

Treated wastewater viability of use in green areas according to nitrogen compounds concentration

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Abstract

Treated wastewater is commonly used for park irrigation in arid zones of Mexico without considering groundwater contamination. The objective was to investigate the possibility of nitrogen compounds leaching into the groundwater and their subsequent reactions in the main types of soils. Lysimeters samples were taken to scrutinize the soil characteristics of the green areas irrigated with treated wastewater from the Northern Wastewater Treatment Plant in the city of Chihuahua. Testing samples were setup to recreate treated wastewater irrigation conditions. Nitrogen-based compounds were identified and measured before and after percolation through the soil columns. Based on the results, one meter of sand column was sufficient to remove 68 to 100% of nitrogen compounds present in the residual water. The removal of all nitrogen-based compounds as they percolate through one meter of clay soil column was not enough, due to the biochemical reactions that occur through the percolation process. Results indicate minimal risk of nitrate and nitrite leach into the aquifer, since the average static level depth is 20 m which provide broad filtration. This demonstrates an opportunity for *in-situ* investigations to reevaluate the standards for soil aquifer treatment recharge, based on the soil type and water quality of the area.

Key words: artificial recharge, nitrate, wastewater treatment

INTRODUCTION

There is a constant need around the world to meet the increasing water demand. The reutilization of non-potable water is an effective solution for sparing drinkable water, especially in regions with water scarcity like the northern part of Mexico (Bixio *et al.* 2005). The benefit is derived from the use of treated wastewater (TW) in activities where potable water quality is not required, allowing drinkable water to be spared, and satisfy the population's growing demand. Additionally, use of TW for agriculture and gardening can lead to a reduction or elimination of fertilizer applications (Heinze *et al.* 2014). Excessive infiltration of irrigation water can introduce contaminants (e.g. nitrate) to shallow groundwater (Böhlke 2002; Brown *et al.* 2011). Ammonia (NH₃) is one of this contaminants that might produce numerous adverse effects on the environment (Jana 1994), it is recycled naturally in the environment as one of the steps of the nitrogen cycle. Because of its reactivity, NH₃ does not last long in its pure form (Agency for Toxic Substances & Disease Registry 2015). In the transformations of NH₃ to nitrate (NO₃), the quality of stored water can be affected (Power *et al.* 2000). NO₃ is currently known as the main source of diffuse contamination of surface and groundwater

in some environments (Majumdar & Gupta 2000; Laftouhi *et al.* 2003; Widory *et al.* 2004; Moore *et al.* 2006; Murgulet & Tick 2009; Ako *et al.* 2014). NO_3 itself is relatively nontoxic, its toxicity is determined by its reduction to NO_2 in the human body, which in high concentration can lead to methemoglobinemia (Camargo & Alonso 2006).

The possible percolation of nitrogen compounds to groundwater is produced due to the induced recharge from the non-visible leaks of hydro-sanitary networks and the irrigation of green areas with TW. Several studies have correlated land use/cover and environmental variables in the soil's capture zone with variations in water quality in sampling stations, using different statistical modelling approaches (Nkotagu 1996; McLay *et al.* 2001; Spruill *et al.* 2002; Gardne & Vogel 2005; Kaown *et al.* 2007; Stigter *et al.* 2008; Mfumu Kihumba *et al.* 2016). Teng *et al.* (2018) for instance, applied a multivariate analysis of the hydraulic connection between the superficial and groundwater, analyzing the hydrogeochemistry and pollutant data, to evaluate their interaction, they were able to prove that the degree of hydrogeochemical response depends on the different types of hydraulic connection; and therefore interaction process, dilution, adsorption, redox reactions, nitrification, denitrification, and biodegradation contributed to the concentration of pollutants.

The water quality degradation has become more urgent due to the close hydraulic relationship between irrigated water and groundwater (Teng *et al.* 2018). However, to date, the effects of TW interaction on percolated water quality have not been evaluated in the city of Chihuahua.

The objective of this study was to assess the possibility of nitrogen compounds leaching through the main types of soils on the green areas irrigated with TW in Chihuahua. The use of multivariate methods in this study is based on the fact that the groundwater nitrogen compounds concentration can be influenced by several independent environmental variables, considering mainly the type of soil. Multivariate statistical models allow an understanding of which terms have different response (Mfumu Kihumba *et al.* 2016). The results will shed light to perform a proper a MAR (Managed Aquifer Recharge) in Chihuahua-Sacramento aquifer (CHSA).

MATERIAL AND METHODS

The city of Chihuahua is located in the northern part of Mexico between $28^{\circ}50'$ to $28^{\circ}30'$ North latitude and $106^{\circ}12'$ to $105^{\circ}50'$ West longitude, it is the capital of the Chihuahua state, and the second most populated city of the region. The climate is arid to semiarid, averaging a 415 mm annual rainfall. Due to high evapotranspiration and low infiltration, aquifer recharge is not promoted. The city relays almost 100 percent in groundwater to satisfy the drinking water demand, resulting in severe water stress in the different aquifers. Chihuahua city depends mainly from the aquifers: CHSA, Tabaloopa-Aldama and Sauz-Encinillas. The continuous pumping has caused declines in the water table of CHSA from 2 to 3 meters annually (National Water Commission 2010). Green areas above CHSA have been irrigated with TW from the North Treatment Plant (NTP) for more than 20 years according to the municipal water and sanitation board. The NTP gardens were chosen as a representative area of the green areas irrigated with TW, where the TW irrigation began (Figure 1).

