SANITARY AND ENVIRONMENTAL ENGINEERING

# Greenhouse gas emissions from a pilot-scale small decentralized sewage treatment: anaerobic filter + constructed wetland

INGENIERÍA SANITARIA Y AMBIENTAL

# Emisiones de gases de efecto invernadero en un sistema descentralizado de tratamiento de aguas residuales domésticas: filtro anaerobio + humedal construido

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## Resumen

En este estudio se midieron las emisiones de gases de efecto invernadero i.e. $CH_4$, $CO_2$ y $N_2O$, producidas en un sistema descentralizado de tratamiento de aguas residuales municipales (SDST). El sistema consistió de un filtro anaerobio (UAF) seguido de dos humedales construidos operando en paralelo plantados con Heliconia sp L. (HCW) y C. papyrus. (PCW). Las emisiones medidas en el UAF(1.8 - 8.9 $g.m^{-2}.d^{-1}$ $CO_2$; 8.3 y 45 $g.m^{-2}.d^{-1}$ $CH_4$) fueron mayores a las medidas en HCW (-5.1 y 5.5 $gCO_2.m^{-2}.d^{-1}$; 10 y 556 $mgCH_4.m^{-2}.d^{-1}$; -4 and 40 $mgN_2O.m^{-2}.d^{-1}$) y PCW( -0.85 to 6.5 $gCO_2.m^{-2}.d^{-1}$; 2.3 and 1590 $mgCH_4.m^{-2}.d^{-1}$; -7 y 12 $mgN_2O.m^{-2}.d^{-1}$). Las pruebas estadísticas indicaron que para el mismo fotoperiodo (día o noche) no hubo diferencias entre HCW y PCW. Adicionalmente, las variaciones de GEI pudieron ser explicados por la temperatura del agua , el fotoperíodo, la presencia de $NH_4^+$ y la concentración de OD. El incremento en temperatura del agua estimula la producción de GEI. Igualmente durante el fotoperiodo el secuestro de $CO_2$ y $CH_4$ fue favorecido. La presencia de $NH_4^+$ y cambios en el OD en HCW y PCW estimularon la producción de $N_2O$. En conclusión, los resultados indicaron que el sistema descentralizado fue una fuente neta de GEI emitiendo 0.7 kg $CO_{2eq}$. Kg $COD_{rem}$.

*Palabras clave: Filtro anaerobio, gases efecto invernadero, humedales construidos, sistemas descentralizados de tratamiento de aguas residuales.*

## Abstract

Greenhouse gases such as $CH_4$, $CO_2$ and $N_2O$ were measured in a Small-Decentralized Sewage Treatment (SDST). The system consisted of an up-flow anaerobic filter (UAF) and two constructed wetlands (CWs) operating in parallel. The constructed wetlands were planted with Heliconia sp L. (HCW) y C. papyrus. (PCW). The results showed that GHG average (daytime and nighttime) emissions from UAF (1.8 - 8.9 $g.m^{-2}.d^{-1}$ $CO_2$; 8.3 y 45 $g.m^{-2}.d^{-1}$ $CH_4$) were higher than measured in HCW (-5.1 y 5.5 $gCO_2.m^{-2}.d^{-1}$; 10 y 556 $mgCH_4.m^{-2}.d^{-1}$; -4 and 40 $mgN_2O.m^{-2}.d^{-1}$) and PCW (-0.85 to 6.5 $gCO_2.m^{-2}.d^{-1}$; 2.3 and 1590 $mgCH_4.m^{-2}.d^{-1}$; -7 y 12 $mgN_2O.m^{-2}.d^{-1}$). A statistical test showed that there were not significant differences between GHG flux (daytime-nigh-time) measured in HCW and PCW ($p < 0.05$). Further, the variations on GHG could be explained by water temperature, photoperiod, $NH_4^+$ and dissolved oxygen mainly. High temperature stimulates GHG production. Photoperiod in CWS increased $CO_2$ and $CH_4$ sequestration. Further, the presence of dissolved oxygen and $NH_4^+$ influenced $N_2O$ production. Overall, in this study was determined that SDST act like a net source of GHG emitting 0.7 kg $CO_{2eq}$. Kg $COD_{rem}$..

*Keywords: Anaerobic filter, constructed wetlands, greenhouse gases, small and decentralized wastewater treatment.*

# 1. Introduction

The growing awareness of the impact of greenhouse gases such as $CO_2$, $CH_4$ and $N_2O$ from human activities on climate change triggers the need to identify and quantify the main sources of these gases. According to Forster et al. (2007), a wide range of direct and indirect measurements confirm that the atmospheric mixing ratios of dioxide carbon ($CO_2$), methane ($CH_4$), and nitrous oxide ($N_2O$) have increased globally over the last 250 years by 36, 250, and 18%, respectively due to anthropogenic activities such as production and use of fossil fuels, industrial and agricultural activities, and waste management among others. Thus, compilation of data covering these sectors is the basis for collective action on the reduction of anthropogenic GHG emissions (UNFCCC, 2007).

