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DOI 10.1007/s11270-011-1041-0

The Effects of Hydraulic and Organic Loadings on the Performance of a Full-Scale Facultative Pond in a Temperate Climate Region (Argentine Patagonia)

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Received: 10 June 2011 / Accepted: 8 December 2011

Abstract This work focuses on the performance of a primary facultative pond, in a full-scale waste stabilization pond system, located in a temperate climate region (average air temperature in winter, 7.7°C; spring and autumn, 14.0°C; and summer, 19.9°C) in Puerto Madryn city—Argentine Patagonia (42°45′S; 65°05′W). Experimental work was conducted for 43 months in seven sampling points. During the experimental time frame, the influent flow rate increased from 12,000 to 15,500 m^3/day; the surface organic loading ranged from 55 to 68 kg BOD_5/ha·day and the theoretical retention time decreased from 31 to 24 days. The results indicate that a primary facultative pond performing in this region, to keep predominant facultative conditions and acceptable filtered biochemical oxygen demand (BOD_5) removal, should be loaded with an organic loading rate of up to 60 kg BOD_5/ha·day. The flow and organic loading increase affected the ammonium removal process, extending the period time in which ammonium removal was less than 50% and nitrate was not detectable; at first, this period occurred during winter strictly and then covered part of autumn and part of spring, too. Ammonium removal was clearly temperature dependent and directly related to chlorophyll *a* and nitrate concentrations (i.e. higher ammonia removals were reported under summer conditions when chlorophyll *a* and nitrate concentrations were higher), but was not linked with high pH values. The ammonium volatilization as a predominant removal process could be discarded, while ammonium nitrification–denitrification and algal nitrogen uptake seems to be the dominant mechanisms.

Keywords Natural wastewater treatment · Nitrogen dynamics · Organic matter removal · Organic loading · Hydraulic loading · Temperate climate

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1 Introduction

Waste stabilization pond (WSP) technology is used worldwide for domestic and industrial wastewater treatment. WSPs are large, shallow earthen basins which can be used alone, comprising at any one location one or more series of ponds, or in combination with other wastewater treatment processes as a storage unit, primary and secondary treatment or polishing component. WSPs can be classified based on their depth and the biochemical reactions that occur inside

Published online: 27 December 2011

the pond; according to that, the main types of ponds are: anaerobic (2 to 4 m deep), facultative (≈1.5 m deep) and maturation ponds (≈1.0 m deep). WSP technology is entirely a natural process in which wastewater is treated by biochemical reactions involving both algae and bacteria. For that reason, the rate of oxidation is slower, and as a consequence, hydraulic retention times are longer than in conventional electro-mechanical wastewater processes (Mara 2004) (e.g. activated sludge, which has forced aeration and physical mixing). The reasons behind the popularity of WSP systems are based on its economic advantages over highly electro-mechanized systems and its high efficiency regarding soluble organic matter and pathogen removal, despite variations in the flow rate and associated quality of the pond influent (Athayde Junior et al. 2000; Ellis and Rodrigues 1995; Madera et al. 2002). WSPs in operation have also shown relatively steady removal efficiencies throughout the year, particularly in tropical countries where environmental conditions are more favourable for alga–bacterium drive processes. However, their performance in temperate and cold regions is deeply affected by variations in weather conditions (e.g. air temperature, solar radiation, number of hours of daylight, etc.), mostly during colder months. Such pattern is attributed to well-defined changes in the metabolism of the biological communities supported within WSPs, which includes variations in the dynamics of nutrients, soluble and suspended organic matter concentrations and phytoplanktonic activity (Abis and Mara 2003, 2005; Heaven et al. 2003 and Santos and Oliveira 1987).

In the study of WSP, organic matter and pathogen removal mechanisms are well known and incorporated within design procedures (Mara 2004); however, the controversy is still open regarding the main mechanisms dominating nitrogen transformation and removal and how they are affected by organic loading and hydraulic retention times. One classical example is a study conducted at the WSP system in Corinne, Utah, where three independent authors analysing the same data reached different conclusions about the main mechanism for nitrogen removal—Pano and Middlebrooks (1982) and Reed (1985) attributed it to ammonia volatilization, while Ferrara and Avci (1982) did it through sedimentation of organic nitrogen. Recent works have provided evidence to support other feasible routes for nitrogen removal such as nitrogen algal uptake and sedimentation of dead biomass (Camargo Valero et al. 2010a), and nitrification–denitrification (Camargo Valero et al. 2010b; Lai and Lam 1997; Senzia et al. 2002; Zimmo et al. 2004). But overall, ammonia volatilization is still the most controversial process (Camargo Valero and Mara 2007, 2010; Pearson 1996; Rockne and Brezonik 2006; Silva et al. 1995; Soares et al. 1996; Zimmo et al. 2003).

