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NITROUS OXIDE EMISSION FROM SOILS UNDER SUGAR CANE IN MAURITIUS

Final Report

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MAURITIUS SUGAR INDUSTRY RESEARCH INSTITUTE

Agricultural Chemistry Department

MRC CONTRACT 97/6 – Final Report

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NITROUS OXIDE EMISSION FROM SOILS UNDER SUGAR CANE IN MAURITIUS

SUMMARY

Prompted by environmental and agronomic concerns, emissions of nitrous oxide (N_2O) and nitrogen (N_2) gases from soils under sugar cane were monitored by the acetylene inhibition technique at 4 sites fertilized with and without 140 kg N ha^{-1} . The data obtained showed that the rates of $\text{N}_2\text{O}/\text{N}_2$ emission from the soils cannot be predicted by any single soil variable. Moreover in spite of site to site variabilities, the temporal pattern of the gaseous N emissions invariably showed a base level emission rate ranging from 10 to $50 \text{ g N ha}^{-1}\text{d}^{-1}$. Upon that background flux were pulses of emission which could be as high as 5 times the baseline flux. Following N fertilizer application, large pulses of gaseous N escape with the flux attaining a peak of $500 \text{ g N ha}^{-1}\text{d}^{-1}$ were only observed in the superhumid regions ($> 3\,500 \text{ mm rain y}^{-1}$) such as Belle Rive and Union Park. At the 2 sites (Réduit and Pamplémousses) in the low rainfall areas ($< 1\,500 \text{ mm rain y}^{-1}$), the rates of gaseous N loss from soils were hardly altered by fertilizer application.

The results therefore showed that gaseous losses of fertilizer N could be of agronomic significance only in the high rainfall zone. On average 15% of the 140 kg N ha^{-1} applied at Belle Rive escaped to the atmosphere over a sugar cane growing season.

Soil wetness further controlled the ratio $\text{N}_2\text{O}:\text{N}_2$ emitted from the soils. Conversion of N_2O to N_2 gas was of significance at Belle Rive and Union Park where more than 80% of the soil pore space was shown to be water-filled for frequent prolonged periods. Though the present study showed that gaseous losses of fertilizer N may in general not be of agronomic or economic relevance, the magnitude of yearly N_2O emissions from sugar cane fields which ranged from 5 to 17 kg N ha^{-1} , may be of concern from the environment point of view.

Key words : *Nitrogen fertilizer, denitrification, sugar cane, environment*

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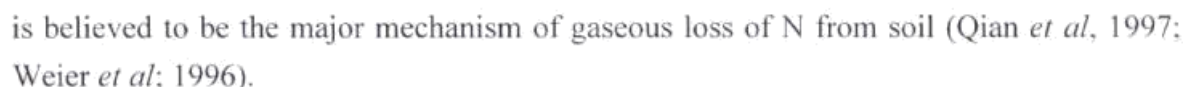
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1 BACKGROUND INFORMATION

Nitrogen (N) is a key plant nutrient. The success of the sugar cane industry in Mauritius over the past 50 years has, to a significant degree, been due to an improvement in the supply of N to the sugar cane. Indeed from approximately 300 000 tonnes sugar with an annual input of 2 500 tonnes N in the late 1940s, a production of 620 000 tonnes sugar annually when 11 000 tonnes N are used is now a regular feature in Mauritius. Uptake of the fertilizer N by the sugar cane (as in most crops) however is low and rarely exceeds 40% (Ng Kee Kwong and Deville, 1987). True, the fertilizer N may be leached from the soils but those leaching losses are rarely significant and amount to less than 5% of the fertilizer N applied (Ng Kee Kwong and Deville, 1984). The poor efficiency with which fertilizer N is used by crops, as reviewed by Mosier *et al* (1996), may instead be attributed to large losses of N occurring in gaseous forms. Denitrification, illustrated as follows :



1

responsible for this deficit. In Mauritius, N balance studies, using ^{15}N have shown that 40% or more of the fertilizer N applied to sugar cane remained unaccounted for and has been presumed to be lost by denitrification (Ng Kee Kwong and Deville, 1987). Those estimates of N denitrification losses must however be viewed with considerable caution because of the shortcomings in the assessment of total N recovery in the N balance procedure (Knowles, 1982; Nieder *et al*; 1989).

More reliable estimates of the magnitude of N losses by denitrification based on direct field measurements have been lacking and need to be obtained before a precise and valid assessment can be made of the real significance of gaseous N losses in the productivity of the sugar cane crop. At present with 11 000 tonnes N applied annually over 78 000 ha of sugar cane lands in Mauritius, 4 500 tonnes of this N are estimated to be lost by denitrification. Reliable estimates of denitrification have been lacking because measuring gaseous loss of N from soils by denitrification under field conditions presented formidable difficulties on account of problems inherent in quantifying either the production of the dominant end-product (N_2) or the specific depletion of the NO_3^- substrate (Groffman, 1995).

Production of N_2 is difficult to measure against an atmospheric background already containing 78% nitrogen gas (N_2). Depletion of NO_3^- by denitrification is also difficult to quantify due to the fact that this substrate ion is consumed by plants, heterotrophic micro-organisms and is also subjected to leaching and run-off. The discovery that acetylene (C_2H_2) blocks the reduction of nitrous oxide (N_2O) to N_2 was a major breakthrough in denitrification research as the total products of denitrification can then be measured as N_2O (Balderston *et al*, 1976; Yoshinari and Knowles, 1976). The measurement of N_2O flux from soil is relatively straightforward due to its low atmospheric background and the easy availability of sensitive gas chromatographic detectors for this gas. Measurement of N_2O flux has therefore provided the basis for elaboration of field techniques used to quantify denitrification losses of N.

1.2 Environmental concern – Denitrification as a source of nitrous oxide

Although, as reviewed by Bremner (1997), it has been known for more than 50 years that N_2O is a regular constituent of the atmosphere, it was not considered to be of any importance as an air constituent until the early 1970's when atmospheric scientists realised that N_2O released to the atmosphere as a result of denitrification of NO_3^- in soils

may trigger reactions in the stratosphere leading to a thinning of the ozone layer protecting the earth from the harmful ultraviolet radiation of the sun. This awareness created international concern that increased use of N fertilizers to aid world food production had also been promoting the destruction of the stratospheric ozone layer and stimulated extensive research to identify and quantify the sources of N₂O.

The research done showed that the concentration of N₂O in the global atmosphere has been rising over the last 20 years at the rate of about 0.25% y⁻¹ to attain 310 ppbv in today's atmosphere (Mosier *et al*, 1996; Yamulki *et al*, 1995). The 0.25% annual increase in N₂O represents a 3.5Tg addition of N₂O-N each year to the global atmosphere where the N₂O is chemically unreactive with an atmospheric lifetime of about 150 years (Weier, 1998; Yamulki *et al*, 1995).