Small-decentralized sewage treatment systems (SDST), such as up-flow anaerobic filter (UAF) + constructed wetlands (CWs), have been suggested as efficient low-cost and low-tech options for sewage treatment mainly in developing countries (Parkinson & Tayler, 2003; Vymazal, 2005; Massoud et al., 2009). SDST use little or no electrical energy, are more appropriate than energy-intensive processes, such as activated sludge, and they are cheaper to construct, operate and maintain. However, SDST may generate secondary negative environmental impacts because they might generate greenhouse gases such as carbon dioxide ($CO_2$), methane ($CH_4$) and nitrous oxide ($N_2O$) related to the intrinsic metabolic processes that occur during wastewater treatment (Inamori et al., 2007; Søvik & Klove, 2007). According to this, has been suggested the necessity to get reliable data of these emissions for a precise assessment of the sustainability of wastewater treatment systems (Bogner et al., 2007; Foley et al., 2015)

Studies reporting Greenhouse flux data from SDST as whole i.e. UAF + Cws are hard to come across in the literature. There are an extensive research about GHG emissions from constructed wetlands but no in combination with up-flow anaerobic filter. Studies reporting GHG emissions from UAF are limited (Martin et al., 2010). The production of $CO_2$, $CH_4$ and $N_2O$ has been studied in different CWs (Teiter & Mander, 2005; Liikanen et al., 2006; Søvik et al., 2006; Søvik & Klove, 2007). Further, these studies were carried out under temperate conditions that involved measurements under spring and winter seasons.

Hence, this study is aimed at: (1) measuring the emissions of $CO_2$ and $CH_4$, from SDST operating under tropical condition (2) to study the influences of environmental factors that regulate the emissions i.e. temperature. A special emphasis was given to estimate the differences between emissions during daytime (higher solar radiation) and night-time (lower solar radiation) conditions, and finally, this research was also aimed at contributing to reduce the knowledge gap on GHG emissions field data from SDST.

# 2. Methodology

## 2.1 Site description and experimental set up

This study was carried out in the experimental research station for wastewater treatment of Ginebra, a small town of 10,000 inhabitants located in Southwest Colombia (3°43'25.98 N, 76°15'59.45 W). The historical average ambient temperature of this town is 26 o C and its altitude is 1,040 m above sea level.

The system studied was a small-decentralized sewage treatment system (SDST) treating sewage wastewater. This system consisted of an up flow anaerobic filter (UAF) connected to two horizontals subsurface flow constructed wetlands (CWs) operating in paralell (Fig. 1). The constructed wetlands were planted with *Heliconia sp* L. (HCW) y *C. papyrus*. (PCW). The system was constructed according to the characteristics shown in Table 1.

**Table 1.** *Design characteristics of small-scale decentralized sewage treatment located in Ginebra (Colombia).*

| Eco-technological System | Scale | Flow rate ($m^3 d^{-1}$) | HRT (d) | Depth (m) |
|---|---|---|---|---|
| Up-flow anaerobic filter (UAF) | Pilot | 1.36 | 0.5 | 1.6 |
| Constructed wetland C. *papyrus sp.* (PCW) | Pilot | 0.68 | 2.2 | 0.6 |
| Constructed wetland *Heliconia sp* L. (HCW) | Pilot | 0.68 | 2.2 | 0.6 |
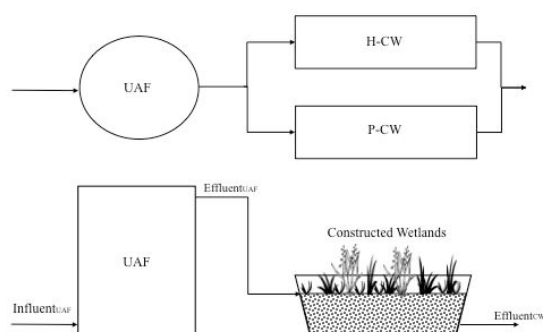


**Figure 1.** *Schematic representation of experimental set up; UAF= Upflow Anaerobic Filter; H-CW: Heliconia sp L. (HCW) y C. papyrus. (PCW)*

## 2.2 Wastewater sampling

The influent and effluent wastewater quality in both UAF and CWs was determined through 24-hour sampling campaigns carried out weekly. The day was divided in periods of eight hours. During each period a composite sample was collected by taking every hour a fixed volume of sample and add these together for analysis. Additional wastewater grab samples were taken in the central point of CWs, at the spot where GHG fluxes were measured, to determine the correlation between GHG fluxes and wastewater characteristics.