Lysimeters were used as the experimental units (EU) for this study (Heinze *et al.* 2014). Lysimeters were designed to directly measure a drainage flux, leading to a representation of the recharge process (Allen *et al.* 1991; Pakrou & Dillon 2000; Walker *et al.* 2002). Water, vegetation, soil and rock material were characterized to establish the EU representativeness of the experimental site (ES). Besides the parameters previously mentioned, meteorological variables (temperature, wind, humidity) were evaluated in a climatological station installed by the Center for Advanced Materials Research (CIMAV) (Figure 2). Precipitation was not allowed to interfere with the different types of irrigation water in the experiment.

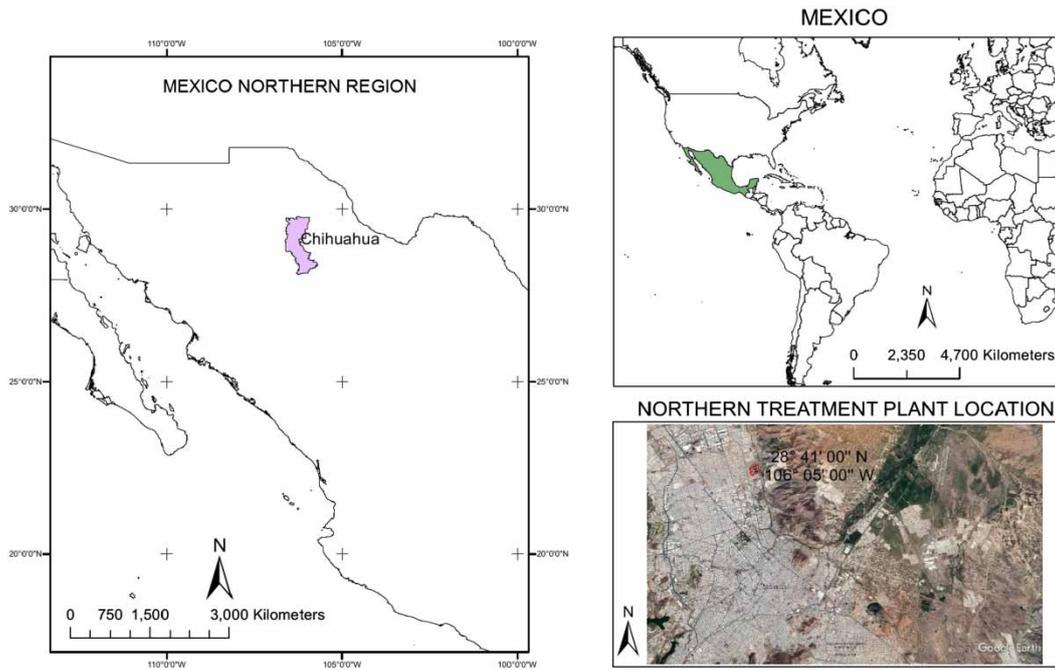


Figure 1 | Green areas irrigated with TW in Chihuahua city.

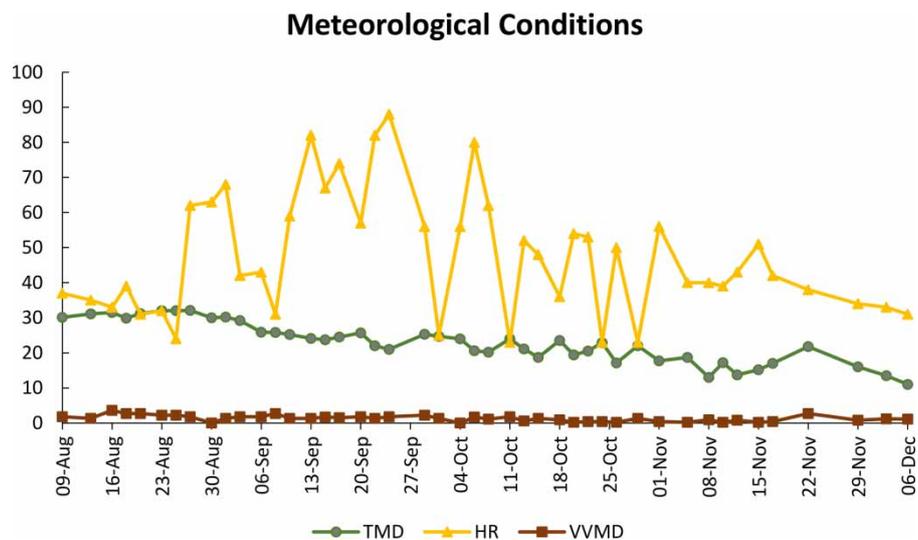


Figure 2 | Meteorological conditions in the climatological station. TMD stands for daily mean temperature in °C, HR stands for daily mean relative humidity in %, as VVMD stands for daily mean wind velocity in km/hr.

Soil characterization was first made by vertical electric sounding (VES). The VES readings were correlated with existing lithological cuts from nearby wells, delivering geological interpretation of the NTP subsoil. The subsoil, composed of a series of granular units, corresponds to an old bed of the Sacramento River. Four points samplings of the ES were characterized by the Agricultural Union of Fruit Growers of the State of Chihuahua (UNIFRUT) laboratory, in order to comply with NOM-014-CONAGUA-2003 regulation. NOM-014 establish the requirements to evaluate a site for MAR in Mexico.

The lysimeters were packed with representative soil of the ES, in which unaltered samples up to 2 m depth were taken. Each sample was analyzed in order to obtain their granulometric components, orientation, compaction, and moisture content. The presence of nitrogenous components was also analyzed in the soil matrix.

Based on the results from the laboratory analyses, two soil horizon sets were classified: the first set of horizons of 15 and 40 cm depth and the second set corresponds to: 80, 100 and 130 cm depth (Table 1).