The 0.25% N₂O loading rate is creating concern primarily for two reasons. Firstly N₂O is one of the major greenhouse gases accounting for 6 to 8% of the greenhouse forcing rate ascribed to the anthropically derived gases, namely CO₂, CFC's, CH₄ and N₂O (Bouwman, 1990). On a molar basis N₂O is about 250 times more potent than CO₂ as an absorber of infrared radiation - in part due to its molecular structure but largely due to the fact that it absorbs in a portion of the infrared transmission window that in our present atmosphere is relatively clean (Duxbury *et al*; 1993). Secondly, N₂O is the major regulator of stratospheric ozone which effectively controls the earth's ultraviolet B (UVB) radiation balance (Cicerone, 1987). Through a series of reactions elucidated over the last 20 years, the oxidation of N₂O in the upper stratosphere (> 25 km) produces nitric oxide (NO) that in turn reacts with ozone (O₃) to form nitrogen dioxide (NO₂) and oxygen gas (O₂). This results in a thinning of the ozone layer and consequently increases the potential health hazards caused by more penetration of UV radiation. It has been estimated that doubling the concentration of N₂O (which was 285 ppbv in the 1850's) in the atmosphere would result in a 10% decrease in the ozone layer and increase the ultraviolet radiation reaching the earth by 20% (Crutzen and Ehhalt, 1977) thereby, as cited by Mosier *et al* (1996), enhancing the incidence of skin cancer and other health problems.

Recent research, as reviewed by Kawashima *et al* (1996) indicated that N₂O generation by fossil combustion and biomass burning are minor sources of N₂O. About 90% of the N₂O emission are believed to come from the soils (Byrnes, 1990; Freney, 1997). The N₂O release due to fertilizer use and agriculture is estimated to be 3.5 Tg N y⁻¹ and represents, as reviewed by Mosier (1994), 65 to 80% of the total anthropogenic source of

N₂O. In fact, since most of the conversion of forest and grasslands in the northern hemisphere occurred 50 to 200 years ago the increases in atmospheric N₂O concentration could only be linked to the use of nitrogen fertilizers and to irrigation in row crop agriculture (Granli and Bockman, 1995; Smith *et al*; 1997). Indeed with the N input to agricultural systems through both biological and industrial fixation currently standing at about 150 Tg N y⁻¹, the loss of just 2 to 3% of this N as N₂O would account for the entire observed increase in atmospheric N₂O of 3.5 Tg N y⁻¹ (Denmead, 1991).

Most of the N₂O emitted from soils is derived from the microbially mediated reaction of nitrification and denitrification (Byrnes, 1990; Bremner, 1997). Most of the N₂O evolved under aerobic or semi-aerobic conditions is produced by nitrifying organisms while the major part of the N₂O evolved from soils under anaerobic conditions is derived from denitrifying organisms (Knowles, 1982; Smith *et al*; 1997). Since aerobic and anaerobic microsites can develop within the same soil aggregates, nitrification and denitrification could be occurring at the same time (Stevens *et al*, 1997). Also while the mechanisms of N₂O production by denitrification is well understood (Maag and Vinther, 1996), there are, as reviewed by Stevens *et al* (1997), at least 2 possible pathways for N₂O production during nitrification. Thus certain nitrifying organisms may generate N₂O from the reduction of NO₂⁻ which they produce under oxygen-limiting conditions while NH₄⁺ oxidizers may also reduce NO₂⁻ to N₂O to minimize intracellular accumulation of toxic amounts of NO₂⁻.

1.3 Existing knowledge on denitrification

A large body of information now exists on the principles governing denitrification in soils, factors that control production of N₂O and on the source/sink relationships (since microbial processes also consume N₂O, the net emission is a balance between production and consumption). In this context though, as reviewed by Scholes *et al* (1997), denitrification activity is impacted by soil variables such as available organic carbon, nitrate levels, pH, temperature, aeration and soil moisture, the main factors regulating denitrification are believed to be the available carbon and the wetness of the soil (Van Kessel *et al*, 1993). Organic matter is essential to the occurrence of denitrification and this effectively limits denitrification to the upper soil horizons (Scharf and Alley, 1988; McCarty and Bremner, 1992). In arable land temperature and moisture content rather than the type of crop were shown to be the determining factors of denitrification (Van Cleemput *et al*; 1994). Fertilizer induced N₂O emission have been observed to vary between 0% and 7% of the N applied and is not related to fertilizer type (Bouwman,

1996). As reviewed by Mosier *et al* (1996), soil management and rainfall have a greater effect on N₂O emission than type of fertilizer nitrogen and Byrnes (1990) even suggested that N₂O emissions from fertilizers may be more closely related to soil properties than the N source supplied. Existing knowledge further shows that N₂O emissions from agricultural soils are generally greater than those from uncultivated land or natural ecosystems. Fluxes from uncultivated land or natural ecosystems of the temperate regions are mostly below 1 kg N₂O-N ha⁻¹y⁻¹ while those from cultivated/fertilized lands tend to be greater than 3 kg N₂O-N ha⁻¹y⁻¹ (Bouwman, 1990; Granli and Bockman, 1995).

Current understanding of N₂O emissions in the terrestrial ecosystems is based largely on studies conducted in temperate regions (Davidson *et al*, 1996). There are little reliable data on actual N losses by denitrification in tropical soils. Davidson *et al* (1996) also drew attention to the fact that it is unclear how applicable the findings in temperate regions are for predicting the effects of increased use of N fertilizers on N₂O emissions in tropical ecosystems. A priori, as cited by Smith *et al* (1997), the predictions suggest that as long as NO₃ is available, denitrification rates ought to be high in the humid tropics where the soils tend to be highly aggregated and often moist resulting in low O₂ diffusion potentials (Robertson, 1989). Bouwman (1990) in fact estimated the N₂O fluxes from natural ecosystems of tropical regions to be in the vicinity of 2 kg N₂O-N ha⁻¹y⁻¹ as compared to less than 1 kg N₂O-N ha⁻¹y⁻¹ in temperate regions. In one of the earliest studies, Balasubramanian and Kanehiro (1976) found denitrification to be the major process of N loss in 4 of 6 Hawaiian soils with 30 to 50% of gaseous N loss occurring as N₂O. In Florida, Duxbury *et al* (1982) observed in the unfertilized organic soils cultivated with sugar cane a maximum daily N₂O flux of 3.1 kg N ha⁻¹ between May 1979 and May 1981. Weier (1998) estimated that if emissions from bare soils, trash-blanketed soils and burning are added together, a total of 10.7 kt N₂O-N ha⁻¹y⁻¹ is emitted to the atmosphere from sugar cane fields in Australia.

1.4 *Current and future needs*

With economic constraint and concern for environmental quality challenging producers and researchers to develop agricultural production systems that use resources more efficiently and that reduce nutrient losses to the environment, a clear need exists for measurement of gaseous N losses in tropical agricultural systems. Even if the suspicion that gaseous N emission is not an important loss mechanism in tropical soils may in the end prove to be true, the suspicions are not a substitute for direct measurement. Indeed a better understanding of N cycling processes, particularly the microbial conversion of plant

available N to gaseous N forms, is needed for an evaluation of current agricultural management effects on air quality, and for aiding in the selection of the best N management practices for sustainable agriculture. This is particularly pertinent in view of the inescapable fact that N fertilizer use is set to further increase during the next century (Mosier *et al*, 1996; Smith *et al*; 1997).

