005	9.2	8.2	7.2	8.5	119.0	9.5	2.3	41.8	0.7	2.2
	9.2	6.2	7.3	8.4	102.1	9.5	1.7	37.5	0.6	2.2
	(8.6–98)	(0.4–18.9)	(6.6–7.4)	(8.1–9.0)	(48.3–238.6)	(7.1–10.7)	(0.9–4.5)	(11.0–75.4)	(0.4–1.3)	(1.2–3.3)
Sep. 2005	4.8	24.9	8.4	10.7	320.8	40.7	7.1	25.9	91.3	162.4
	4.8	24.8	8.5	10.8	287.9	30.0	4.7	23.2	46.1	48.9
	(4.6–5.1)	(24.3–26.0)	(8.1–8.6)	(8.3–12.8)	(8.7–870.9)	(9.6–134.1)	(0.1–21.6)	(11.3–41.4)	(5.8–394.9)	(7.2–761.1)

For eight sampling sites, average, median and range of variation among parenthesis are presented

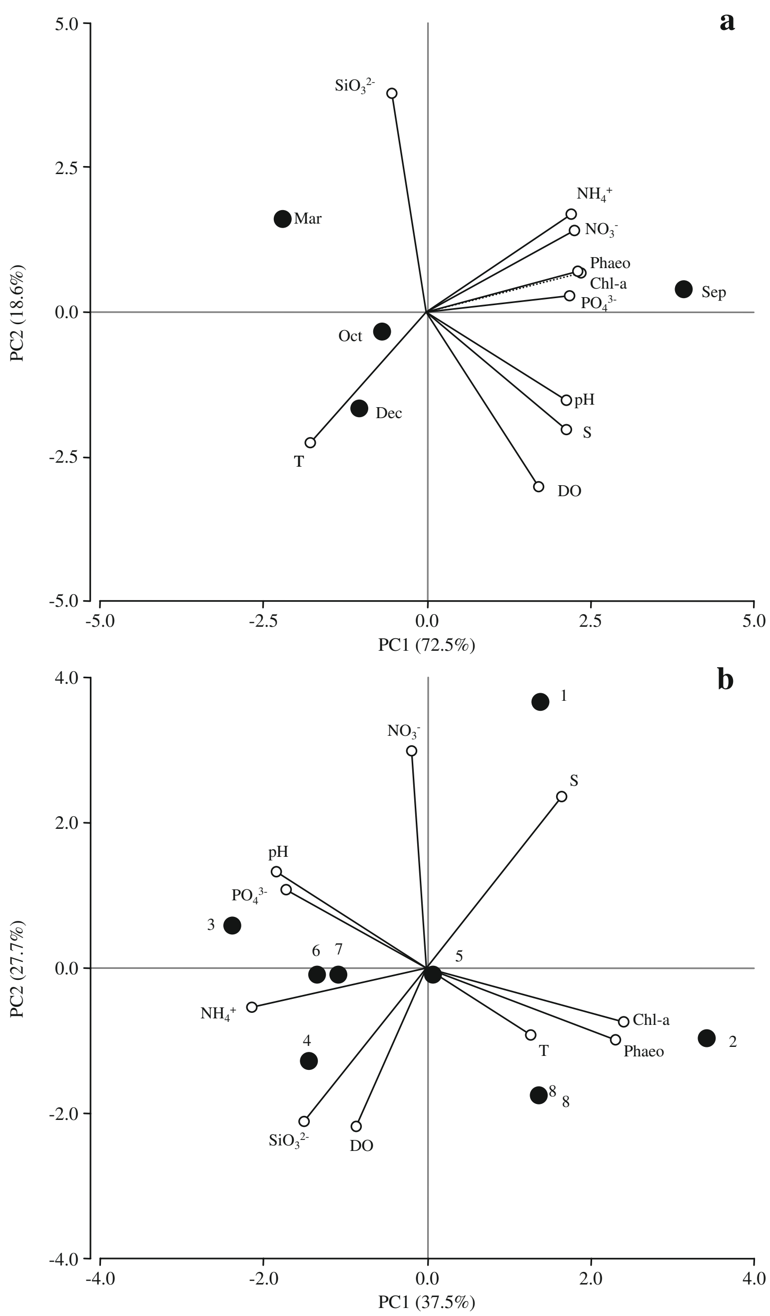

Fig. 2 Principal component analysis ordination (*PC1×PC2*), criterion of classification: **a** sampling period and **b** sampling sites

seasonal variability of the system. The lowest *T* and the highest *S* were registered in September (when the lowest income of freshwater from the land is expected); the highest of nutrients, phaeopigments, DO and pH were also measured at this time, suggesting an intense development of primary production with nutrients at non-limiting levels.