Table 1 | Physicochemical properties of the NTP soil

Physicochemical properties	Soil horizon (15–40 cm)	Soil horizon (80–100–130 cm)
Electric conductivity (mm/cm)	2.16	0.76
NO ₃ (Kg/Ha)	267.3	23.4
Phosphorus (ppm)	54.53	11.15
Organic matter (%)	15.93	1.174
CaCO ₃ (%)	0.0	0.0
Moisture (%)	17.17	13.23
pH	7.4	8.69
Na (ppm)	464	440
K (ppm)	1,452	208
Fe (ppm)	7.96	9.36
Total N (ppm)	0.679	0.058
Sand (%)	64.44	42.44
Silt (%)	25.84	29.84
Clay (%)	9.72	27.72
Hydraulic conductivity (cm/hr)	13.42	1.25
Texture	Sandy loam	Sandy loam

The soil hydraulic conductivity was also measured, which is an intrinsic property of the soil matrix and the fluid (Scanlon *et al.* 2002). The infiltration capacity was also obtained, in order to evaluate the volume of water that can be infiltrated (Scanlon *et al.* 2002). The infiltration capacity was obtained in the ES through infiltrometers, allowing the identification of hydraulic conductivity to be done through the excavation of soil horizons, in which exhaustion and recovery tests are carried out (Walker *et al.* 2002). Similar values were obtained for the horizons of 15 and 40 cm of 2.1 and 2.2 cm/h respectively. The horizons of 80, 100 and 130 cm delivered values of: 2.8, 2.0 and 0.8 cm/h respectively. This all defined a sandy behavior, according to infiltration rates established by the Environmental Protection Agency (EPA). The vegetation samples used as representation were obtained by doing a set of four individual pieces, collected in a single composite sample from the ES, in accordance to the method described by Brady & Weil (2002). This composite sample was sent to the laboratory specialized in edaphology from UNIFRUT.

Experiment design

The drinking water quality was used as the control, to compare it with the TW. Drinking water and TW infiltrated in two types of packing and filling material in lysimeters. Lysimeters one and two were the basis for the integrated approach; the experiment design resulted in the construction of four EU. Each lysimeter was identified by a number (L1 or L2) corresponding to the water quality of irrigation, L1 corresponding to drinking water and L2 to TW. Letters S and C (sand and clay) were used to identify the type of soil in the packaging.

This experiment was based on the transformations of nitrogen in the soil-water interaction. The process sought to quantify the transformation of the main inorganic nitrogenous components, as well as defining if the climate factor has an impact in these transformations. This was raised with the conceptual model in Figure 3.

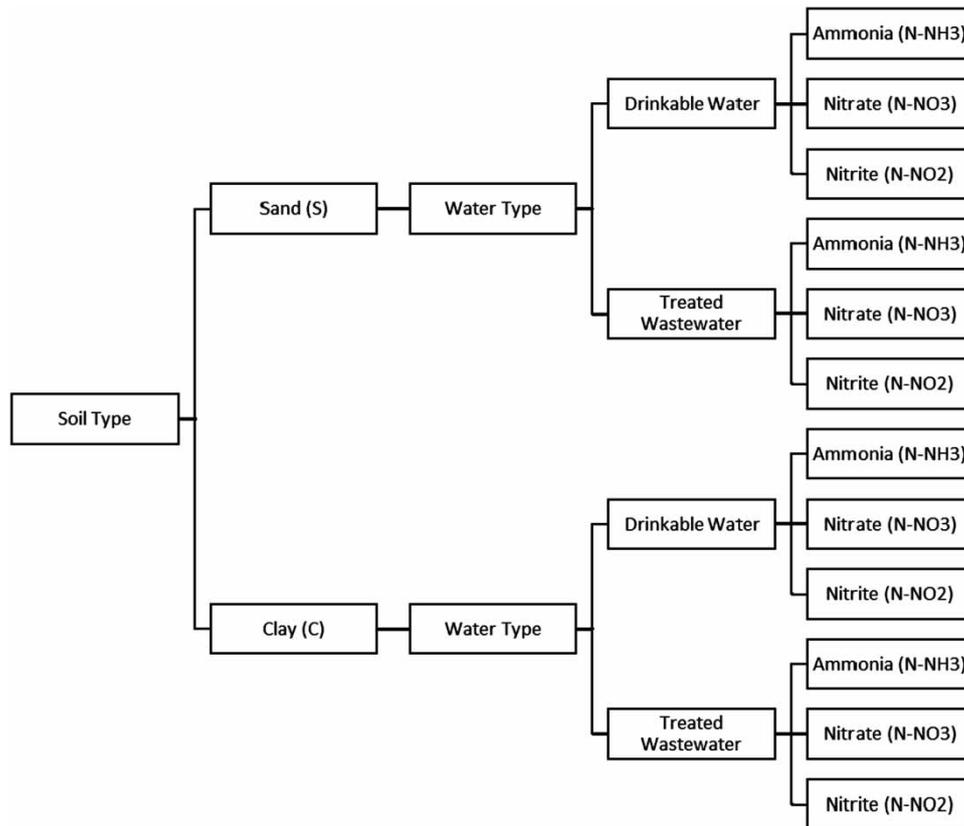


Figure 3 | General diagram of process.

The lysimeter's geometric design was the same used by [Bojorquez *et al.* \(2018\)](#). Each lysimeter measured 50 cm × 50 cm × 100 cm, it was made of acrylic, and had a thickness of 10 mm. They were comprised of a steel structure with three stirrups to prevent thrust during experimentation. Each lysimeter was connected to a supply water tank (drinking water or TW). They also featured an irrigation simulator, which had a valve to control the irrigation rate. In addition to this, the lysimeters had acrylic covers on top of them, to prevent the infiltration of rainfall during the rain season.