According to Matson *et al* (1996), the worldwide consumption of synthetic N fertilizers rose from only 32 Tg in 1970 to about 80 Tg in 1990. Presently, the fertilizer N usage is increasing at the rate of about $1.3\%y^{-1}$ in developed countries and 4.1% in developing countries (cited by Mosier, 1993). Vitousek and Matson (1993) cited USEPA projections pointing to a doubling of N fertilizer use by 2025 in developing countries where half of the world fertilizer consumption is already concentrated and where, as reviewed by Smith *et al* (1997), the success of food grain production programmes hinge on a continued expansion of fertilizer usage to meet the needs of the rapidly growing population. If the projected extra fertilizer usage over the next few decades were to be distributed over additional arable land at broadly the same rate of application as practised today, the most likely outcome would be a *pro rata* increase in N₂O emission. However the situation looks to be less hopeful because the potential for additional land is limited with the consequence that the more likely scenario is the intensification of existing land use with higher fertilizer application, more double cropping and more irrigation (Smith *et al*, 1997). On the basis of existing knowledge this more likely scenario could well accentuate increases in N₂O emission unless the best available agronomic practices are worked out to maximize efficiency of N fertilizer use by crops and to minimize the proportion of the applied N emitted to the atmosphere as N₂O.

2 OBJECTIVES OF STUDY

The gaseous losses of N from soils by denitrification therefore have implications for both the agricultural community concerned with productivity and for the environmentalists concerned with the impact of the N lost on climate change and the ozone layer. The interests of each are in harmony rather than in conflict with one another as it is axiomatic that any strategy improving the efficiency of fertilizer N use by crops will ensure a greater return/productivity to the agricultural community and will minimize the opportunity for N₂O emission. Elaboration of new strategies, if there is a need to amend current practices, however requires above all a knowledge of the significance of N₂O losses from the sugar

cane soils. To acquire that knowledge, a study was initiated with the following objectives :

- (i) to quantify annual N losses from soils by denitrification under the present cultural and fertilizer management practices for sugar cane in different climatic zones of Mauritius, and
- (ii) to determine how denitrification rates fluctuate in relation to such biotic factors as soil moisture in sugar cane fields.

3 MATERIALS AND METHODS

The study was conducted at four sites, namely Belle Rive, Pamplémousses, Réduit and Union Park. The characteristics of the 4 sites are given in Table 1. At each site fertilizer N was surface banded in a single dressing along the sugar cane rows in the form of calcium ammonium nitrate (26% N) and at the rate of 0 (control) and 140 kg N ha⁻¹. These two fertilizer treatments were replicated four times in a completely randomised block design with each treatment plot size consisting of 4 sugar cane rows of 6 m at Belle Rive, 7 rows of 9 m at Réduit, 4 rows of 18 m at Pamplémousses and 6 rows of 6 m at Union Park. In the present study gaseous N flux measurements in the current practice of surface banding 140 kg N ha⁻¹ along sugar cane rows was therefore being compared against the emission in fields receiving no fertilizer N.

3.1 *Monitoring of N₂O fluxes*

The gaseous N flux measurements were carried out using the vented closed chamber technique outlined by Hutchinson and Mosier (1981) and Coolman and Robarge (1995). The chambers, constructed locally from cylindrical PVC pipes, had an internal diameter of 240 mm and a height of 300 mm. At every study site, they were inserted to a depth of 50 mm into the soil at least one week before gas sampling commenced and were left undisturbed during the study period. Prior to gas sampling, the open end of each chamber were covered with a screw-cap lid fitted with a rubber septum, through which a two-way syringe needle was inserted to enable withdrawal of air from the chamber headspace into a 10 mL evacuated glass vacutainer tube (100 mm x 16 mm) sealed with butyl rubber

stopper. Before evacuation, some granules of soda lime and calcium oxide were placed inside the vacutainer tube to remove CO₂ and water vapour and the tube was evacuated to 3×10^{-5} mbar using a Leybold Heraeus vacuum pump. In every treatment plot at each site four chambers were installed at randomly selected positions along the sugar cane rows. The N₂O flux measurements following acetylene injection in the soil began shortly after fertilizer N application in September/October and repeated at regular intervals of 3-4 days for the next 3 months and thereafter every fortnight until harvest of the sugar cane in August.

A small-scale diurnal study of the flux was also carried out in the N fertilized plots at Belle Rive and at Réduit prior to the initiation of the present study. To this end, 10 series of 3 chambers were installed along the sugar cane rows and the gaseous N flux was measured at hourly interval as from 8.00 a.m.

3.2 *Acetylene supply to the soil*

In each treatment plot, acetylene (C₂H₂) gas was supplied to the soil atmosphere surrounding two chambers only through probes constructed from rubber tubings (3 mm id), one end of which was sealed and inserted in the soil. The rubber tubings were punctured with 1mm holes at 20 mm intervals along 40 cm of its length measured from the sealed end. To treat the soil atmosphere surrounding a chamber with C₂H₂, six supply probes were inserted to a depth of 50 cm around the chamber. Each probe was attached to a supply manifold and to an acetylene cylinder. Since commercially available C₂H₂ contained substantial amount of acetone, which may act as carbon source to micro-organisms, the acetone was removed as described by Gross and Bremner (1992) by passing the gas through concentrated sulphuric acid and water. The acetone free C₂H₂ was then applied to the soil at a flow rate of 18L h⁻¹ for approximately 2 hours in order to establish the 1% C₂H₂ concentration in the soil air necessary to completely inhibit reduction of N₂O to N₂ (Ryden and Dawson, 1982)

3.3 *N₂O analyses*

All air samples were analyzed using a Varian 3300 gas chromatograph equipped with one 10-port and one 6-port flow injection valves, a 1.5 m pre-column, a 3.5 m analytical column, a ⁶³Ni Electron Capture Detector (ECD) and a Varian 4400 integrator. Instrument parameters associated with the operation of the gas chromatograph were as follows:

Parameters	Settings
Detector	^{63}Ni ECD
Sample loop	4.0 ml
N_2 Carrier gas	99.995%
Sample size	10 ml
Precolumn	1.5m Porapak Q (80-100 mesh)
Analytical column	3.5m Porapak Q (80-100 mesh)
Injector temperature	50°C
Oven temperature	70°C
ECD temperature	300°C
Range	10
GC attenuation	64
Carrier gas pressure	30 psi
Flow rate	30 ml min ⁻¹
GC valve rotation	Compressed air (50 psi)

Gas samples were injected into a 4.0 ml sample loop and loaded onto the pre-column which was controlled by the first 10-port flow injection valve. The second 6-port valve located between the pre-column and the analytical column controlled the gas components passing through the analytical column to the ECD. Both flow injection valves were actuated through a precise timing program so that CO_2 , O_2 and moisture were vented by backflushing the pre-column while N_2O continued its course through the analytical column to be quantified by the ECD. ECD detector response for N_2O was determined for N_2O concentrations ranging from 0 to 30 ng N_2O .