During September, monthly-integrated irradiance increases significantly in the visible band at these latitudes (400—600 nm where photosynthesis occurs; Diaz et al. 2001), which could become the main trigger for phytoplanktonic activity, even with the low temperatures found during the sampling period. October sampling was identified as the period with the highest environmental stability, as appreciated by its central location in the principal component (PC) 1 and PC 2 plan (Fig. 2a; explained variance 72.5% and 18.6%, respectively).

Figure 2b shows spatial variability of the system. The sample sites were divided in two groups, at each side of PC 1 (explained variance 37.5%): stations closer to the freshwater discharge go left (stations 3, 4, 6 and 7, with negative values on the PC 1 and enriched in nutrients) while those closer to the vents go right (stations 1, 2 and 8, with positive values over PC 1, high S and phytoplankton development). The extreme points are occupied by station 3 (maximum values of NH_4^+ and PO_4^{3-}) and by station 2 (maximum values of chl-a, phaeopigments, and T). These facts suggest a high influence of BES in the first case, and favorable conditions for phytoplankton biomass development or accumulation (considering the higher *T* of the site) in the second case. The exceptional development of macroalgae (such as: *Ulva rigida* (C. Agardh, 1824); *Enteromorpha intestinalis* (Linnaeus, 1753); *Cladophora falklandica* (J.D. Hooker and Harvey, 1845); *Porphyra woolhouseae* (Harvey, 1863), and *Pilayella littoralis* (L. Kjellm.)) observed in March within this station may support this hypothesis. Station 1 in particular, on the positive part of PC 2 (explained variance 27.7%), presents the highest S levels which would be related to the influence of UB and/or to a lower reach of continental water. High NO_3^- concentration and low DO level were also observed in this site, which suggests the development of sediment nitrification processes. Station 5 was placed at the center of the axis, what makes it the most stable of the system in relation to all the measured variables.

4.3 Sediment Quality in Encerrada Bay

Average density of sediment was 1.26±0.07 g cm^{-3}, porosity of 63.1±4.7% and the granulometry was predominantly fine. The high OM content (from 7.8 to 18.9%) showed dependence with sediment physical characteristics (r^2=0.82 p<0.05 for OM vs. porosity). Similar OM levels were found in estuarine environments with high anthrophogenic influence (Clavero et al. 2000; Cotano and Villate 2006). Anoxic conditions (E_h ranged from −261.8 to −416.2 mV, black coloration due to H_2S production by sulfate-reducing bacteria) were always observed (Table 3). Pore water NO_3^- concentration was not detected in all the stations. Total NH_4^+ concentrations varied from 0.41 to 0.92 μmol cm^{-3} wet sediment while those in pore water ranged from 0.14 to 0.59 μmol cm^{-3} wet sediment (~191.38 and 786.23 μM, respectively). The latter were similar to the ones determined in other impacted coastal environments of high latitude (Pollehne 1986; Conley et al. 1997; Tuominen et al. 1999; Maksymowska-Brossard and Piekarek-Jankowska 2001; Graca et al. 2006).

4.4 Nutrient Flux Across the Sediment–Water Interface in Encerrada Bay

The benthic fluxes study was carried out during October 2005, regarded as the season with the highest environmental stability within the studied periods (Fig. 2a). On the other hand, stations 1 and 3 were chosen for presenting different features in relation to water mass quality (Fig. 2b). In both cases, NH_4^+ and PO_4^{3-} were liberated towards the near-bottom water layer while DO and NO_3^- were incorporated by the sediment

Table 3 Sediment characteristics of Encerrada Bay

Station	Density (g cm^{-3})	Porosity (%)	O.M. (%)	E_h (mV)	NH_4^+ porewater (μM)	NH_4^+ adsorbed (μM)
1	1.22	57.4	8.5	−326.2	259.7	356.6
2	1.38	55.7	7.8	−307.0	262.2	345.9
3	1.29	61.1	9.0	−378.0	477.1	294.7
4	1.30	62.7	8.9	−416.2	481.9	117.3
5	1.22	63.7	12.0	−298.0	507.2	228.8
6	1.18	68.3	18.9	−302.7	336.2	329.6
7	1.27	64.7	12.4	−261.7	339.8	415.5
8	1.18	67.2	12.0	−330.5	416.1	133.3

E_h redox potential, *O.M.* organic matter

(Table 4). This pattern is indicative of highly impacted sediments, where oxygen is not enough for OM mineralization and NO_3^- is used as an electron acceptor. Phosphate liberation from sediment is favored by anoxic transformation of oxyhydroxides of iron on which they remained adsorbed in aerobic environments (Graca et al. 2006), and/or by frequent salinity changes in the system (Clavero et al. 2000).