The techniques and equipment used to evaluate the water samples at the entrance and exit of the lysimeters are shown in [Table 2](#).

Table 2 | Analyzed parameters entering and evacuating the lysimeters

Parameter	Technique	Norm
Turbidity	Nephelometric colorimeter method	NMX-AA-038-SCFI-2001
pH	Portable potentiometer Corning no. 180225	NMX-AA-008-SCFI-2000
Temperature	Mercury thermometer	NMX-AA-007-SCFI-2000
Electric conductivity	Electrometric conductivity meter	NMX-AA-093-SCFI-2000
Total Dissolved Solids, TDS	Gravimetric.SM-2540-C-1998	NMX-AA-034-SCFI-2001
Nitrate N-NO ₃	Colorimetric method in Hach spectrophotometer with specific reagents. model 890,	NMX-AA-079-SCFI-2001
Nitrites, N-NO ₂	Colorimetric method in Hach spectrophotometer with specific reagents. model 890,	NMX-AA-026-SCFI-2001
Ammoniacal nitrogen, N-NH ₃ ;	Colorimetric method in Hach spectrophotometer with specific reagents. model 890,	NMX-AA-026-SCFI-2001
Chemical oxygen demand, COD	Spectrophotometric method	NMX-AA-030-SCFI-2001

Nitrogen compounds were measured 46 times over half a year for each EU, with a repeatability of five times per observation. The leaching analysis of the columns was fulfilled, starting from the natural humidity of the soil with a continuous flow of water. The irrigation in the EU was carried out as it is done in the green areas of the NTP. The irrigation sheet had a maximum of 10 cm, because most of the cover was grass. The type of irrigation was by sprinkling.

RESULTS AND DISCUSSION

Nitrogen compounds transformation

N-NH₃ was not present at the potable water input, for that reason the L1 sample had a N-NH₃ minimum concentration increase throughout the experiment, produced by organic nitrogen mineralization (Schulten & Schnitzer 1997) (Figure 4).

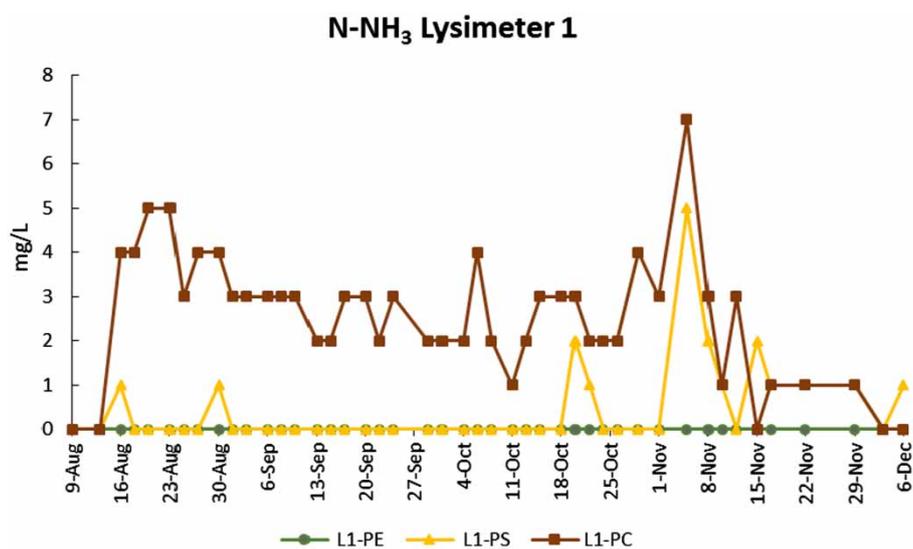


Figure 4 | N-NH₃ content in Lysimeter 1 irrigated with drinking water. PE stands for concentration level at lysimeter input, PS stands for concentration level at L1S (with sandy soil) output and PC stands for concentration level at L1C (with clay soil) output.

The L2 sample showed a total removal of N-NH₃. This parameter had continuous variations in the TW input and was completely removed by the nitrification process, probably coupled with N-NH₃ volatilization (Müller *et al.* 2002) (Figure 5). In both L1 and L2 systems, sands and clays soils had a 100% N-NH₃ removal efficiency.

The experimental system L1S presented an increase of N-NO₃ after passing through the soil column, due to the presence of total N and N-NO₃ in the sand. On average this increase from 0.6 mg/l at the entrance of the soil column to 5 mg/l at the exit. Leachate N-NO₃ content is associated with N-NO₃ solubilization in the soil and the ammonification reactions of organic nitrogen in the soil and subsequent N-NH₃ oxidation to N-NO₃ (Bernat *et al.* 2011) (Figure 6). L1 irrigated with drinking water presented an increment of N-NO₃ content by soil column washing, higher for clay soil type (Ye *et al.* 2012), however, this tends to stabilize and gets removed. The L2S experimental system presented N-NO₃ detriment after soil column washing, even in the presence of total nitrogen and N-NO₃ in the sand. On average from 19 mg/l at the input to 5.8 mg/l at the exit, which represented a consistent and similar removal with the behavior of COD. L2C presented an average increase of 19 mg/l at the entrance to 31 mg/l at the output, as a consequence of N-NO₃ compound transformations and percolation rate, allowing longer permanence in the soil (Figure 7). The N-NO₃ behavior showed periods of