4 RESULTS AND DISCUSSION

4.1 *Diurnal and spatial variations of gaseous N fluxes*

Emissions of gaseous N compounds to the atmosphere have been reported to fluctuate with time of the day (e.g. Yamulki *et al*, 1995). This diurnal variation in gaseous N losses to the atmosphere has been linked to temperature fluctuations in the soil, N_2O emission rising with increasing day-time temperature and decreasing as the temperature drops in the evening (Denmead *et al*, 1979; Smith *et al*, 1997). Though, as shown in Fig 1, a clear diurnal variation in the rate of gaseous N emission could be observed in the soils of Mauritius, the variation tended to be modest, rising from 1.2 g $\text{N}_2\text{O-N ha}^{-1} \text{ h}^{-1}$ in the morning (8.00 a.m) to a maximum rate of 2.1 g $\text{N}_2\text{O-N ha}^{-1} \text{ h}^{-1}$ in the afternoon (2.00 p.m) at Réduit (1.7 g $\text{N}_2\text{O-N ha}^{-1} \text{ h}^{-1}$ in the morning to 4.2 g $\text{N}_2\text{O-N ha}^{-1} \text{ h}^{-1}$ in the afternoon at Belle Rive), because the range of temperature fluctuation was narrow.

More importantly and inherent to the gaseous N emission process when measured *in situ* N fluxes from soils are known to exhibit a high degree of spatial variability due to the presence of "hot spots" in the soils, that is, microsites of high population of microorganisms responsible for N₂O production and consumption (Corre et al, 1996; Parkin, 1987). As a result of the high spatial variability, coefficients of variation of more than 100% are normal in the measurement of gaseous N fluxes with values typically between 100 and 300% (e.g Williams *et al*, 1992). In the present study, irrespective of fertilizer treatments, the arithmetic mean values of 12 replicate measurements such as those presented in Fig 1 had coefficients of variation as high as 78% (range of 3.5 to 72% at Réduit and 5.3 to 78% at Belle Rive).

4.2 *Gaseous fluxes from the soils*

4.2.1 *Influence of soil characteristics*

Though measurements of gaseous N fluxes indicated that site to site variability clearly existed (Fig 2, a to d), some similarities in the fluxes are nevertheless clearly discernible. At every site even when no fertilizer N was used and in agreement with observations of other workers e.g Arah *et al* (1991) and Bouwman (1990), there was always a low background emission of gaseous N ranging from 10 to 50 g N ha⁻¹d⁻¹. Superimposed on this background rate of gaseous N flux were pulses of emission which could be as much as 5 times higher than the background emission rate. These pulses of gaseous N flux were more evident during the first year of study (October 1996-September 1997) than during the second year.

In addition, though soil texture has been reported to exert a major influence on denitrification losses from soils, including those under sugar cane (Velthof and Oenema, 1995; Weier *et al*; 1996) with gaseous losses increasing as the clay content is raised, the effect of soil texture and soil properties in general, as shown in Table 1, on gaseous N escape from soils under sugar cane in Mauritius was not clear cut as had also been observed elsewhere (e.g Ball *et al*, 1997; Nieder *et al*, 1989). Researchers (e.g. Myrold and Tiedje, 1985; Wagner-Riddle *et al*, (1996) have also reported C availability to be the predominant regulator of denitrification activity under field conditions. The data obtained from the present study cannot support this inference as well. Indeed though the soil at

Union Park has a higher organic or biomass carbon content than the soil of Belle Rive (Table 1), the cumulative gaseous N flux at Union Park was not higher than that at Belle Rive (Table 2). Therefore although there are sample data in the literature to show that usually just one soil variable dominantly influences gaseous N emission rate e.g Aulakh *et al*, 1992; Thornton and Valente, 1996), the temporal patterns of gaseous N flux observed in the present study tend to concur with the conclusion that the relation between soil type and N flux to the atmosphere is difficult to predict because of the numerous chemical and physical soil characteristics that can directly or indirectly control the N fluxes. In fact the soil physical properties at the 4 sites studied (e.g bulk density or porosity) bear no direct relation to any single characteristic, be it texture or organic matter content. A high (75%) or low (36%) clay content can both be associated with a low bulk density of 1.0 g cm^{-3} (Table 1).

4.2.2 *Impact of N fertilization*

The present study illustrates why in spite of the numerous studies done to determine the impact of nitrate supply on denitrification (e.g Skiba *et al*, 1996), the dependence of denitrification on nitrate concentration in soils still remain a contentious issue, with some workers (e.g Qian *et al*; 1997) finding the gaseous N flux to be independent of N supply while others (e.g Weier *et al*, 1993) obtained good correlations between NO_3^- concentrations and gaseous N losses to the atmosphere. Vermoesen *et al* (1996) showed that 97% of the variation in N_2O emission to the atmosphere could be entirely explained by the factor soil with fertilizer N accounting for the remaining 3% variation only. Tiedje *et al* (1984), on the other hand observed the gaseous N emissions from N fertilized land to be greater than from non fertilized areas, not only because of the direct loss of the mineral N applied by nitrification and denitrification but also because when the greater volume of crop residues produced as a result of the N fertilization were returned to the soil, those residues increased the soil respiratory activity and the anaerobic zones within which denitrification could occur.

A markedly and consistently larger escape of N to the atmosphere as a result of N fertilization was only observed at Belle Rive (Fig 2a) where pulses of N_2O flux attaining $500 \text{ g N}_2\text{O} - \text{N ha}^{-1} \text{ d}^{-1}$ in December 1997 occurred after the application of 140 kg N ha^{-1} and lasted up to approximately six weeks before returning to the background level of 30 to $50 \text{ g N}_2\text{O} - \text{N ha}^{-1} \text{ d}^{-1}$. The generally greater gaseous N flux thereafter observed in the N fertilized treatment at Belle Rive, as compared in treatment where no N was applied, may be attributed to the fact that, as explained by Ellis *et al* (1996), soil microorganisms under

anaerobic conditions and with sufficient labile C are able to actively or passively store quantities of NO_3^- in excess of their immediate energetic requirements and later release the N into the atmosphere. At the 2 sites in the lower rainfall regions, namely at Pamplémousses and Réduit (Fig 2, b & c) the temporal patterns of gaseous N flux to the atmosphere was hardly affected by fertilization with $140 \text{ kg N ha}^{-1}\text{y}^{-1}$.

Temporal variation in gaseous N losses from soils are regulated by factors such as soil water content, nitrate supply and the availability of biodegradable carbon (Mosier, 1993). Whether denitrification responds to N fertilization would depend upon whether the supply of nitrate was the most limiting factor to denitrification (Qian *et al*, 1997). The data obtained showed that nitrate was consistently the most limiting factor only at Belle Rive (Fig 2a) whereas at Pamplémousses and Réduit (Fig 2, b & c) where denitrification did not respond to N fertilization, other constraints such as soil wetness were more limiting than N supply.