Chemical oxygen demand (COD), total suspended solids (TSS), alkalinity, total Kjeldahl nitrogen (TKN), ammonium nitrogen (N-$NH_4^+$), nitrate nitrogen (N-$NO_3^-$) were measured according to Standard Methods (APHA, 2005). Conductivity, pH, dissolved oxygen (DO), temperature and oxidation-reduction potential were measured with electrodes.

## 2.3 Greenhouse gas measurement

Greenhouse gas emissions ($CH_4$, $CO_2$ and $N_2O$) were measured in situ weekly during 6 months by using the technique of closed static chambers (Silva et al., 2015). GHG emissions from UAF were measured using a cylindrical acrylic transparent chamber with a diameter of 0.30 m height of 0.80 m and a volume around 21.4 l. On the other hand, to measure GHG emissions in CW was used a large chamber of 0.4 diameter, 0.9 m height, and 113 l volume. These dimensions permit contain whole plant body. In the constructed wetlands the chamber was fixed on a cylindrical base (D=0.4m), which was inserted to 10 cm depth and sealed by a water-filled ring on the slag surface. The chambers were installed only at the times of measurement.

The gas samples were measured at two different times of the day: between 12 am and 2 pm (day-time) and 10 pm and 12 pm (night-time). In UAF GHG samples were taken during 12 minutes at 4-minute intervals (0,4,8,12) whereas in CW the samples were

taken during 45 minutes at 15-minute intervals (0,15,30,45). The gas samples were taken out with syringes, which were withdrawn directly through a needle into a pre-evacuated vial.

These gas samples were subsequently analyzed. $CO_2$ was measured by an infrared spectrophotometer Qubit S151 $CO_2$ analyser (Loligo Systems, Denmark) using 75 ml min$^{-1}$ air as the mobile phase with a temperature of the injector set equal to the ambient temperature. $CH_4$ and $N_2O$ were analyzed by gas chromatography (Shimadzu Co., Japan) equipped with a flame ionic detector (FID) and electron capture detector (ECD).

The fluxes were calculated using equation 1 from linear and no linear changes in the gas concentration in the chamber headspace (Silva et al., 2015).

$$F = \frac{dC}{dt}_{t=0} \; x \; \frac{Vc}{A} \; x \; \frac{1440}{d} \qquad (1)$$

*F= the flux of $CH_4$, $CO_2$, $N_2O$ (gm$^{-2}$d$^{-1}$); dC/ dt$_{-0}$ = slope of the gas concentration curve (gm$^{-3}$min$^{-1}$); $V_c$= volume of the chamber (m$^3$); A=the cross sectional area of the chamber (m$^2$).*

### 2.4 Data analysis

All statistical analyses Statistical were performed using SPSS® software (v. 17.0 for Windows). The Kolmogorov-Smirnov test was used to check normality of data. Further, an ANOVA analysis and Wilconson test ($\alpha$ = 0. 05) were used to compare the differences between daytime and night-time fluxes.

## 3. Results and discussion

### 3.1 Wastewater characteristics

Average COD and TKN concentrations measured in the influent were 333.1±110 mg COD.l$^{-1}$ and 40.2±4.28 mg TKN.l$^{-1}$. The removal efficiencies for COD and TKN in of SDST (UAF+CW) were 74 and 31.3 % respectively (Fig. 2). According to the statistical analysis there were no significant differences between HCW and PCW related to

COD removal, TKN and $NH_4^+$-N (p<0.05). As shown, COD removal was lower in UAF than obtained in HCW or PCW. On the other hand, TKN coming from UAF was removed by 31.3±10.1 and 26.2± 8.3 % for HCW and PCW respectively. In addition, $NH_4^+$-N was removed in a 20.8±8.4 and 27.1±5.1 % for HCW and PCW respectively (Fig. 2). In UAF probably formed $NH_4^+$-N was due to organic nitrogen hydrolysis in the effluent.
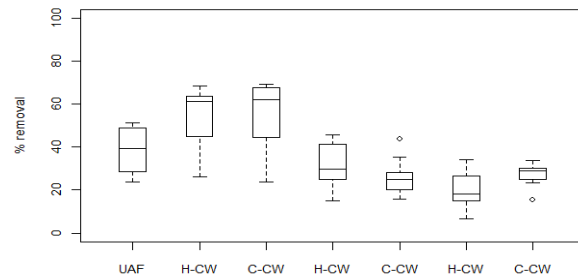