The use of WSP technology in Argentina has been widespread, mainly due to its low cost (i.e. capital, operating and maintenance costs), simplicity of operation and maintenance, and the availability of large areas of land near urban centres. Full-scale WSP systems are in operation both in important cities with more than one million inhabitants (e.g. Mendoza) and in small communities like Sierra Grande, Patagonia, with less than 8,000 inhabitants. Along the Patagonian coast, there are ten WSP systems in operation (Esteves and González 2008); nevertheless, there is little information about their performance, and further research is needed to adjust design criteria to local conditions (Anzorena 2001; Esteves et al. 1996). The aims of this work was to study the dynamics of organic matter, inorganic nitrogen and chlorophyll *a*, in a full-scale WSP system currently in operation in the Province of Chubut (Argentina), and to determine the more suitable surface loading rates to maintain a good quality effluent throughout the year, under the very variable weather conditions found in the Argentine Patagonia.

2 Material and Methods

2.1 Area of Study and Climate Conditions

The present study was conducted at the sewage treatment works (STW) in Puerto Madryn, Argentina. Puerto Madryn is a small city in the Province of Chubut in the Argentine Patagonia, on the coast of the Atlantic Ocean (≈90,000 inhabitants). Its STW is a natural wastewater treatment system based on WSP technology, and it is located at 10 km northwest from the city (42° 45′ S; 65° 05′ W). That WSP system has been in operation since 2001 and comprises a screening unit (3-mm bar screen) followed by a primary facultative pond in U-shape (surface area, 25 ha; depth, 1.5 m) and two maturation ponds connected in series (35 ha surface area and 1 m deep each); see Fig. 1. Although the original design includes a secondary facultative pond, with similar specifications with the existing one, it has not been commissioned yet. Therefore, this work focuses on the performance of