4.2.3 *Influence of soil wetness*

As denitrification is an anaerobic process, oxygen (O_2) availability, as reviewed by Henrich and Haselwandter (1997), is generally considered to be the dominant factor limiting the rate of denitrification in soils. Indeed oxygen diffuses 10^4 times slower through water than through an equivalent thickness of air (Drury *et al*, 1992; Petersen and Andersen, 1996). High soil moisture content therefore reduces the diffusivity of O_2 in the soil and thereby increases the extent of anaerobic microsites and thus the rate of denitrification (Yamulki *et al*, 1995). The wetness of the soil also controls the ease of escape of N_2O and N_2 by diffusion through the soil pores (Granli and Bockman, 1995). Gaseous N emission, especially N_2O to the atmosphere, would consequently be specially favoured when the soil was just sufficiently wet to restrict oxygen availability to the microorganisms (Granli and Bockman, 1995). As the soil becomes very wet, nitrification would cease while denitrification would proceed increasingly to N_2 and gas escape from the soil would become hindered (Qian *et al*, 1997). That soil water content where both denitrification and nitrification proceed to give the maximum emission of N_2O and N_2 generally contain 50 to 60% water filled pore space (WFPS) and is normally close to the soil's field capacity (Teira-Esmatges *et al*, 1998).

In the context of the above discussion, the temporal patterns of gaseous N flux observed in this study (Fig 2, a to d) had been largely determined by the impact of rainfall pattern on soil. Thus, the large gaseous N pulses often observed during the period December 1996 to

April 1997 could be attributed to the occurrence of high and frequent rainfall events whereas the base line emission activity would reflect a low soil wetness. A more detailed analysis of the temporal flux of N escape to the atmosphere further revealed that the gaseous emission after two similar rainfall events most often differed in magnitude. Consistent with the observations reported in the literature, (e.g Scholes *et al*, 1997) a larger pulse of gaseous N flux was observed when the rainfall event was preceded by a prolonged dry period. Alternating dry and wet conditions rather than being constantly wet is believed to induce the mineralization of soil organic matter thereby increasing the availability of both nitrate and degradable organic carbon (Granli and Bockman, 1995).

Moreover in affecting the balance between diffusive escape of N_2O and its further reduction to N_2 , soil wetness determined the proportion of N_2O and N_2 in the flux. Indeed since a rising soil moisture content would increase the residence time of N_2O and hence would facilitate its subsequent conversion to N_2 (Yamulki *et al*, 1995), it was therefore not surprising that in the high rainfall regions of Belle Rive and Union Park the gas emitted contained invariably more N_2 than N_2O (Fig 3, a & d). According to Davidson (1992) and Smith *et al* (1997), the production of N_2 becomes dominant when water filled pore space in the soils exceeds 80% over prolonged periods.

The effect of rainfall on WFPS in the soils studied showed that only at Belle Rive and at Union Park would the level of WFPS stay close to 80% for prolonged periods during the year (Fig 4). Furthermore N_2O production in soils at moisture content below field capacity (50-60% WFPS) has generally been attributed to nitrification (Davidson, 1992; Qian *et al*, 1997) with denitrification being only of minor significance (Linn and Doran, 1984). As cited by Jambert *et al* (1997), it is only beyond a threshold of 55% WFPS that nitrifier activity would start to decrease. Thus the impact of rainfall events on WFPS in the low rainfall regions of Pamplémousses and Réduit (Fig 4), apart from explaining why the gas fluxes were richer in N_2O than N_2 (Fig 3, b & c), indicates that in those dry areas nitrification often was the dominant process in the flux of N_2O to the atmosphere. This observation concurred with the inference of Skiba *et al* (1993) who concluded that under circumstances where moderate to low gaseous N losses were measured, e.g under dry soil conditions, the nitrification process acquired a higher relative importance than denitrification in the emission of N_2O to the atmosphere.

4.3 *Estimates of yearly N losses to the atmosphere*

As the gaseous N flux is controlled by several independent factors (Aulakh *et al*, 1992; Scholes *et al*, 1997) which undergo rapid temporal fluctuations and interact with each other, denitrification in soils does not occur as a continuous process of constant intensity (Qian *et al*, 1997; Thornton and Valente, 1996). Denitrification usually occurs as individual events taking place at irregular intervals depending on the constellation of the relevant factors. To overcome the uncertainties related to this temporal variability, continuous measurements of gaseous N flux to the atmosphere, as pointed out by Velthof *et al* (1996), would be needed throughout the whole year. In practice however most measurements of gaseous N flux such as those performed in the present study are not continuous and are performed at intervals during the growing season. Hence it may be quite likely that some N₂O/N₂ flux peaks had not been recorded in the present study. On the other hand it was also possible that fluxes measured could have overestimated the N₂O and N₂ emissions occurring in the time interval between 2 measurements if the flux peak did not last for the whole interval. In spite of the uncertainties mentioned it was still assumed in calculating the yearly losses of N to the atmosphere from the sugar cane fields, that each measured flux represented the average emission during the interval of two measurements and the yearly total N₂O/N₂ fluxes were obtained by integration of those mean fluxes at all sampling dates.

As shown in Table 2, total fertilizer N losses by gaseous emission to the atmosphere ranged on average from 0 to 21 kg N ha⁻¹. While the fertilizer N losses were insignificant at Réduit and Pamplémousses, the gaseous fertilizer N loss represented about 15% of the fertilizer used at Belle Rive. The fertilizer N recovery figures observed at Réduit and Pamplémousses compared favourably with results of Mosier *et al* (1986) who reported that less than 2.5% of N fertilizer applied would be emitted as (N₂ + N₂O). Using data from studies with a coverage of at least one year on cropped fields, Bouwman (1996) obtained the following linear relationship between fertilizer N applied and N₂O emission:

$$E = 1 + 0.0125 \times F,$$

where E is the emission rate in kg N ha⁻¹, and

F is the fertilizer N application rate in kg ha⁻¹y⁻¹.

The equation of Bouwman (1996) states that on average 1.25 kg N is emitted to the atmosphere as N₂O for every 100 kg N applied to agricultural land, in addition to a background flux of 1 kg N caused principally by the cycling of non-fertilizer N in cultivated lands, i.e N from crop residues, soil organic matter and previous years fertilization. While the amount of fertilizer derived N₂O emission observed in the low

rainfall regions in the present study (Réduit and Pamplemousses) tended to be in line with the average loss of 1.75 kg N ha⁻¹ predicted by Bouwman (1996), the total N loss as N₂O from unfertilized soils under sugar cane in Mauritius (Table 2) was considerably higher than background emission of 1 kg N ha⁻¹ predicted by Bouwman (1996) for soils not receiving any fertilizer N. The substantially higher values of gaseous N losses in the present study therefore showed that the simplistic approach of Bouwman (1996) for calculating N₂O emission is not adequate for the estimation of gaseous N escape in soils of Mauritius.