**Figure 2.** *Average removal efficiencies of COD, NTK y $NH_4^+$-N in the Up-Flow anaerobic filter and Constructed Wetlands (H-CW and P-CW).*

### 3.2 Greenhouse gas emissions from anaerobic filter

Figure 3 shows $CH_4$ and $CO_2$ fluxes measured in UAF during the monitoring campaign. As shown, UAF acted like a permanent source of $CH_4$ and $CO_2$. $CO_2$ fluxes during daytime ranged between 1.8 and 8.9 g.m$^{-2}$.d$^{-1}$ (median=5.3; SD=2.3; n=20) while at night-time they ranged from 1.8 and 4.1 g.m$^{-2}$.d$^{-1}$ (median=3.2; SD=0.8; n=20). According to the statistical test there were not significant differences between $CO_2$ produced in UAF during daytime and night-time (p<0.05). On the other hand, $CH_4$ fluxes for the daytime period ranged between 13 and 45 g.m$^{-2}$.d$^{-1}$ (median= 25.7; SD = 12.9; n= 20) while at night-time were ranging between 8.3 and 19.8 g.m$^{-2}$.d$^{-1}$ (median= 13.3; SD= 4.1; n=20). Comparison of $CH_4$ released fluxes in UAF for the periods day-time and night-time through the statistical test, showed significant differences in the emissions of this gas for both periods (p=0.026). Regarding to nitrous oxide the emissions in UAF were not detectable under experimental conditions.
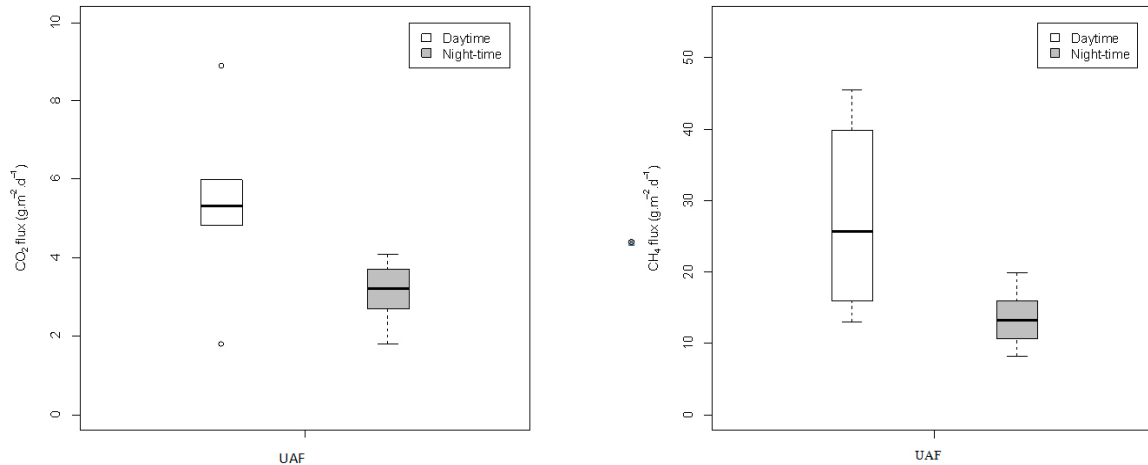
**Figure 3.** *Greenhouse gas fluxes (a) CH₄ y (b) CO₂ from Up-Flow Anaerobic Filter during daytime and night- time periods.*

The average emissions from UAF obtained in this study were lower than those reported in the literature for anaerobic lagoons operated under the Mediterranean climatic conditions (Toprak, 1995; Picot et al., 2003), tropical conditions (Silva et al., 2011) and operating at high organic loads (DeSutter & Ham, 2005; Yacob et al., 2005; Hasanudin et al., 2006; Konaté et al., 2013).

According to the amount of $CH_4$ produced and the COD removed was calculated that in UAF were produced 0.09 $m^3CH_4.kg_{COD\ removed}^{-1}$. This value was higher than the value of 0.06 $m^3CH_4.kg$ $COD^{-1}$ reported in an UAF treating low strength domestic wastewater and operating under similar conditions of OLR and temperature (Kobayashi et al., 1983), but lower than those observed in an UAF treating synthetic domestic sewage (0.15 $m^3CH_4.kg\ COD^{-1}$) operating under psychrophilic conditions (15-17 o C)(Martin et al., 2010).