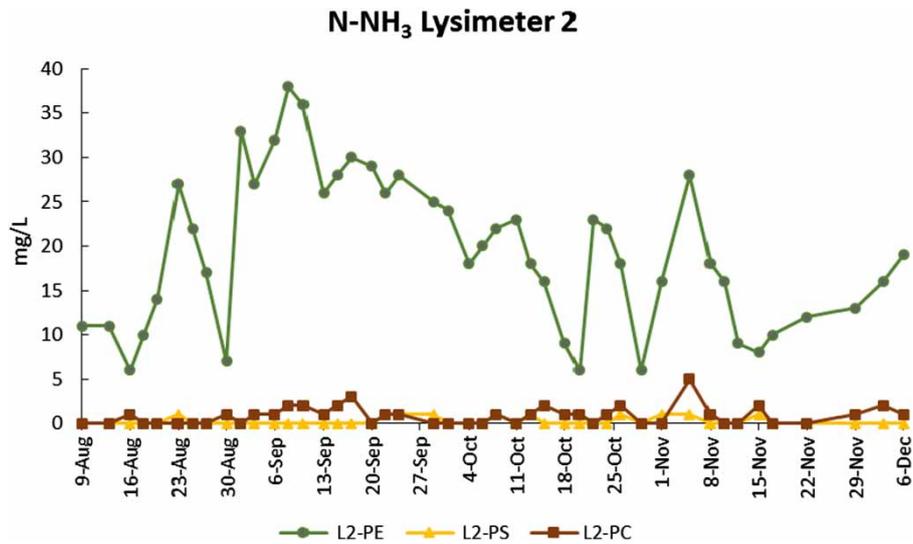


Figure 5 | N-NH₃ content in Lysimeter 2 irrigated with treated waste water. PE stands for concentration level at lysimeter input, PS stands for concentration level at L2S (with sandy soil) output and PC stands for concentration level at L2C (with clay soil) output.

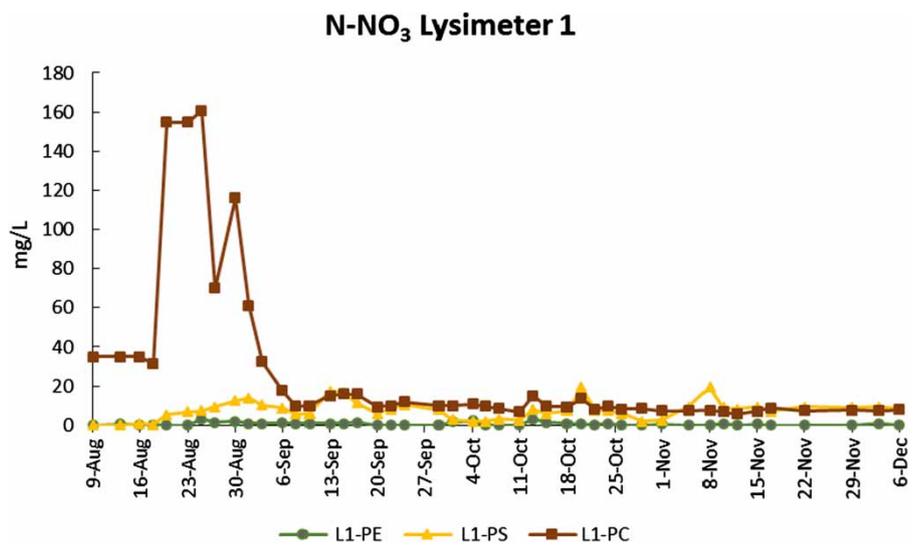


Figure 6 | N-NO₃ content in lysimeter 1 irrigated with drinking water. PE stands for concentration level at lysimeter input, PS stands for concentration level at L1S (with sandy soil) output and PC stands for concentration level at L1C (with clay soil) output.

increase and decrease over time, this is attributed to the soil column washing and the TW N-NH₃ transformation to N-NO₃. This phenomenon occurs mainly in clay soil because of the slow vertical velocity of water, when N-NO₃ is converted to N-NO₂ (Welch *et al.* 2011). The sand oscillates in their N-NO₃ removal efficiency in this period from 60 to 90% and in the case of clay up to 50%.

L1S and L1C showed an increase of N-NO₂ after soil washing from 0.3 mg/l to 2.5 mg/l, in the same N-NO₃ increase ratio; subsequently having a tendency to stabilize and present greater removal (Figure 8). Otherwise L2S and L2C presented a N-NO₂ removal from 3.8 mg/l to 0.8 mg/l and 4 mg/l to 2 mg/l on average, respectively (Figure 9). The sand oscillated in their N-NO₃ removal efficiency in this period from 90 to 100% and the clay samples up to 80%.

Sandy soil columns provided further efficiency and effectiveness in terms of inorganic nitrogen compounds concentration removal in the TW; N-NO₂ and N-NO₃ were eliminated almost entirely. From October onwards, the denitrification phenomenon could be identified in the system as a