Significant differences (Kruskal–Wallis $p<0.05$) between both sites were only obtained for NH_4^+ fluxes. The NH_4^+ liberated and NO_3^- consumed in station 3 were around two times higher than the fluxes supplied by BES in the same season 2,780 μmol m^{-2} h^{-1} and 144 μmol m^{-2} h^{-1}, respectively.

According to literature, DO and NH_4^+ benthic fluxes measured in this study were between 1 and 2 orders of magnitude higher than those obtained by Gil (2001) in four continental Patagonian coastal stations (Nueva Bay, Argentina). Besides that, our fluxes are higher or equal to those reported for other coastal marine environments with eutrophication signs around the world (Kemp et al. (1990) in USA; Conley et al. (1997) in Sweden; Nicholson et al. (1999) in Australia; Clavero et al. (2000) in Spain; Niencheski and Jahnke (2002) in Brazil; Graca et al. (2006) in the southeast Baltic Sea and Sakamaki et al. (2006) in Japan).

4.5 Nutrient Exchange with Ushuaia Bay

Water exchange assessment between UB and EB was carried out in October 2005: a season with high environmental stability (Fig. 2a). Figure 3a shows the tide relative amplitude in the vent at high tide (water income from UB) and low tide (water outcome from EB), respectively. Ammonium concentration in water incoming into EB at high tide (Fig. 3b) remained in 87.9±8.5 μM for the first 45 min. During the next 15 min it decreased to 1.6±0.2 μM, remaining in stationary state until the change of flux direction. Similar behavior was observed for PO_4^{3-} (0.8±0.1–0.4±0.1 μM) and inverse for conductivity (26.08±0.31–32.63±0.29 mS cm^{-1}; Fig. 3c). This evolution suggests that water incoming at the beginning of a high tide is nearly the same that out coming from EB along the last phase of at low tide. The remaining parameters measured remained homogeneous in time (Fig. 3c).

At low tide, NH_4^+ and PO_4^{3-} concentrations showed an increasing tendency with time (Fig. 3d); the values ranged from 45 to 104.6 μM and from 2 to 3.5 μM, respectively. Temperature and DO displayed slight variation, whereas conductivity remained in about 33 mS cm^{-1} until the last sampling hour, when it decreased to 25.7 mS cm^{-1}, due to dilution with freshwater (Fig. 3e).

F_i and F_e fluxes for NH_4^+ and PO_4^{3-} were estimated both at high as well as at low tide. There was a net contribution from EB to UB of 74.4 kg of NH_4^+ and of 11.4 kg of PO_4^{3-} for the tide cycle studied (Table 5). This means that EB exports a total of 41.7 kg day^{-1} and 15.7 kg day^{-1}, respectively. If we compare these to the income from continental waters (Table 1), we may say that exist a positive budget that remains in the EB promoting eutrophication. In the same sense, it is a clear evidence of nutrient input and contamination from EB to UB.

Table 4 Benthic fluxes measured in Encerrada Bay median and standard error

	Station	Benthic fluxes
NH_4^+ (μmol m^{-2} h^{-1})	1	158±51
	3	4061±390
NO_3^- (μmol m^{-2} h^{-1})	1	−148±43
	3	−264±54
PO_4^{3-} (μmol m^{-2} h^{-1})	1	35±3
	3	–
DO (mg m^{-2} h^{-1})	1	−143±17
	3	−251±54

5 Conclusions

This study shows that Encerrada Bay is a receptor of wastewater transported by four discharges (Buena Esperanza Stream and three pluvials), holding typical features of sewage. BES in particular, is the main nutrients source to the Bay, with a daily inorganic N contribution higher than 100 g N m^{-2} year^{-1} which makes the system highly disturbed (Boynton et al. 1982). The shallowness of EB, the strong southwest winds and the daily exchange with UB favor mixture and oxygenation of the water column. Notwithstanding, high nutrient and phaeopigments concentrations registered in the water mass, the high phaeopigments/chl-a ratio, the sediment anoxia (with high OM and NH_4^+