Results of statistical test indicated that the organic load rate (OLR) and water temperature influenced $CH_4$ fluxes ($R^2=0.79$) while changes in $CO_2$ emissions were attributed to water temperature ($R^2=0.49$). The positive correlation found among water temperature and GHG fluxes has been also observed in low strength wastewater anaerobic treatment (Langenhoff & Stuckey, 2000; Lettinga et al., 2001). An increase of temperature favours the anaerobic process by increasing the anaerobic

bacteria activity and therefore $CH_4$ and $CO_2$ emissions are increased. In anaerobic treatment systems operating under high a temperatures i.e. tropical conditions is expected a higher GHG production than systems under low temperature i.e. psychrophilic. However, the temperature not is the only operational variable that control GHG production. This can be observed in the fact that the emissions obtained from our study (Tw=27 C) were lower than reported in anaerobic reactor operating under psychrophilic conditions (Martin et al., 2010). This suggests that another operational factor such as the organic load rate also influence GHG production in anaerobic treatment systems. The low OLR (0.7 kg $COD.m^{-3}.d^{-1}$) applied to UAF limited the concentration of active bacterial biomass in the reactor that influencing the low COD removal observed and decreasing in the $CH_4$ production (Chiang & Dague, 1992).

### 3.3 Greenhouse gas fluxes from HCW and PCW constructed wetlands

*CO₂ Emissions*

The fluxes of $CO_2$ from CWs measured during day-time and nigh-time were significantly different (p=0.041). Carbon dioxide fluxes in HCW (Fig. 4) during day-time ranged between -5.1 and 0.3 $gCO_2.m^{-2}.d^{-1}$ (median= -1.5; SD=1.6;n=20), while at night-time were measured values between 1.7 y 5.5 $gCO_2.m^{-2}.d^{-1}$

(median=2.4;SD=1.3;n=20). On the other hand, for PCW at day-time $CO_2$ fluxes were ranging from -0.85 to 5.1 gCO2.m$^{-2}$.d$^{-1}$ (median=-2.1;SD=1.4; n=20) whereas at night-time the flux ranged between 2 and 6.5 gCO$_2$.m$^{-2}$.d$^{-1}$ (median=3.5;SD=1.4;n=20).

These differences between daytime and night-time $CO_2$ emissions suggest that during the day the CWs systems behave as $CO_2$ sinks while at night they produced this gas.
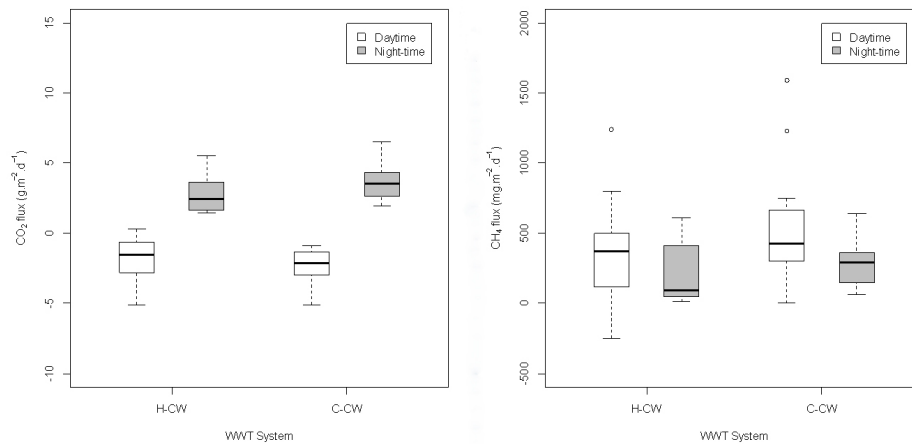


***Figure 4.*** *Dioxide Carbon fluxes from HCW and PCW for day-time and night-time.*

The significant difference observed for the $CO_2$ fluxes measured during day-time and night -time periods is due to the different dynamic between photosynthesis and respiration processes. During day-time there is a higher solar radiation favouring the photosnthetic activity of the plants. Because, during this period the plant capture $CO_2$ the CWs act as sink of this gas. During night time the plants swicht to respiration process and $CO_2$ is emitted. This $CO_2$ in addition to produced by heterotrophic bacteria in the CW is release into the atmosphere, thus in this period the CWs behave as source. This dynamics suggests that $CO_2$ emisisosn from CWs operating under tropical conditions are influenced by the photoperiod.

Carbon dioxide from different CWs has been reported in the literature; however, most of them in their GHG measurements have not taken into account the photosynthesis process due to utilization of opaque static chambers (Teiter & Mander, 2005; Liikanen et al., 2006; Ström et al., 2007). This suggests that it is only possible to compare the night-time flux of $CO_2$ obtained in this study with those reported in the literature. The results obtained in Ginebra CWs were in the range

reported for CW treating municipal wastewater (Teiter & Mander, 2005; Picek et al., 2007) but were lower than those observed for a CW planted with Typha in Sweden, treating domestic wastewaters (Ström et al., 2007) and to the reported in a CW treating mining runoff (Liikanen et al., 2006).