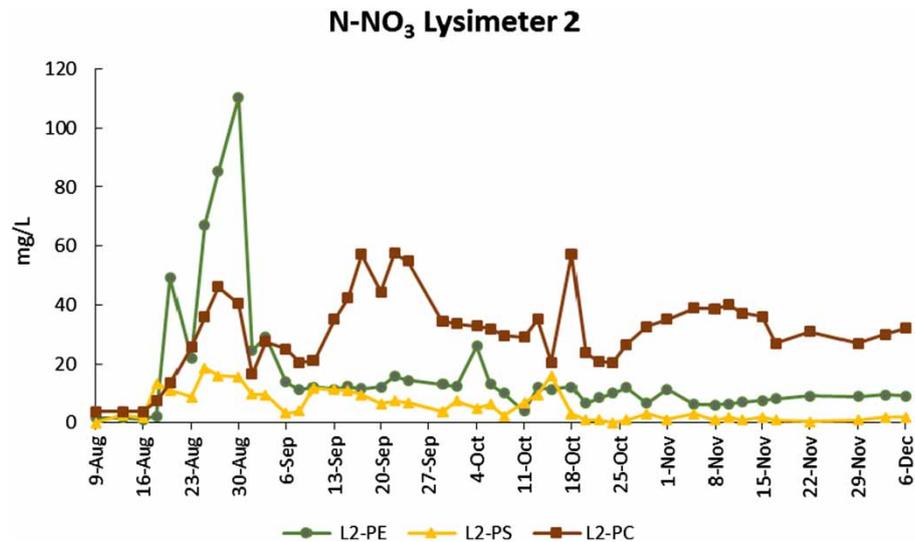


Figure 7 | N-NO₃ content in lysimeter 2 irrigated with treated waste water. PE stands for concentration level at lysimeter input, PS stands for concentration level at L2S (with sandy soil) output and PC stands for concentration level at L2C (with clay soil) output.

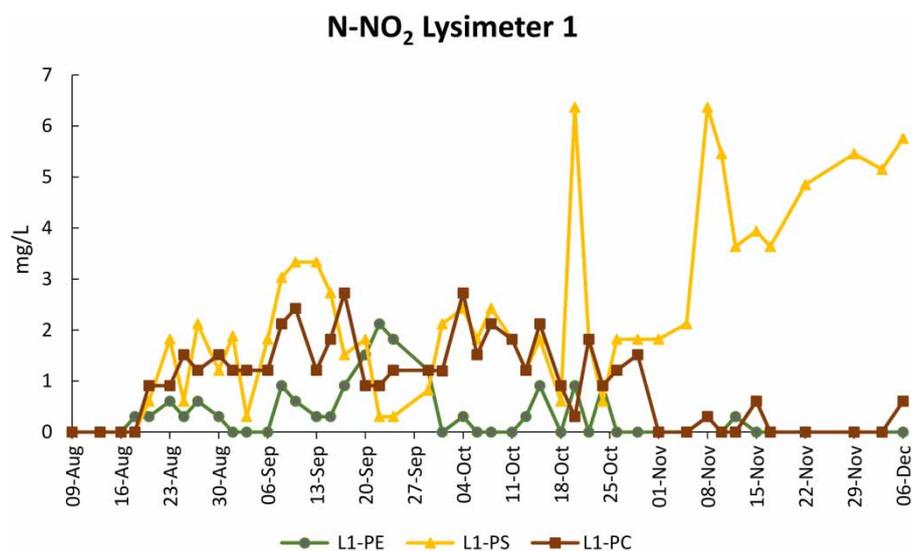


Figure 8 | N-NO₂ content in lysimeter 1 irrigated with drinking water. PE stands for concentration level at lysimeter input, PS stands for concentration level at L1S (with sandy soil) output and PC stands for concentration level at L1C (with clay soil) output.

consequence of the organic matter consumption, in the way of a carbon source for denitrifying heterotrophic organism. The nitrification-denitrification process could be verified even with the drastic decrease in temperature observed during the last month of the experiment. The nitrogenous component losses found could be associated to adsorption processes in the soil itself or to microbiological transformations (denitrification, dissimilative reduction) (Teng *et al.* 2018).

The results obtained allowed to distinguish different concentration magnitudes, according to water quality drained from both columns (Table 3). To determine the statistical difference in the results, the experimental values were contrasted with the model's results. To consider a significant contrast, the probability must be less than $P = 0.05$. A uni-factorial ANOVA between groups was performed (TWO WAY option), this procedure provides a regression analysis and an ANOVA for a dependent variable (N-NO₂ and N-NO₃ in this case) by one or more factors (soil type and type of water). The effects of the variables were compared on the means of several pairings of a single dependent variable also giving

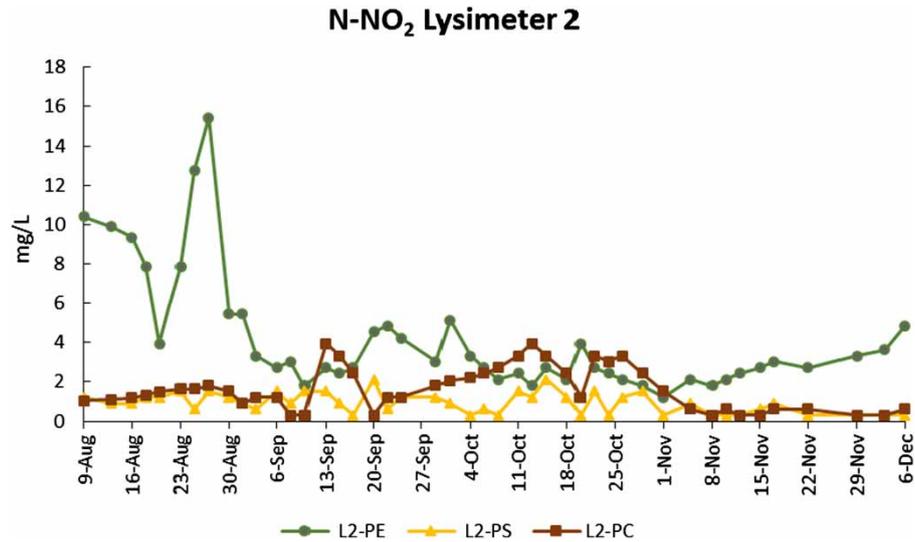


Figure 9 | N-NO₂ content in lysimeter 2 irrigated with treated waste water. PE stands for concentration level at lysimeter input, PS stands for concentration level at L2S (with sandy soil) output and PC stands for concentration level at L2C (with clay soil) output.