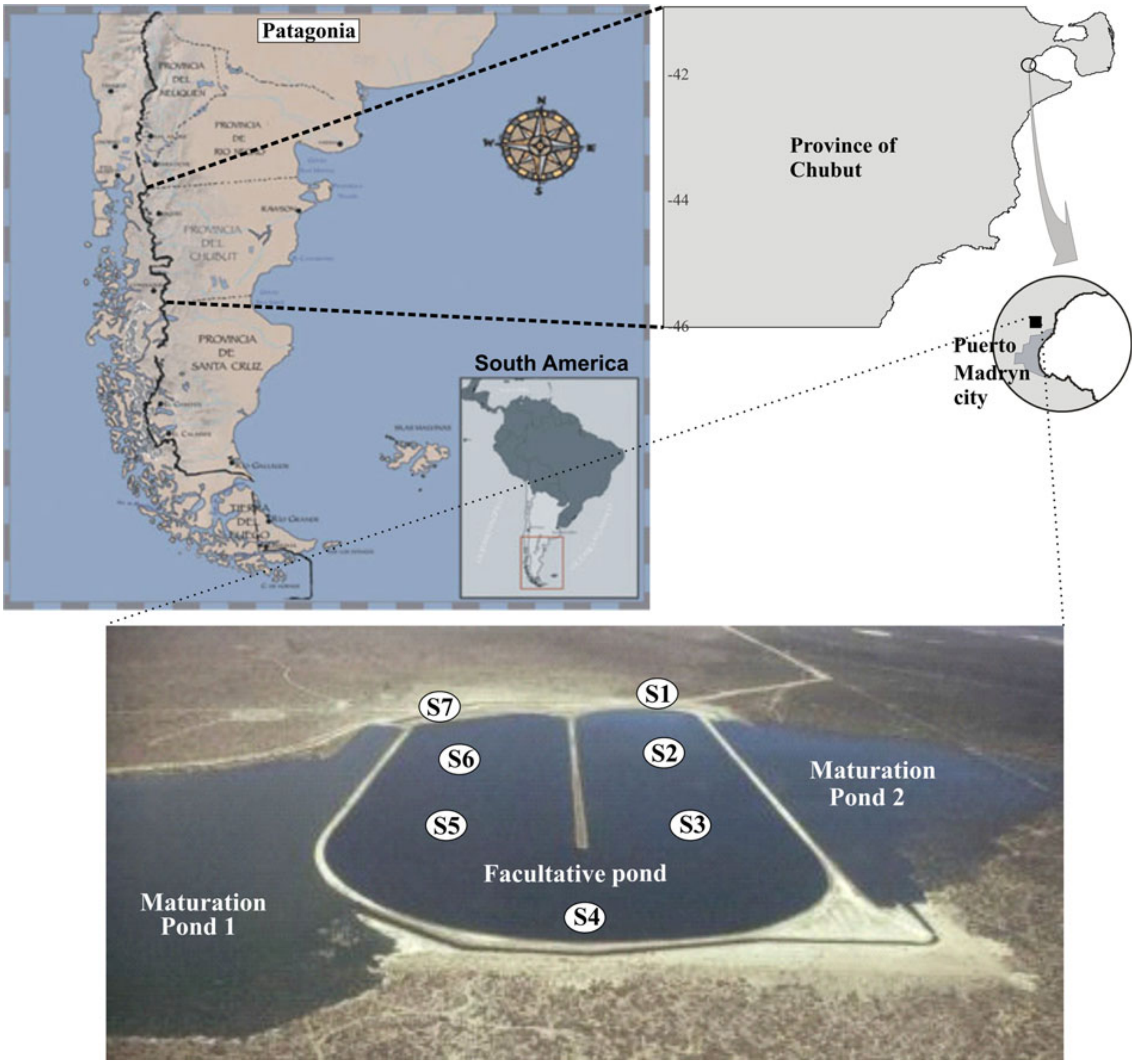


Fig. 1 Location of Puerto Madryn's sewage treatment works and water sampling points. *S1* inlet channel after screening, *S2* inside facultative pond at 200 m from S1, *S3* at 400 m from S1, *S4* at 600 m from S1, *S5* at 800 m from S1, *S6* at 1,000 m from S1, and *S7* outlet channel

the existing primary facultative pond, in order to provide reliable information for the future expansion of the current WSP system.

Puerto Madryn is located in a semi-arid zone with a temperate–cold climate and an important water deficit. Weather conditions clearly determined three water temperature intervals into WSP system throughout the year with average figures of 7.7°C in winter (June–August), 14.0°C in spring and autumn (September–November and March–May, respectively) and 19.9°C in summer (December–February); average seasonal weather conditions are shown in Table 1. Considering that the average precipitation and evaporation rates are 200 and 2,000 mm/year, respectively (Battro 1983), the local government has included the final effluent from their STW within a comprehensive water management programme. It is aimed to achieve zero discharge of treated wastewater into the sea, and therefore, nearly a half of the final effluent would be pumped back to the city for wastewater reuse in urban irrigation (i.e. irrigation of

Table 1 On-site average weather characteristics by season

	Air temperature (°C)			Solar radiation (W; h/m^2/day)	Wind speed[a] (km/h)
	Average	Average maximum	Average minimum		
Spring	14.0	29.0	0.8	5,778	15.5
Summer	19.9	35.4	6.3	7,092	16.7
Autumn	14.4	29.8	0.7	3,185	14.0
Winter	7.7	19.9	−5.0	2,064	13.4

Information provided by CENPAT meteorological station from data collected during the period of study and based on monthly average figures

[a]The prevailing wind in Puerto Madryn is westerly

parks, street gardens, etc.) and to feed several reservoirs as part of the urban fire-fighting system. The other 50% of treated effluent will be used to fight forest fires in the Valdes Peninsula Nature Reserve—listed as a World Heritage Site by UNESCO in 1999.

2.2 Sampling and Laboratory Analysis

Water samples were collected fortnightly from selected sampling points in the primary facultative pond as follows: (a) inlet channel after screening (sampling point S1), (b) intermediate point inside facultative pond at surface level (sampling point S4) and (c) outlet channel (sampling point S7). Composite samples were collected from S1 and made up by blending equal volume of single samples collected hourly from 0800 to 1900 hours; single samples were collected from S4 and S7. In total, 62 sampling sessions were conducted over 43 months. Additionally, one sampling session was conducted per season (nine in total) for the five sampling stations within the facultative pond (i.e. sampling points S2, S3, S4, S5 and S6); it included collecting composite of superficial water samples through the cross section by using a floating autonomous sampler developed in house. Sampling points are illustrated in Fig. 1. Each sample was processed on site for temperature, dissolved oxygen (DO) and pH (DO metre, model YSI 58, YSI Inc., Yellow Springs, OH; pH metre, model PH82, Yokogawa Electric Corp., Tokyo). In the laboratory, water samples were processed following standard analytical methods (APHA 1980) for unfiltered and filtered biochemical oxygen demand (BOD_5; method 507), nitrite (419), nitrate (418 C), ammonium (417 C), chlorophyll *a* (1002 G) and suspended solids (SS; 209 D). Suspended organic nitrogen in S7 was estimated from SS readings using the relationship found by Camargo Valero et al. (2010a) in the effluent of a primary facultative pond.