Comparison of the gaseous fluxes of N observed in the present study by the acetylene-inhibition technique with those estimated indirectly by the deficit in the fertilizer N balance (Ng Kee Kwong and Deville, 1987) showed that the acetylene inhibition method only accounts for a minor fraction of the deficit in the fertilizer N balance. Although this discrepancy may to some extent be due to errors in the determination of total N (Knowles, 1982), it could also very likely be caused by the acetylene-inhibition method underestimating gaseous N losses since N diffusion into deeper soil horizons had not been taken into account (Germon, 1985; Heinemeyer *et al*, 1988). Furthermore though it is usually assumed that the rates of N₂O + N₂ emission measured by the acetylene inhibition technique represent the actual denitrification in soil, recent systematic studies indicate (i) in well-aerated soils, little N₂O may evolve but much larger amounts of nitric oxide (NO) may be emitted during nitrification (Scholes *et al*, 1997), and (ii) plants including those grown on normally aerated soils can release significant amounts of N₂O into the atmosphere (Chang *et al*, 1998). The N₂O released is apparently not produced within the plant tissues but within the soil. The N₂O dissolved in soil solution is absorbed by roots along with water and then conveyed to the leaves in the transpiration stream. With the evaporation of the water at the leaf surface the N₂O is released into the atmosphere, the plant thus serving as a mechanical conduit for N₂O emission.

5 CONCLUSION

The two primary mechanisms for fertilizer N loss from the soil ecosystems are nitrate leaching and gaseous diffusion to the atmosphere (Scholes *et al*, 1997). While the little significance of nitrate leaching under the climatic conditions prevailing in Mauritius have already been established (Ng Kee Kwong and Deville, 1994), the present study extends existing knowledge by providing evidence to show that gaseous N emission as N₂O and N₂ are in general not of any economic or agronomic significance except in the superhumid

regions of Mauritius. Indeed, in general less than 0.5% of the applied N is expected to escape into the atmosphere as $\text{N}_2\text{O}/\text{N}_2$ from the soils under sugar cane in the low rainfall areas (< 1500 mm/year) of Mauritius. However though the fertilizer derived emission of N_2O from sugar cane plantations may be of little economic importance from the growers point of view, the N_2O emissions from sugar cane soils, which could be as high as $17 \text{ kg N ha}^{-1}\text{y}^{-1}$ may however be of sufficient magnitude to be of concern to society because of their likely effect on the accumulation of heat trapping gases in the atmosphere and in the quality of sunlight radiation reaching the earth.

The present study has also shown that the most important proximal control of N_2O emission to the atmosphere is soil wetness. Consequently of the many approaches suggested for controlling losses of fertilizer N, including choice of appropriate fertilizer form, methods of application, timing N supply or even use of slow-release fertilizers, none of them can be expected to have a large impact on N_2O losses. As noted by Mosier (1993), soil management can be expected to impact N_2O emissions more than mineral N management practices.

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Table 1. Characteristics of soils at study sites

Site and soil depth (cm)	Soil (FAO/UNESCO classification)	Annual rainfall (mm)	pH (H ₂ O)	Organic C (gkg ⁻¹)	Total N (gkg ⁻¹)	C/N	Biomass		Mechanical analysis (%)		Bulk density (gcm ⁻³)	Porosity
							C	N	Sand	Silt	Clay	
Belle Rive												
0-10	Humic Acrisol	3800	4.9	34.0	2.5	13.6	576	53	34.1	29.9	36.0	1.16
10-25			4.9	23.4	1.7	13.8	494	47	31.6	30.6	37.8	1.27
Union Park												
0-10	Dystic Cambisol	3850	4.7	57.8	4.4	13.1	827	88	33.2	27.0	39.8	0.99
10-25			4.6	49.0	4.0	12.3	370	66	29.1	26.2	44.7	1.11
Rédut												
0-10	Humic Nitosol	1500	4.7	21.7	2.3	9.4	1189	46	10.6	20.7	68.7	1.01
10-25			5.7	20.7	2.2	9.4	856	24	12.2	22.7	65.2	0.96
Pamplemousses												
0-10	Humic Nitosol	1500	4.9	24.5	2.2	11.1	310	6.4	9.0	16.3	74.7	1.17
10-25			4.8	20.5	2.0	10.3	97	3.5	9.4	15.4	75.3	1.34

Table 2. Yearly N_2O and N_2 flux ($kgNha^{-1}$) from soils under sugar cane fertilized with $140\text{ kgNha}^{-1}yr^{-1}$ in Mauritius

Sites	1996-1997 growth season						1997-1998 growth season						Mean 2 years	
	Control (no N)			140 $kgNha^{-1}yr^{-1}$			Control (no N)			140 $kgNha^{-1}yr^{-1}$				
	N_2O	N_2	(N_2O+N_2)	N_2O	N_2	Total (N_2O+N_2)	N_2O	N_2	N_2O+N_2	N_2O	N_2	Total (N_2O+N_2)	Fertilizer (N_2O+N_2)	Total (N_2O+N_2)
Belle Rive	13.7	3.2	16.9	17.3	21.3	38.6	3.9	3.8	7.7	5.4	21.3	26.7	19.0	32.7
Union Park	14.7	1.5	16.2	15.2	3.8	19.0	4.6	1.3	5.9	5.4	10.0	15.4	9.5	17.2
Réduit	16.9	2.6	19.5	16.4	1.8	18.2	8.9	1.0	9.9	8.3	0.4	8.7	0.0	13.5
Pamplemousses	14.6	1.5	16.1	15.1	1.1	16.2	5.9	0.2	6.1	6.1	0.8	6.9	0.8	11.6