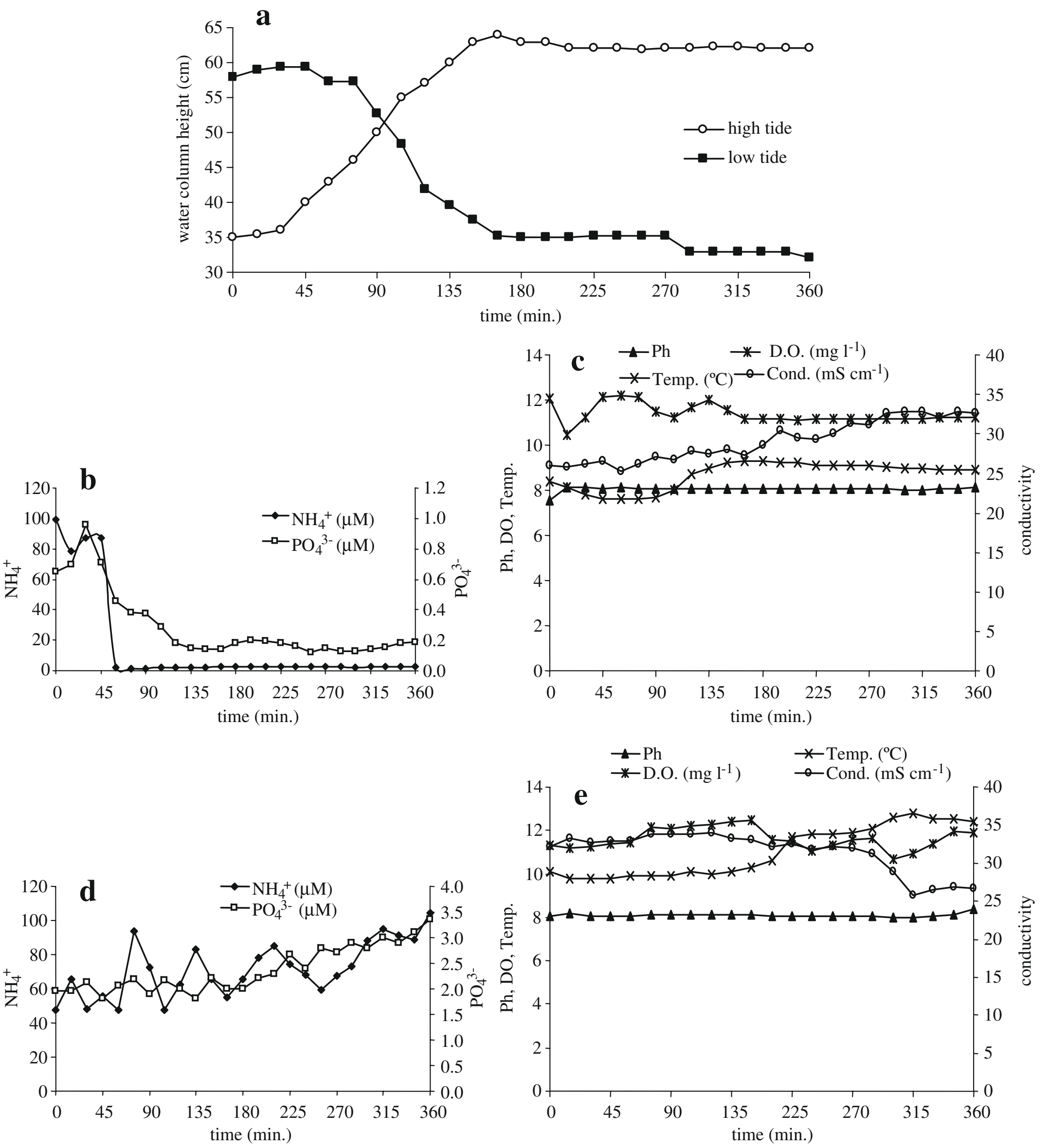

Fig. 3 **a** Water column height variation (cm) during high and low tide in the vent that communicates Encerrada and Ushuaia Bays. Ammonium and phosphate concentrations in **b** water income from Ushuaia bay and **d** water outcome from Encerrada bay during high and low tide respectively. Salinity, temperature, conductivity, pH and DO distribution in **c** water income from Ushuaia bay and **e** water outcome from Encerrada bay