3 Results and Discussion

3.1 Characteristics of Pond Influent

Laboratory results from sampling point S1 are shown in Table 2. The quality of the screened wastewater entering the primary facultative pond reported mean characteristics within the lower typical figures for raw wastewater (i.e. weak-strength wastewater). However, mean ammonia concentration was considerably high, which may be a combined result of the biological hydrolysis of organic matter and the long residence time within the sewer system (i.e. 10 km from the city). Similar analysis can be made to explain the low BOD_5/chemical oxygen demand (COD) ratio and BOD_5 contribution per capita based on the quality of pond influent (i.e. BOD_5/COD= 0.45; ≈25 g BOD_5 pppd). The weak-strength character of the raw influent can also be linked to high water consumption in Puerto Madryn (≈370 l/person day). It is indeed higher than most figures reported from developing countries and even from EU countries (e.g. developing countries, 40–100 l/person day (Mara 2004) and EU countries, 80–260 l/person day (UNEP 2004)). During the experimental time frame (43 months), Puerto Madryn had an important increment in sewer coverage, and for that reason, both inlet flow and organic loading rates rose about 23% in the primary facultative pond (i.e. average inlet flow rate increased from 12,051 to 15,500 m^3/day, and organic loading (λs) increased from 55 to 68 kg BOD_5/ha·day; see Fig. 2). As a consequence, the theoretical hydraulic retention time (HRT) decreased from 31 to 24 days. The mean sewage temperature was 13.5°C during winter and 22.5°C during summer.

3.2 Organic Matter Removal Efficiency

Unfiltered and filtered BOD_5 concentrations from samplings points S1 to S7 are shown in Fig. 3. During

Table 2 Characteristics of raw wastewater entering the WSP system

Parameter	Unit	Mean value[a]	Typical sewage strength[b]	
			Weak	Strong
Suspended solids (SS)	mg/l	103±37	100	350
Settleable solids 2 h	ml/l	1.7±1.4	5	20
Ammonium	mg NH_4–N/l	42.2±7.9	12	50
Nitrite+nitrate	mg N/l	0.04±0.02	0	0
BOD_5(unfiltered)	mg/l	115±42	110	400
COD	mg/l	251±61	250	1,000

[a] Values reported as mean±one standard deviation

[b] Metcalf and Eddy (1996)

the experimental time frame, unfiltered BOD_5 dropped by 41% on average at sampling point S2, mainly due to sedimentation of organic matter, whereas in the pond effluent (S7), unfiltered BOD_5 analysis reported a mean value of 43.6±20.8 mg/l, and filtered BOD_5 had an average value of 16.1±12.0 mg/l (i.e. figures reported as mean±one standard deviation). Both S4 and S7 stations presented lower concentrations, compared with the other internal stations of the STW, which could be originated in that S4 and S7 stations were sampled from the beginning to the end of the study (more than 60 opportunities), and therefore covering greater behaviour variability and greater number of low values mainly, while the other internal stations were sampled in lesser quantity (nine samples) and towards the end of the study, which is accounted for higher values of unfiltered and filtered BOD_5. Considering the big discrepancy between average precipitation and evaporation rates, BOD_5 removal efficiency was calculated on waste load removal basis for unfiltered and filtered BOD_5 load in pond effluent (Ferrara and Avci 1982).

Results from the primary facultative pond under study showed that unfiltered BOD_5 removal was affected by the increasing surface organic loading, resulting in significant differences between the unfiltered BOD_5 removal when surface organic loading (λs) was less than 60 kg/ha·day and when it was more than 60 kg/ha·day (p=0.004). The mean unfiltered removal was 66.7% (λs <60) and 54.6% (λs >60). The same behaviour was registered for filtered BDO_5 removal (p=0.001), with mean values of 89.8% (λs <60) and 81.8% (λs >60).

Comparing unfiltered BOD_5 removal between summer and winter, were observed significant differences (p=0.02), being the removal more important during cold months (72.6%) than in warm months (58.4%). The unfiltered BOD_5 concentration increased in the warm season, coinciding with higher concentrations of chlorophyll a and suspended solids, whereas filtered BOD removal had not presented significant differences between seasons (p=0.81), with mean removals of 89.6% (summer) and 88.9% (winter).