According to the statistical analysis the flux of $CO_2$ from CWs was correlated to the water temperature (50.2%), environment temperature (27.2%) and water pH (10.3%). Thus, high water and environment temperatures regulated by tropical conditions increased $CO_2$ emissions from CWs. The pH contributes to capture $CO_2$ and to reduce the emisisons of this gas in the CWs. Under alkaline conditions $CO_2$ can be converted to bicarbonate alkalinity and probably this could favour less $CO_2$ emissions. On the other hand, decreasing pH (6.8-7.2) lead to release more $CO_2$ into the atmosphere.

*$CH_4$ emissions*

In the Figure 4 are shown the methane flux from CW measured during daytime and night-time periods. In general, the emissions of CH4 in HCW were ranging during daytime between – 248 and

1240 mg $CH_4.m^{-2}.d^{-1}$ (median=375; SD=428.9; n=12). These emission were higher than those measured for the night-time period which were ranging between 10 and 556 $mgCH_4.m^{-2}.d^{-1}$ (median=95; SD=222; n=12). In PCW the fluxes during the day-time varying between 2.3 and 1590 $mgCH_4.m^{-2}.d^{-1}$ (median=430; SD=448; n=12) while during night-time ranged from 80 to 640 $mgCH_4.m^{-2}.d^{-1}$ (median= 289.5; SD=162.7; n=12). On the other hand, the statistical test showed that there were no significant differences between the fluxes measured in both constructed wetlands (p=0.132). Therefore, $CH_4$ emissions for both wetlands considered at the current study were similar.

Methane emissions observed in this study were comparable to those observed in CW operating under subtropical climatic conditions treating domestic wastewaters (Johansson et al., 2004; Picek et al., 2007). Maximum flux values found were compatible with those found in studies with CW treating agricultural wastewater (Tanner et al., 1997; Vander Zaag et al., 2010). In addition, negative $CH_4$ flux values found in this study, coincided with those observed in other studies (Johansson et al., 2004; Teiter & Mander, 2005; Søvik & Klove, 2007; Ström et al., 2007). The flux of methane is the net difference among production and oxidation of this gas in the CW. A positive flux value of $CH_4$ indicates that there is generation of this gas, while a negative value indicates that there was consumption. Thus, $CH_4$ negative fluxes obtained in the CW under study, indicate that probably this gas was consumed which may be attributed to oxidation of methanotrophs that favours the oxidation of $CH_4$ (Inamori et al., 2007; Wang et al., 2013).

The correlation between environmental parameters and methane fluxes indicate that production of this gas in both CWs was mainly influenced by the water temperature (67.8%) and in a lesser proportion by the organic load applied (8.9%) and pH (7.4%). The high temperatures under tropical conditions increase the microbial activity and GHG were increased. The influences of temperature on GHG emissions have been also observed in different CW

from Europe. The $CH_4$ emissions were higher during high temperature seasons (summer) that in low temperatures (winter)(Tanner et al., 1997; Johansson et al., 2004; Søvik et al., 2006; Picek et al., 2007; Vander Zaag et al., 2010). As reported Johansson et al. (2004) 33-43% of $CH_4$ flux variations were attributed to water temperature and sediments. Likewise, in a HSSF-CW treating municipal wastewater in Estonia, sediment temperature was the variable that showed a correlation positive related to $CH_4$ emissions (Teiter & Mander, 2005). By contrast, the results obtained on a CW treating mining runoff suggest that soil and water temperatures poorly explained the emission of GHG (Liikanen et al., 2006). In addition, the $CH_4$ fluxes were also positively correlated to the OLR. A higher load of organic matter (BOD, COD, TOC) contributes to provide larger substrate decomposition and to increase of more reducing zones in CW, favouring production of $CH_4$ (Gui et al., 2007; Inamori et al., 2007; Picek et al., 2007; Uggetti et al., 2012). Therefore the OLR influence methane production in CWs.