Table 5 Nutrient masic flux exchanged in each tide cycling between Encerrada and Ushuaia Bays

Nutrient	F_i (kg/tide cycle)	F_e (kg/tide cycle)	F_e−F_i (kg/tide cycle)	Contribute daily to UB (kg day^{-1})
NH_4^+	53.6	74.4	20.8	41.7
PO_4^{3-}	3.6	11.4	7.9	15.7
NO_3^-	18.5	7.4	−11.1	−22.2

values, DO and NO_3^- demands and NH_4^+ and PO_4^{3-} liberation), as well as the great development of macro-algae, show the severe impact that the system is undergoing. Encerrada Bay is acting as a stabilization pond, reducing the impact that direct sewages outflow would produce in Ushuaia Bay, but at its own deterioration risk.

Strong population growths in Ushuaia City and scarce planning have left a large number of homes without sewage network. This favored illegal waste-water outflows to runoff and streams across the city, which finally discharges into the coastal systems. A proper wastewater management in Ushuaia city would permit to mitigate the impact on BES and other effluents, and in consequence to improve the environmental condition in Encerrada Bay. This study supplies background information that will contribute to the development of management and control measures to preserve the coastline of Ushuaia city.

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References

APHA. (1980). *Standard methods for the examination of water and wastewater* (15th ed.), p. 1134. Washington, District of Columbia: American Public Health Association.

Boynton, W.R., Kemp, W.M., & Keefe, C.W. (1982). A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In National Academy Press (Ed.), *Managing wastewater in coastal urban areas. The Role of nutrients in coastal Waters.* Appendix A (pp. 177-202). Washington, D.C.

Christensen, B., Vedel, A., & Kristensen, E. (2000). Carbon and nitrogen fluxes in sediment inhabited by suspension-feeding (*Nereis diversicolor*) and non-suspension-feeding (*N. virens*) polychaetes. *Marine Ecology Progress Series, 192*, 203–217. doi:10.3354/meps192203.

Clavero, V., Izquierdo, J. J., Fernández, J. A., & Niell, F. X. (2000). Seasonal fluxes of phosphate and ammonium across the sediment-water interface in a shallow small estuary (Palmones River, southern Spain). *Marine Ecology Progress Series, 198*, 51–60. doi:10.3354/meps198051.

Conley, D. J., Stockenberg, A., Carman, R., Johnstone, R. W., Rahm, L., & Wulff, F. (1997). Sediment-water nutrient fluxes in the Gulf of Finland, Baltic Sea. *Estuarine, Coastal and Shelf Science, 45*, 591–598. doi:10.1006/ecss.1997.0246.

Cotano, U., & Villate, F. (2006). Anthropogenic influence on the organic fraction of sediments in two contrasting estuaries: A biochemical approach. *Marine Pollution Bulletin, 52*, 404–414. doi:10.1016/j.marpolbul.2005.09.027.

Diaz, S., Deferrari, G., Booth, D. R., Martinioni, D., & Oberto, A. (2001). Solar irradiances over Ushuaia (54.49º S, 68.19º W) and San Diego (32.45º N, 117.11º W) geographical and seasonal variation. *Journal of Atmospheric and Solar-Terrestrial Physics, 63*, 309–320. doi:10.1016/S1364-6826(00)00178-4.

Esteves, J. L., Bonin, P., Blanc, F., Mille, G., & Bertrand, J. C. (1986). Reduction du nitrate dans les sediments marins comparaison des activites mesurees in vitro et in situ. *Deuxième Colloque International de Bactèriologie Marine. Actes de Colloques., 3*, 319–324.

GESAMP. (1990). Joint Group of Experts on the Scientific Aspects of Marine Pollution: the State of the Marine Environment. UNEP. Regional Seas Reports and Studies. Nº 115. 87 pp.

Gil, M.N. (2001). *Eutroficación: Rol del nitrógeno en ecosistemas marinos costeros. Ph.D. Thesis*. Universidad Nacional del Sur, Bahía Blanca, Buenos Aires. 132 pp.

Graca, B., Witek, Z., Burska, D., Bialkowska, I., Lukawska-Matuszewska, K., & Bolalek, J. (2006). Pore water phosphate and ammonia below the permanent halocline in the south-eastern Baltic Sea and their benthic fluxes under anoxic conditions. *Journal of Marine Systems, 63*, 141–154. doi:10.1016/j.jmarsys.2006.06.003.