The dynamics of suspended organic matter in the pond effluent (i.e. unfiltered BOD_5–filtered BOD_5) was

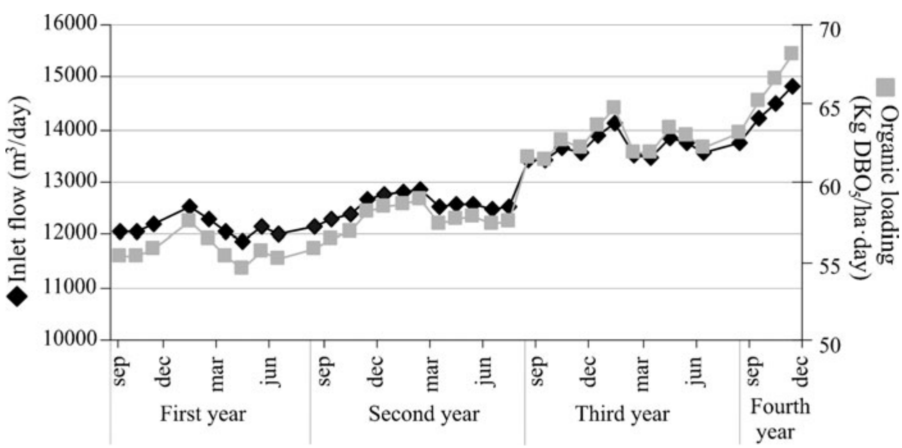


Fig. 2 Time evolution of raw wastewater flow and organic loading during the experimental timeframe

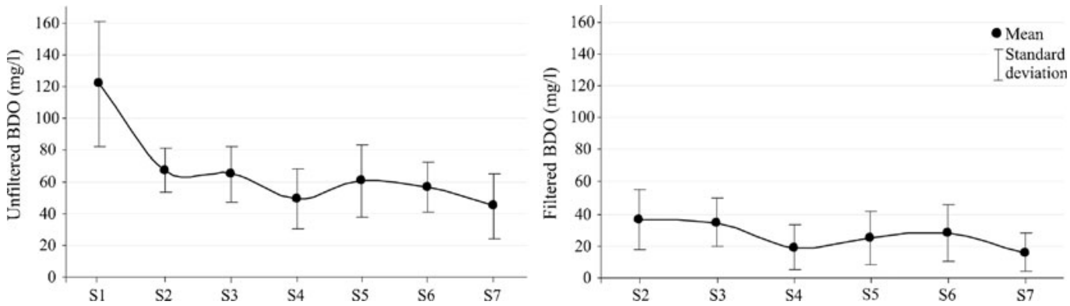


Fig. 3 BOD_5 concentration from sampling points *S1* to *S7*

found clearly dependent on phytoplankton production, which at the end responds to weather conditions and affects BOD removals. Minimum unfiltered BOD_5 values occurred usually at the beginning of spring, and it was coincident with a major zooplankton proliferation and low concentration of chlorophyll *a*, whereas maxima figures occur in summer, when chlorophyll *a* concentration also reaches its maximum.

Theoretically, the maximum organic loading rate that can be applied to this facultative pond is 128 kg/ha·day for the coldest period of the year in Puerto Madryn (i.e. 7.7°C in winter) and before it becomes anaerobic (McGarry and Pescod 1970). Although to maintain facultative conditions under similar temperature conditions like in Puerto Madryn, the US EPA (1983) has suggested that the organic loading rate should range from 48 to 56 kg BOD_5/ha·day. On the other hand, Abis and Mara (2005) reported that UK primary facultative ponds could be loaded at around 80 kg BOD_5/ha·day with a minimum HRT of 20 days; even though algae may not be present during the winter at this loading rate, they reported no detrimental effect to pond performance removals regarding unfiltered BOD_5 (70–90% removal efficiency), filtered BOD_5 (>95%) and SS (>90%).

Performance indicators for primary productivity were strongly affected by the organic loading rate (Table 3). It has been suggested that the absence of facultative conditions in a waste stabilization pond may produce a poor performance, and therefore, Pearson (1996) proposed a minimum concentration of chlorophyll *a* in the pond liquid to avoid failure conditions (i.e. 300 µg/l of chlorophyll *a*). At the end of this study when the facultative pond was fed with a higher organic loading rate, chlorophyll *a* concentration was found in some cases lower than 50 µg/l, and consequently, DO dropped to a minimum of 0.45 mg/l. Such conditions clearly affected pond performance in terms of organic matter removal (i.e. mainly filtered BOD removal).

3.3 Nitrogen Dynamics

The behaviour of inorganic nitrogen was clearly seasonal, with high ammonium concentrations found in the pond effluent during the colder months and lowest during summer. In fact, a significant correlation was observed between ammonium concentration and temperature both during summer and winter months ($r_{\text{NH3-Temp}}=-0.84$; $p<0.001$). Ammonium concentration was also well correlated with parameters associated with primary productivity ($r_{\text{NH3-SS}}=-0.79$, $p=0.001$; $r_{\text{NH3-Chlo}}=-0.72$, $p<0.0001$). Ammonium nitrification was reported during the warmer months of the year; in

Table 3 Mean DO and chlorophyll *a* concentrations in winter and summer

Parameter	First year		Second year		Third year	
	Summer	Winter	Summer	Winter	Summer	Winter
DO, mg/l	11.5±4.6	7.9±2.4	5.0±1.5	5.9±3.3	2.5±0.7	2.0±1.1
Chlorophyll *a*, µg/l	840±354	345±103	2,295±755	326±149	2,041±1,034	203±219
λ_S, kg BOD_5/ha day	56.4±1.1	55.6±0.6	58.3±0.8	60.2±2.3	62.8±1.6	64.1±1.4

Mean±one standard deviation

fact, nitrate concentrations up to 14 mg N/l were found in the pond effluent in early autumn, late spring and summer (Fig. 4).