*N$_2$O emissions*

In general $N_2O$ flux values indicated that constructed wetlands become an emitting source of this gas (Fig. 5). In HCW during day-time $N_2O$ flux ranged between -16 and 32 $mgN_2O.m^{-2}.d^{-1}$ (median= 9.1; SD=11; n=20), while for night-time the flux remained between -4 and 40 $mgN_2O.m^{-2}.d^{-1}$ (median=3.8; SD=12; n=20). On the other hand, in PCW for the daytime period the $N_2O$ fluxes were ranging between -7 and 9 $mgN_2O.m^{-2}.d^{-1}$ (median= 3; SD=4;n=20) whereas that at the night-time remained between 0 and 12 $mgN_2O.m^{-2}.d^{-1}$ (median=3;SD=3.8;n=20). It is important to clarify that negative nitrous oxide in this study was detected only once during the monitoring campaign. According to the statistical test there was no difference between the $N_2O$ fluxes measured for daytime and night-time in both constructed wetlands. In addition, using the removed mass from TKN as N removal indicator it was calculated that those of ratio N2O-N/TKN for H-CW and C-CW were 0.3% y 0.1% respectively.

107

The $N_2O$ fluxes from Ginebra CWs were in the range reported for FWS-CW in Sweden (Johansson et al., 2003; Ström et al., 2007), HSSF in Norway (Søvik & Klove, 2007), Estonia (Teiter & Mander, 2005), Czech Republic (Picek et al., 2007) and Finland (Liikanen et al., 2006; Søvik et al., 2006). However, in CW treating dairy wastewater (Tanner et al., 1997; Vander Zaag et al., 2010) and municipal wastewaters (Inamori et al., 2007) lower values of $N_2O$ were obtained compared with CWs studied at the current research. On the other hand, $N_2O$ flux values reported by Uggeti et al. (2012) for a CW treating waste activated sludge were 20 times higher than the estimated in this study.

$N_2O$ in constructed wetland is mainly produced by nitrification, denitrification, and denitrifier denitrification (Wrage et al., 2001): (i) Nitrifiers i.e. ammonia-oxidizing bacteria (AOB) produce $N_2O$ mainly by the incomplete oxidation of hydroxylamine and by nitrifier denitrification. In nitrifier denitrification, the oxidation of $NH_3$ to $NO_2^-$ is followed by the reduction of $NO_2^-$ to $N_2O$ and $N_2$. In the case of nitrification the $N_2O$ is formed during incomplete $NH_4^+$-N oxidation to $NO_3$—N by low DO concentration (Zeng et al., 2003). (ii) Denitrifiers produce $N_2O$ as an intermediate possible end product of the reduction of $NO_3^-$-N to $N_2$. Nitrous oxide is formed by incomplete denitrification related to the availability of nitrate (Søvik et al., 2006), increase of oxygen in water (Von Schulthess et al., 1994), or low COD/N ratio (3.5) (Hanaki et al., 1992).
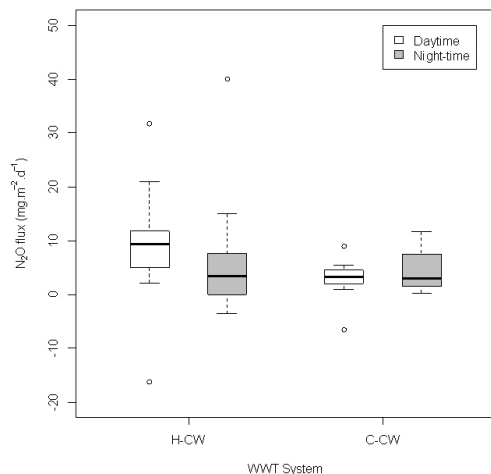


***Figure 5.*** *Nitrous Oxide fluxes from HCW and PCW for day-time and night-time.*

According to statistical test $NH_4^+$ could explain 23.2 % on the variation of the $N_2O$ emitted from CWs. Removal of $NH_4^+$ in both HCW and PCW suggests that nitrification and subsequent denitrification were carried out in these systems. During nitrification AOB bacteria, which are the dominant microorganisms of the rhizosphere (Huang et al., 2013) may produce NO and $N_2O$ either as a side-product in the catabolic pathway (oxidizing ammonia to nitrite), or, alternatively, by denitrification of nitrite with ammonia, hydrogen or pyruvate as electron donor (Colliver & Stephenson, 2000; Wrage et al., 2001; Law et al., 2012). In addition, the $NO_3^-$ formed during nitrification may be reduced to $NO_2$ by incomplete denitrification due to the presence of some amount of oxygen in the wetland. In this study both CW's was negatively correlated with dissolved oxygen explaining 22% of $N_2O$ variation. As reported in conventional wastewater treatment systems the $N_2O$ formation via denitrification increases when DO levels are increased (Von Schulthess et al., 1994). For instance, it has been reported that a concentration of DO around 0.09 mg.l$_{-1}$ decreased the denitrification rate by 35% and under DO concentrations around of 5.9 mg.l$^{-1}$ practically the denitrification fell down to zero (Oh & Silverstein, 1999). Summarizing, the formation of $N_2O$ in HCW y PCW probably was influenced by both the nitrification and denitrification process. The temperature could explain 34.3 % on the variation of the $N_2O$ fluxes.