Table 3 | Summary values of the nitrogenous components

	Soil type					
	Sand (S)			Clay (C)		
Lysimeter 2 = water type (TW)						
Water Quality Entry (mg/L)	N-NO ₃	N-NO ₂	N-NH ₃	N-NO ₃	N-NO ₂	N-NH ₃
Max.	38	15	110	38	15	110
Mean	19	4	17	19	4	17
Min	6	1	1	6	1	1
Water Quality Exit (mg/L)	N-NO ₃	N-NO ₂	N-NH ₃	N-NO ₃	N-NO ₂	N-NH ₃
Max.	19	2	1	58	4	5
Mean	6	1	0	31	2	1
Min	0	0	0	4	0	0
Lysimeter 1 = water type (Potable Water)						
Water Quality Entry (mg/L)	N-NO ₃	N-NO ₂	N-NH ₃	N-NO ₃	N-NO ₂	N-NH ₃
Max.	3	2	0	3	2	0
Mean	1	0	0	1	0	0
Min	0	0	0	0	0	0
Water Quality Exit (mg/L)	N-NO ₃	N-NO ₂	N-NH ₃	N-NO ₃	N-NO ₂	N-NH ₃
Max.	19	6	5	161	3	7
Mean	5	1	0	27	1	3
Min	0	0	0	6	0	0

information on the interactions between the factors. In this case by the reduction of N-NO₃ to N-NO₂, the analysis was applied for both variables separately. In the study, a balanced model is presented. To carry out this analysis, the Minitab 15 software was used. After the ANOVA analysis was performed, the model regression value observed for the data (R-Sq = 4.35%) and the adjusted value (R-Sq (adj) = 2.75%) were low and similar, although with high reliability because of its low dispersion. It was observed that both models (observed and adjusted) have a normal distribution behavior, stating

certainty that the experimental and adjusted regression values are reliable. Table 4 shows an example of the standard deviation for the standard and adjusted method in L2S for the variable N-NO₃. Figure 10 shows an example of control in the repeatability and quantification of the measurements in N-NO₃. This graph displays the consistency between the variations of the maximums, averages and minimums values obtained in the 46 samples analyzed by quintupled. The decrease between the input and output values of the N-NO₃ is stated.

Table 4 | Test and CI for one standard deviation in L2S: N-NO₃ mean, min. and max

Sampling moment	Variable	N	St Dev ^a	Variance	Method	CI for St Dev ^b 95%	CI for Variance ^b 95%
Input	N-NO ₃ mean	46	21.0	441	standard	(17.4, 26.4)	(303, 699)
					adjusted	(13.9, 42.2)	(194, 1,779)
	N-NO ₃ min	46	20.9	436	standard	(17.4, 26.4)	(303, 699)
					adjusted	(13.9, 42.2)	(194, 1,779)
	N-NO ₃ max	46	21.1	446	standard	(17.4, 26.4)	(303, 699)
					adjusted	(13.9, 42.2)	(194, 1,779)
Output	N-NO ₃ mean	46	5.10	26.1	standard	(4.23, 6.43)	(17.9, 41.3)
					adjusted	(4.28, 6.32)	(18.4, 39.9)
	N-NO ₃ min	46	5.07	25.7	standard	(4.20, 6.38)	(17.7, 40.7)
					adjusted	(4.25, 6.28)	(18.0, 39.4)
	N-NO ₃ max	46	5.15	26.5	standard	(4.27, 6.48)	(18.2, 42.0)
					adjusted	(4.32, 6.36)	(18.7, 40.4)

^aSt Dev stands for standard deviation.

^bCI stands for confidence interval.

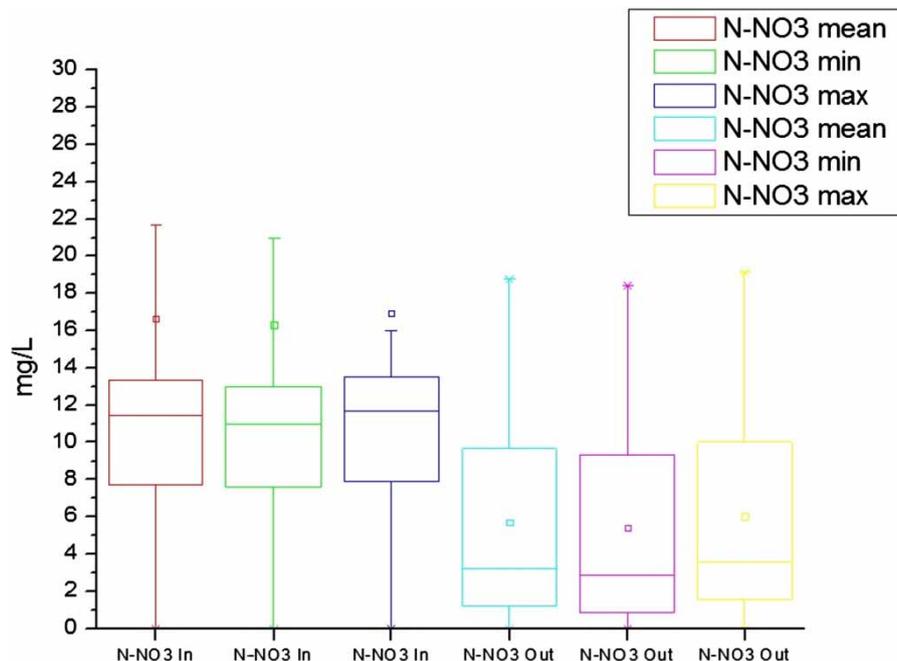


Figure 10 | Box plot for N-NO₃ content in lysimeter L2S.