It was noted that the most remarkable behaviour was found between the two extreme stations (winter–summer): in winter, ammonium concentration was between 30 and 51 mg N/L, while in summer was less than 0.8 mg N/L (except for a couple of specific events during a crash period in last summer). A clear transition between extreme seasons was observed in the other seasons of the year: the first part of the autumn drag the summer behaviour, with low ammonium concentrations, and as time progresses, the temperature decreases, and the ammonium begins to rise up to 30 mg N/L when winter started. The reverse transition happens during spring: at early days, the ammonium behaviour resembles the winter (still not recovered the phytoplankton or nitrifying bacteria populations) so the concentration was high, and near the end of spring, the concentration began to be below 20 mg N/L, and even at early December registered values were below 1 mg N/L.

The opposite behaviour was observed for nitrite plus nitrate concentrations: detectable values during summer (except during the crash period), late spring and early autumn and not detectable along all the winter, early spring and late autumn. Hurse and Connor (1999) reported a similar seasonal pattern, with predominant nitrification process during warmer months and complete loss of nitrification in winter.

Ammonium removal was consistently high in early autumn, late spring and summer (>90%) and negligible in winter, independent of organic loading. Nitrification occurred from late spring to early autumn during the first year of this study; however, the loading increment over time shortened the presence of nitrate in the pond effluent to a few months in the third year. In fact, ammonium concentration in the pond effluent rose up to 40 mg N/L in the last monitored summer; low concentrations of dissolved oxygen and nitrate were also reported during the same monitored period. On the other hand, suspended organic nitrogen (mainly algal biomass) played a very important role in ammonium nitrogen removal as that fraction accounted for most of the nitrogen found in the pond effluent, and along with ammonium nitrification, nitrogen algal uptake is seem to be the dominant mechanism for ammonium removal when both in-pond and ex-pond environmental conditions are favourable for phytoplanktonic activity (Fig. 5). However, the increment of hydraulic and organic loadings was detrimental to ammonium removal performance, and that consequently produced a longer ammonium supremacy over oxidising nitrogen forms in the pond effluent, which indeed is an evidence of changes in biochemical processes ruling nitrogen transformation and removal.

Considering the results for monitoring ammonium, nitrate and chlorophyll *a* from in-pond water sampling points (Fig. 6), it is clear that ammonium is better removed during summer and autumn, when temperature and in-pond environmental conditions were more favourable for nitrification and phytoplanktonic activity (see Fig. 6b and c, respectively). Moreover, it comes to light the fact that most of the ammonium is removed within the first 200 m of the pond (from S1 to S2), where also most of the suspended organic matter is removed by sedimentation and further accumulation in the sludge layer. Such conditions may represent additional evidence to support the theory of ammonium removal via alternative routes like nitrification supported by