In this study the CWs were operated under tropical conditions characterized by high temperatures, which probably favored more $N_2O$ production. The influence of the temperature on $N_2O$ fluxes from CWs was observed in different CW operating in Europe (Johansson et al. 2003; Teiter & Mander 2005; Liikanen et al. 2006; and Søvik & Klove 2007). In this studies was reported that low temperatures slow down the nitrification process since the activity of ammonia oxidizing bacteria decrease and also the denitrification is limited.

### 3.4 Overall GHG emissions from small-decentralized system (UFA + CW)

The greenhouse gas emissions in terms of grams of $CO_2$eq m$^{-2}$ d$^{-1}$ are reported in Table 3. According

to the $CO_2$e calculated, methane is the GHG that offers more contribution to GWP in the small-decentralized wastewater treatment system. In general, on the basis of $CO_2$e, $CH_4$ represents 99% of total emissions of the whole treatment. $CH_4$ to $CO_2$ ratio for the whole treatment system was 91 while the ratio of $CH_4$ to $N_2O$ was 178.9. Moreover, calculation of ratios $CH_4/ N_2O$ for H-CW and C-CW reported average values of 8.3 and 3.1 respectively. This suggests that upon combining anaerobic technologies and systems such as the CW (Hasanudin et al., 2006) is the major ecological footprint is attributed to the GHG generated in the anaerobic i.e. $CH_4$.

**Table 3.** *Summary of greenhouse gas emissions converted to carbon dioxide equivalent global warming potential using methane ($CH_4$=25) and nitrous oxide ($N_2O$=298).*

| System | $CO_2$ | $CH_4$ emission | $N_2O$ emission | Total |
|---|---|---|---|---|
| | | ($gCO_{2eq}$ $m^{-2}$ $d^{-1}$) | ($gCO_{2eq}$ $m^{-2}$ $d^{-1}$) | ($gCO_{2eq}$ $m^{-2}$ $d^{-1}$) |
| UFA | 4.3 | 487.5 | n.d | 491.8 |
| C-CW | 0.5 | 7.5 | 0.9 | 8.4 |
| H-CW | 0.7 | 5.9 | 1.9 | 7.8 |

Comparing the $CO_2$e for two wetlands, it is concluded that there was no significant difference in their impact to GWP. Moreover, upon comparing carbon and nitrogen rates of transformation to $CH_4$ and $N_2O$ in HCW y PCW, these were lower. The foregoing suggests that only a small amount of carbon and nitrogen removed is transformed to $CH_4$ minimizing the generation of this GHG in CWs studied. The 3% value of carbon transformed into $CH_4$ obtained in this study was lower than reported for different European wetlands (Søvik et al., 2006; Søvik & Klove, 2007). On the other hand, the ratios of $N_2O$-N emissions to TKN removal are within the range of that reported by different studies in CW (Johansson et al., 2003; Søvik et al., 2006; Søvik & Klove, 2007).

**4. Conclusions**

This study has provided information about GHG emissions from decentralized wastewater treatment system operated under tropical conditions. According to the results, SDST was a net

source of $CO_2$, $CH_4$ and $N_2O$, having to UAF as the treatment unit that offers a higher contribution to GWP than constructed wetlands. Thus, reducing the environmental impact of UAF involve capture and recover of methane via energy production or flared to avoid its release into the atmosphere. On the other hand, the GWP determined for CWs suggest that this technology had lower environmental impact than anaerobic technology.

A significant difference between $CO_2$ fluxes measured in CW during day-time and night-time periods was observed in this study. This suggest that $CO_2$ emisisosn from CWs operating under tropical conditions are influenced by the photoperiod. Thus, during photo period CW are net sink and therefore these systems can act as regulators of atmospheric $CO_2$ reducing their environmental impact(GWP).

A similar pattern due to the photoperiod was observed in CW related to methane production. The oxygen translocated into the roots probably inhibited methane release into the atmosphere via methane oxidation by methanotrophs. Based on results, a next step would be to adress experimental setup to determine the influence on methane emissions due to methane oxidation process.

The results found adress the necessity to follow researching about the real impacts of wastewater treatment systems from perespective of their GWP contribution.
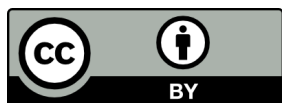
**5. Acknowledgements**

**6. References**

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