InfoStat. (2008). Grupo InfoStat, FCA, Universidad Nacional de Córdoba. versión 2008. Córdoba, Argentina.

Instituto Nacional de Estadísticas y Censo, (INDEC). (2008). Retrieved from http://www.indec.mecon.ar/censo2001s2_2/ampliada_index.asp?mode=94

Iturraspe, R., Sottini, R., Schroder, C., & Escobar, J. (1989). Hidrología y variables climáticas del territorio de Tierra del Fuego. Centro Austral de Investigaciones Científicas y Técnicas. Contribución Científica Nº7. 196 pp.

Kemp, W. M., Sampou, P., Caffrey, J., Mayer, M., Henriksen, K., & Boynton, W. R. (1990). Ammonium recycling versus denitrification in Chesapeake Bay sediments. *Limnology and Oceanography, 35*(7), 1545–1563.

Kontas, A., Kucuksezgin, F., Altay, O., & Uluturhan, E. (2004). Monitoring of eutrophication and nutrient limitation in the Izmir Bay (Turkey) before and after wastewater treatment plant. *Environment International, 29*, 1057–1062. doi:10.1016/S0160-4120(03)00098-9.

Maksymowska-Brossard, D., & Piekarek-Jankowska, H. (2001). Seasonal variability of benthic ammonium release in the surface sediments of the gulf of Gdansk (southern Baltic Sea). *Oceanologia, 43*(1), 113–136.

Metcalf, & Eddy. (1991). *Wastewater engineering treatment disposal and reuse* (Thirdth ed.), p. 1334. Singapore: McGraw-Hill.

Nicholson, G. J., Longmore, A. R., & Berelson, W. M. (1999). Nutrient fluxes measured by two types of benthic chamber. *Marine & Freshwater Research, 50*, 567–572. doi:10.1071/MF98047.

Niencheski, L. F., & Jahnke, R. A. (2002). Bentic respiration and inorganic nutrient fluxes in the estuarine region of Patos Lagoon (Brazil). *Aquatic Geochemistry, 8*, 135–152. doi:10.1023/A:1024207220266.

Pinto, A., Von Sperling, E., & Moreira, R. (2001). Chlorophyll-a determination via continuous measurement of plankton fluorescence: methodology development. *Water Research, 35*(16), 3977–3981. doi:10.1016/S0043-1354(01)00102-6.

Pollehne, F. (1986). Benthic nutrient regeneration processes in different sediment types of Kiel Bight. *Ophelia, 26*, 359–368.

Sakamaki, T., Nishimura, O., & Sudo, R. (2006). Tidal time-scale variation in nutrients flux across the sediment-water interface of an estuarine tidal flat. *Estuarine, Coastal and Shelf Science, 67*, 653–663. doi:10.1016/j.ecss.2006.01.005.

Saunders, J. E., Al Zahed, K. M., & Paterson, D. M. (2007). The impact of organic pollution on the macrobenthic fauna of Dubai Creek (UAE). *Marine Pollution Bulletin, 54*, 1715–1723. doi:10.1016/j.marpolbul.2007.07.002.

Servicio de Hidrografía Naval, (SHN). (2008). Retrieved from http://www.hidro.gov.ar/Oceanografia/Tmareas/R_Mareas.asp

Strickland, J.D.H., & Parsons, T.R. (Eds.). (1972). *A Practical Handbook of the Seawater Analysis*. (Fisheries Research Board of Canada, Bulletin 167). 310 pp.

Tuominen, L., Mäkelä, K., Lehtonen, K. K., Haahti, H., Hietanen, S., & Kuparinen, J. (1999). Nutrient fluxes, porewater profiles and denitrification in sediment influenced by algal sedimentation and bioturbation by *Monoporeia affinis*. *Estuarine, Coastal and Shelf Science, 49*, 83–97. doi:10.1006/ecss.1999.0492.

Vollenweider, R.A. (1992). Coastal marine eutrophication: principles and control. In R. A. Vollenweider, R. Marchetti & R. Vicviani - Elsevier (Ed.), Science of the Total Environment, Supplement: Marine Coastal Eutrophication (pp. 1-120). Amsterdam.

Yakub, G. P., Castric, D. A., Stadterman-Knauer, K. L., Tobin, M. J., Blazina, M., Heineman, T. N., et al. (2002). Evaluation of Colilert and Enterolert defined substrate methodology for wastewater applications. *Water Environment Research, 74*(2), 131–135. doi:10.2175/106143002X139839.