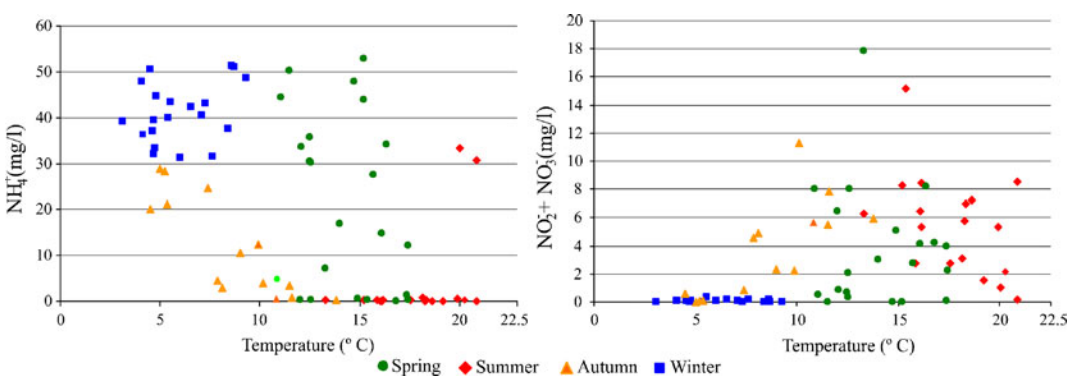
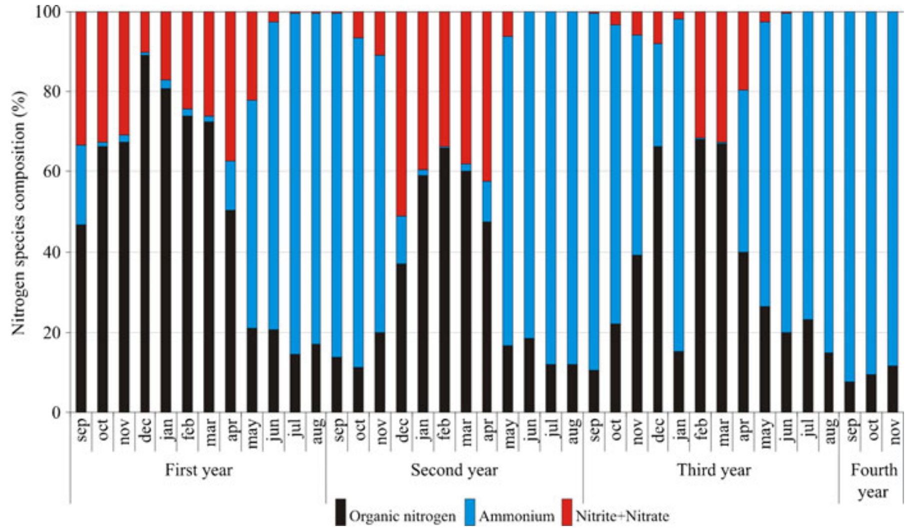


Fig. 4 Behaviour of nitrogen inorganic forms in relation with water temperature at station E7

Fig. 5 Nitrogen fractions found in the pond effluent and organic loading during the experimental time frame



methanotrophs under anoxic conditions in the sludge layer and subsequent denitrification in the water column (Camargo Valero et al. 2010b).

The organic loading increase would act by delaying the recovery of the phytoplankton activity after the winter (directly resulting in a lower ammonium consumption and a diminution on the availability of attachment for nitrifying bacteria and indirect effects, such as a lower dissolved oxygen available for ammonium oxidation and less protection from the sun ray inhibition on the nitrifier bacteria (Hurse and Connor 1999)). As a result, it would affect the nitrifying growth rate of bacteria, being more susceptible to washout, which was described as one of the main factors affecting the nitrification process in stabilization ponds, originated in an increase of hydraulic load or a fault in the rate of bacterial growth (Siripong and Rittmann 2007). Abis and Mara (2003) registered a seasonal pattern in ammonium removal similar to that

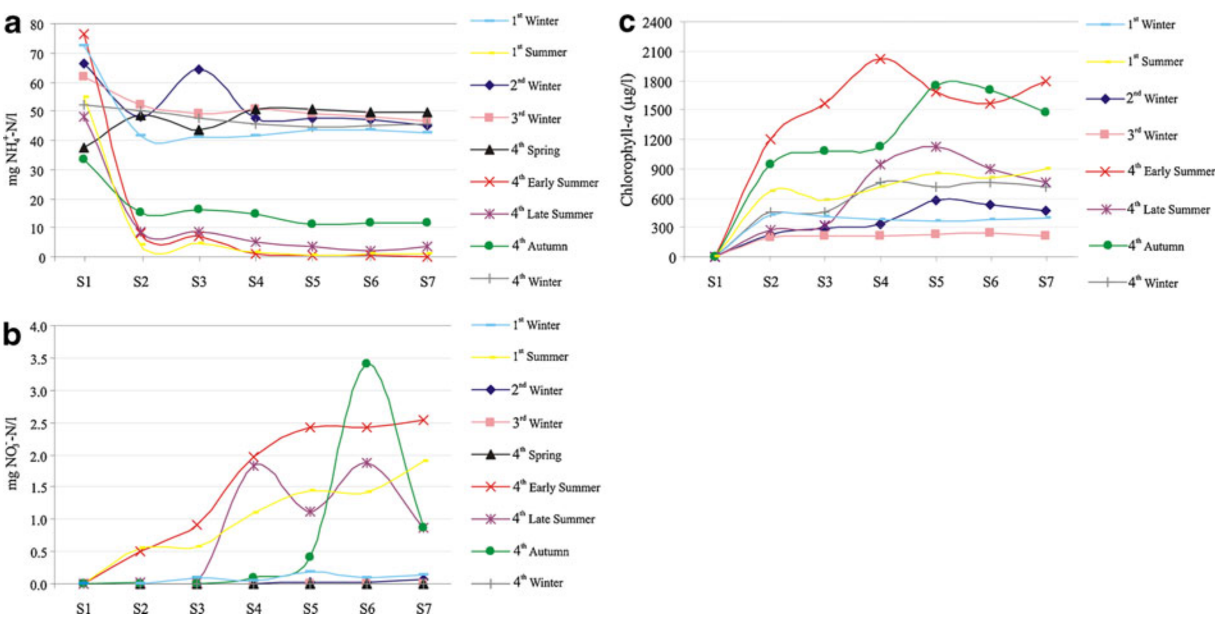


Fig. 6 Seasonal ammonium, nitrate and chlorophyll *a* variation between in-pond sampling points (each data set corresponds to a diary sampling)