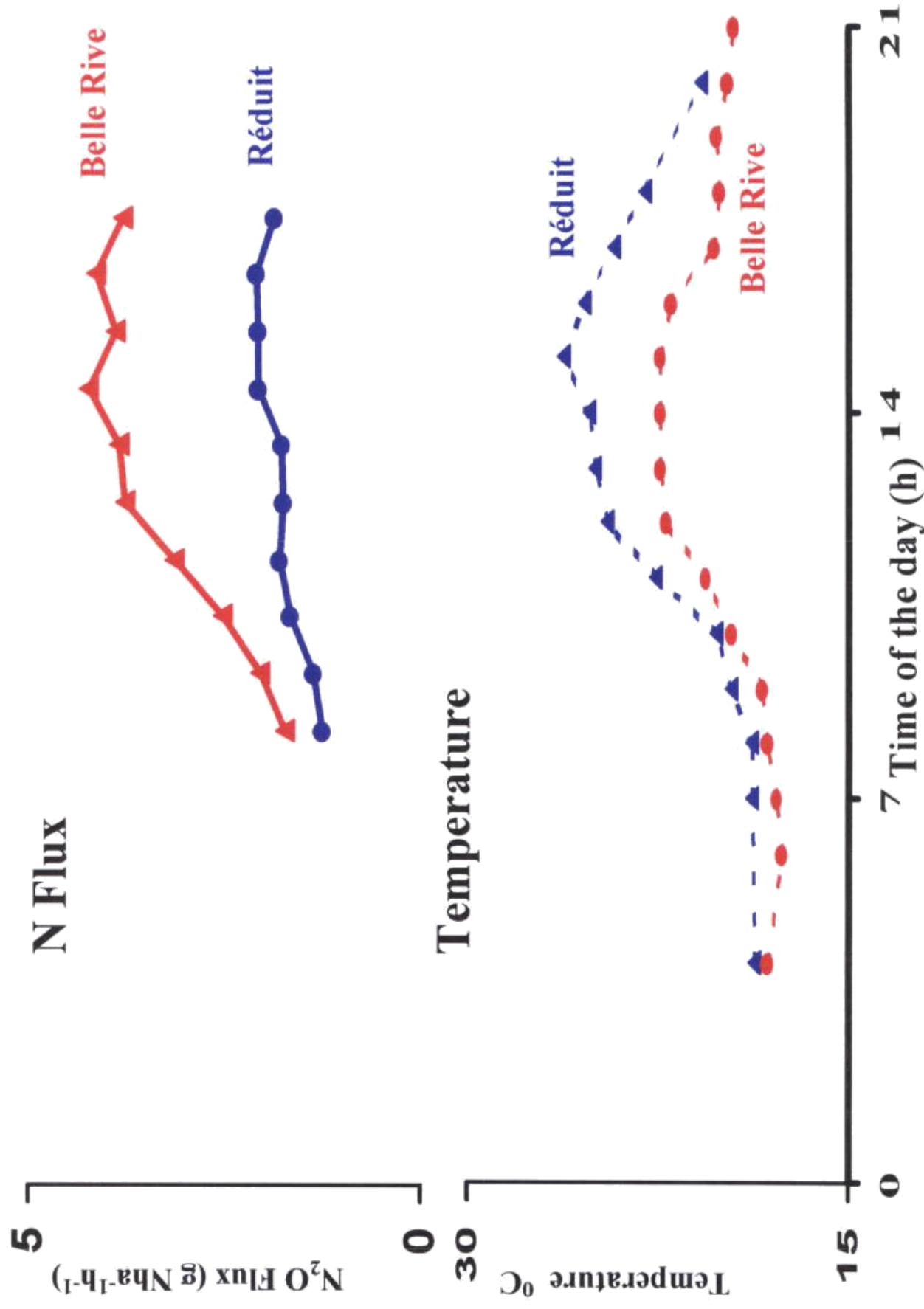


Fig 1 Soil Temperature and variation of gaseous N emission with time of day at Réduit and Belle Rive

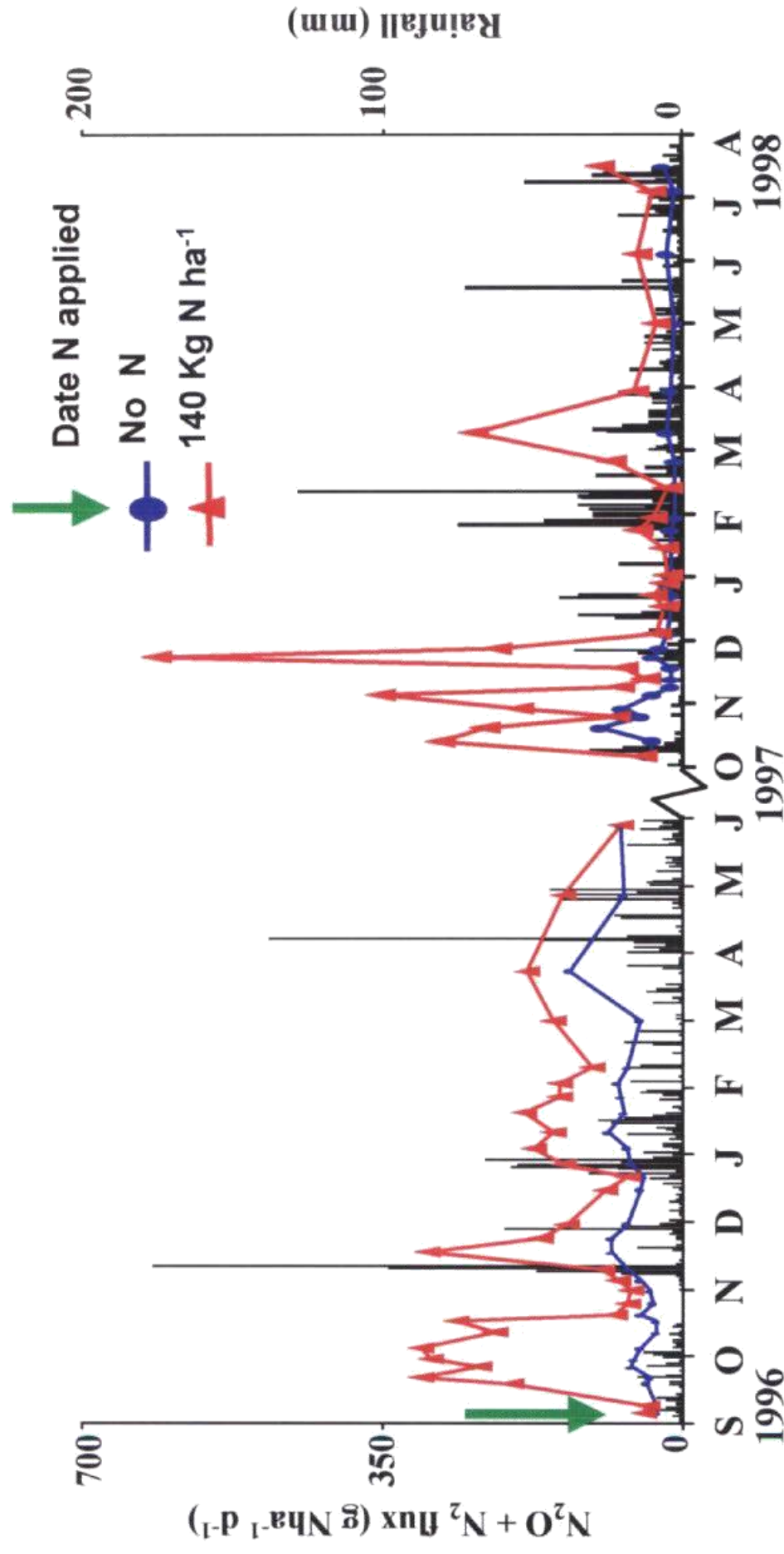


Fig 2a Influence of nitrogen fertilization (140 kg N ha^{-1}) on $\text{N}_2\text{O} + \text{N}_2$ emission from soils under sugarcane at Belle Rive