Fisher tests are presented for means and variance comparison of the Gaussian samples, the variance analyses for the concentrations of N-NO₂ are shown the Table 5. The water type was the factor that most influenced the presence of N-NO₂. The difference between the means of the sampled groups with respect to the N-NO₂ is significant, in regards to the type of water. Alternately, the soil type did not present significant differences.

Table 5 | N-NO₂ ANOVA results for output water

Source ^a	DF ^b	Seq SS ^c	Adj MS ^d	F-value	P-value
TS	1	1.073	1.07284	1.78	0.184
TA	1	2.601	2.060134	4.31	0.009
TS*TA	1	1.261	1.26128	2.09	0.150

^aTA stands for variation of water and TS stands for soil type.

^bDF total degree of freedom, amount of information in data.

^cSeq SS, sequential sums of squares are measures of variation for model components.

^dAdj MS, adjusted mean squares.

For the N-NO₃ concentration, the results obtained are shown in the [Table 6](#), significant differences were obtained concerning type of soil and water quality.

Table 6 | N-NO₃ ANOVA results for output water

Source ^a	DF ^b	Seq SS ^c	Adj MS ^d	F-value	P-value
TS	1	4,194	4,194.36	8.34	0.004
TA	1	7,473	7,472.78	14.85	0.000
TS*TA	1	4,060	4,059.86	8.07	0.005

^aTA stands for variation of water and TS stands for soil type.

^bDF total degree of freedom, amount of information in data.

^cSeq SS, sequential sums of squares are measures of variation for model components.

^dAdj MS, adjusted mean squares.

Although similar tests have been carried out, it is necessary to particularize for the specific TW quality and native soil type; the TW use viability depends deeply on the removal of existing compounds in the TW quality ([Paradis et al. 2017](#)). The obtained results demonstrated that groundwater N-NO₃ contamination can vary depending on the native soil type. Using a statistical model can be a powerful tool to identify and classify major variables explaining groundwater pollution, thus aid the groundwater quality management. Groundwater N-NO₃ pollution is however not only a spatial, but also a temporal process; and N-NO₃ is not a conservative tracer, since its concentration can be affected by complex biogeochemical processes, redox chemistry in particular ([Mfumu Kihumba et al. 2016](#)).

The lysimeter 2 was analyzed against the existing regulations for inorganic nitrogen compounds with recharge possibility (NOM-014-CONAGUA-2003; NOM-127-SSA1-1994). The sand columns had the greatest efficiency in terms of removal of inorganic nitrogenous compounds existing in the TW; even with the intense decrease in temperature during December, the phenomenon of nitrification-denitrification could be verified. N-NH₃ adsorption could be seen in both the clay and sand columns ([Müller et al. 2002](#)). The greater presence of N-NO₃ in leached water denoted the possible nitrification reactions of the N-NH₃.

Highly permeable soils such as sand and gravel, infiltrate water faster than clay, transporting dissolved substances such as N-NO₃. Sandy soils are generally well aerated, providing favorable conditions for the conversion processes to produce N-NO₃. Clay soils have instead slower vertical downward flow due to smaller pores. Establishing anaerobic conditions nitrogen can be obtained as a gas that will escape to the atmosphere by means of the denitrification process ([Bojorquez et al. 2018](#)).

The lysimeter percolate analysis provides the most direct measurements of vertical flux through time at the depth of installation ([Heppner et al. 2007](#)). Lysimeters give a first estimate of the water quality leached variability in response to type of soil ([Walker et al. 2002](#)). However deep-drainage

representation generates some difficulties, as is the case of the ES (Scanlon *et al.* 2002; Walker *et al.* 2002). The unsaturated zone of the ES corresponds to an old channel of the Sacramento River, made of granular units deposited by the river, or in connection with abandoned meanders. The presence of intercalated clay materials slows the infiltration rate. When the average depth of the water table in CHSA is 100 m, piezometric monitoring and groundwater analysis are required before assessing N-NO₃ contamination in the aquifer (Scanlon *et al.* 2002).

CONCLUSIONS

The present study concluded that in the area of NTP irrigated with TW, the risk of nitrogen compounds leaching into the aquifer is minimal. The environmental conditions favor N-NO₃ removal from the subsoil. One meter of sand column is sufficient for the total removal of inorganic nitrogenous components present in TW of the NTP of Chihuahua, following the criteria of Mexican norms NOM-014-CONAGUA-2003; NOM-127-SSA1 (Official Mexican Standard, 2000). In clays, the removal of N-NO₂ and N-NO₃, produced by one meter of soil column is not enough, the smaller pores and lower permeability produce a slower vertical downward flow. It must be considered that in clay soils there may be slightly anaerobic conditions, so by means of the denitrification process, nitrogen will be obtained as a gas that will escape to the atmosphere (Bojorquez *et al.* 2018). In soils with clay presence, it is necessary to confirm the existence of sand layers that are at least one meter thick. The presented results are important in the context of using TW in the development of MAR structures. It is feasible to make a prototype of artificial recharge in the CHSA with TW quality, obtained with a secondary water treatment. The NTP gardens provide an ideal sub-superficial process for artificial recharge. Methods for mapping of MAR opportunities remain poorly founded in the absence of comparative information and in relation to practical experience (Dillon *et al.* 2018).

This research lays down a latent need to evaluate *in-situ* protocols that allow to improve recharge criteria based on soil type and water quality. The lack of values limits the adequate classification and adoption of efficient criteria, according to the MAR technique or device to be used.

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