observed in our study and directly related to the organic load, which limited the phytoplankton population development and ultimately affect ammonium removal processes. In our study, since in the first winter there was a clear decline in the nitrification activity, which added to the loading increase (hydraulic and organic) experienced in the system, this led to the retardation of the re-establishment of viable nitrifying populations in the water column, resulting to the absence of detectable nitrate in the effluent in November of the last year.

Although in view of the common low nitrite and nitrate accumulation in WSP, it has been suggested that nitrification is not likely to be performed in facultative or maturation ponds despite prevalent in-pond aerobic conditions. In our case and others reported by Morrison (1984), Lai and Lam (1997) and Hurse and Connor (1999), nitrification was clearly present and appears to be an intermediate step leading either to a further denitrification stage or to algal nitrogen uptake under conditions with limited availability of ammonium as a nutrient. Along with that, predominant hydraulic conditions close to complete mixing in WSP may also help to mask the occurrence of the nitrification processes (Fig. 6c shows nitrate transformations in a WSP with predominant plug flow conditions). Another possible explication could be the variation between day and night in dissolved oxygen concentration (originated in the important algae concentration in summer, producing photosynthetic oxygen during the day and breathing during the night). Both processes could produce and segregate mutually complementary processes along the pond (i.e. anoxic and aerobic condition for simultaneous nitrification–denitrification) allowing nitrate accumulation in the final effluent.

Ammonia removal in WSP has been estimated by using the model developed by Pano and Middlebrooks (1982), which is one of the most widely accepted to describe nitrogen removal processes in facultative and maturation ponds. This model is based on first-order kinetics in a completely mixed reactor and reflects a statistically significant relationship between inlet and outlet ammonia concentration with pH, water temperature and hydraulic loading rate. This model attempts to explain ammonium removal by the increase in pH that usually occurs in stabilization ponds, which favours the predominance of ammonia on ammonium, which leaves the system. However, in our study, the high efficiency of ammonium removal observed was not accompanied with high pH (pH values higher than 8.1 were not registered). Therefore, it is suggested that an ammonium transformation pathway and/or removal process not strictly related to pH changes was contributing to ammonium removal (e.g. nitrification, nitrification–denitrification, algal nitrogen uptake).

4 Conclusions

In order to keep predominant facultative conditions and acceptable filtered BOD_5 removals, it was found that primary facultative ponds in a region with temperate climate from Argentine Patagonia should be loaded with an organic rate of up to 60 kg BOD_5/ha·day. The unfiltered BOD_5 removal was affected with the increase of flow and organic loading; there were other signs that may be considered as clear symptoms of poor treatment and operational conditions in WSP too, such as the prevalence of ammonium in the final effluent and low chlorophyll *a* and DO concentrations.

Important removal efficiencies for ammonium were registered during warm months in the primary facultative pond under study, and it is attributed to nitrification–denitrification processes and algal nitrogen uptake. The ammonium removal had a significant correlation with temperature (during summer and winter) and parameters associated with primary productivity (chlorophyll *a* and suspended solids). The increase in organic loading had direct effects on the phytoplankton population (resulting in a lower nutrient uptake), and it had an indirect effect on nitrifier bacteria activity (diminution of dissolved oxygen, bacteria attachment site and sun's rays protection), and as consequences, more exposure to washout of nitrifying bacteria and retarding their re-establishment in the water column with hydraulic loading increase. The higher ammonium removal was not linked with high pH values, and ammonium volatilization as a predominant removal process could be discarded.

Considering the evident dependency of nitrogen transformation and removal processes on weather conditions, nitrogen control in primary facultative pond effluents should be conducted by polishing units aimed to deal with high concentrations of organic nitrogen in warm periods (i.e. mainly from algal biomass) and high ammonium concentrations in cold periods.

Acknowledgements We especially thank Lic. Maria José Esteves and Tech. Hugo Fuhr from the Municipality of Puerto Madryn and the Public Services Cooperative (SERVICOOP), respectively. These institutions also provided financial support to this work, and we are grateful for their contributions.

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