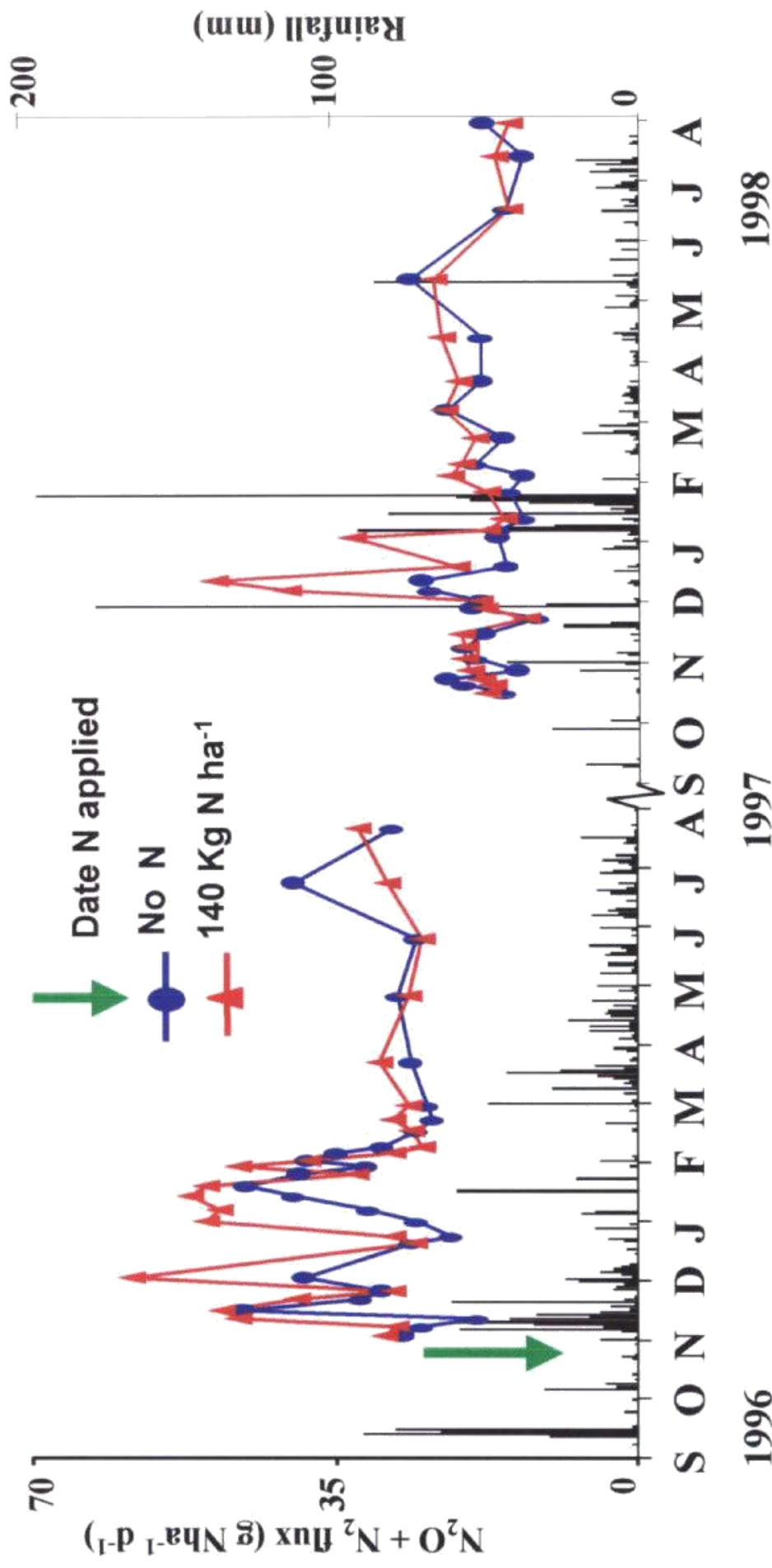


Fig 2b Influence of nitrogen fertilization (140 kgNha⁻¹) on N₂O + N₂ emission from soils under sugarcane at Pamplémousses

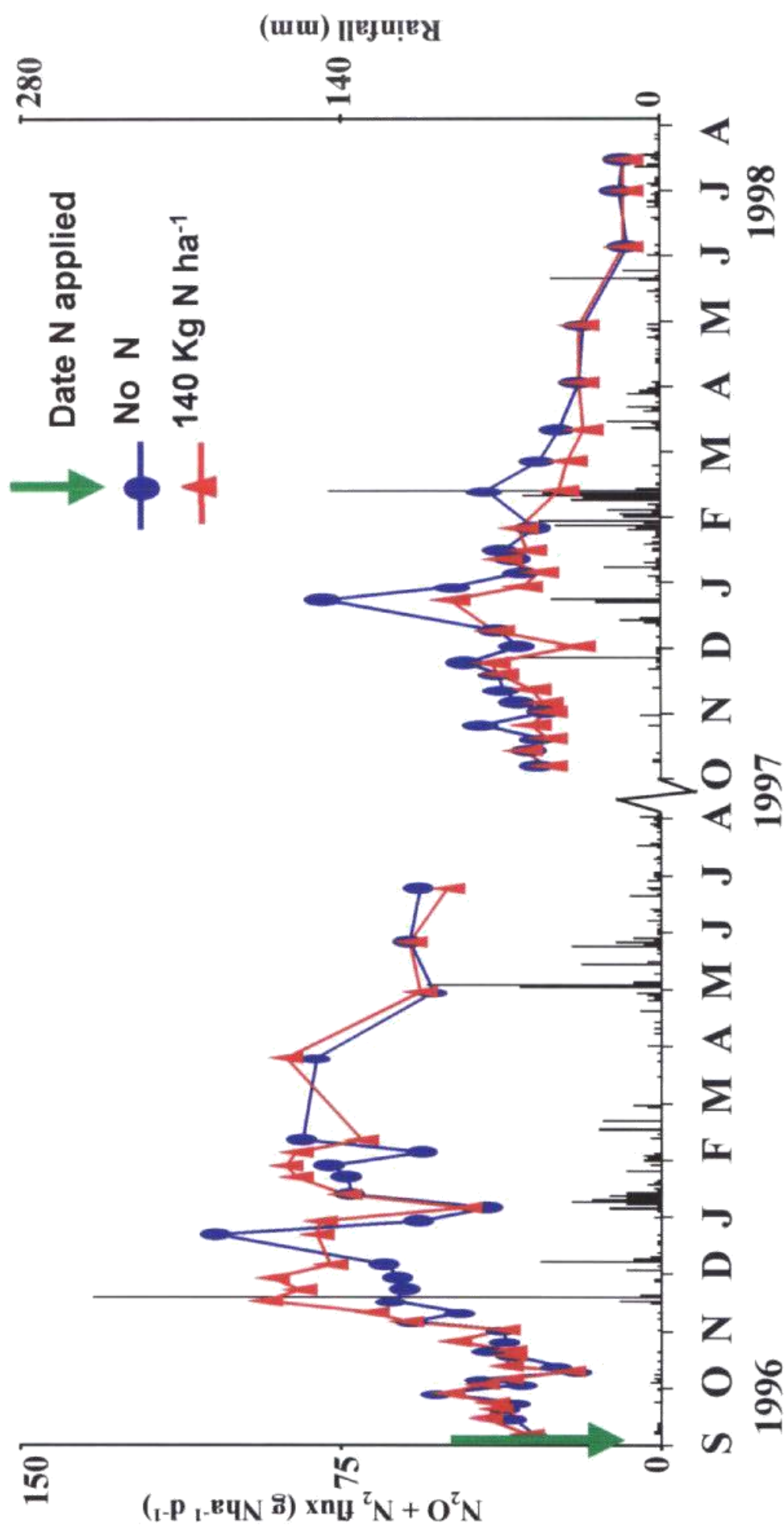


Fig 2c Influence of nitrogen fertilization (140 kgNha⁻¹) on N₂O + N₂ emission from soils under sugarcane at Réduit

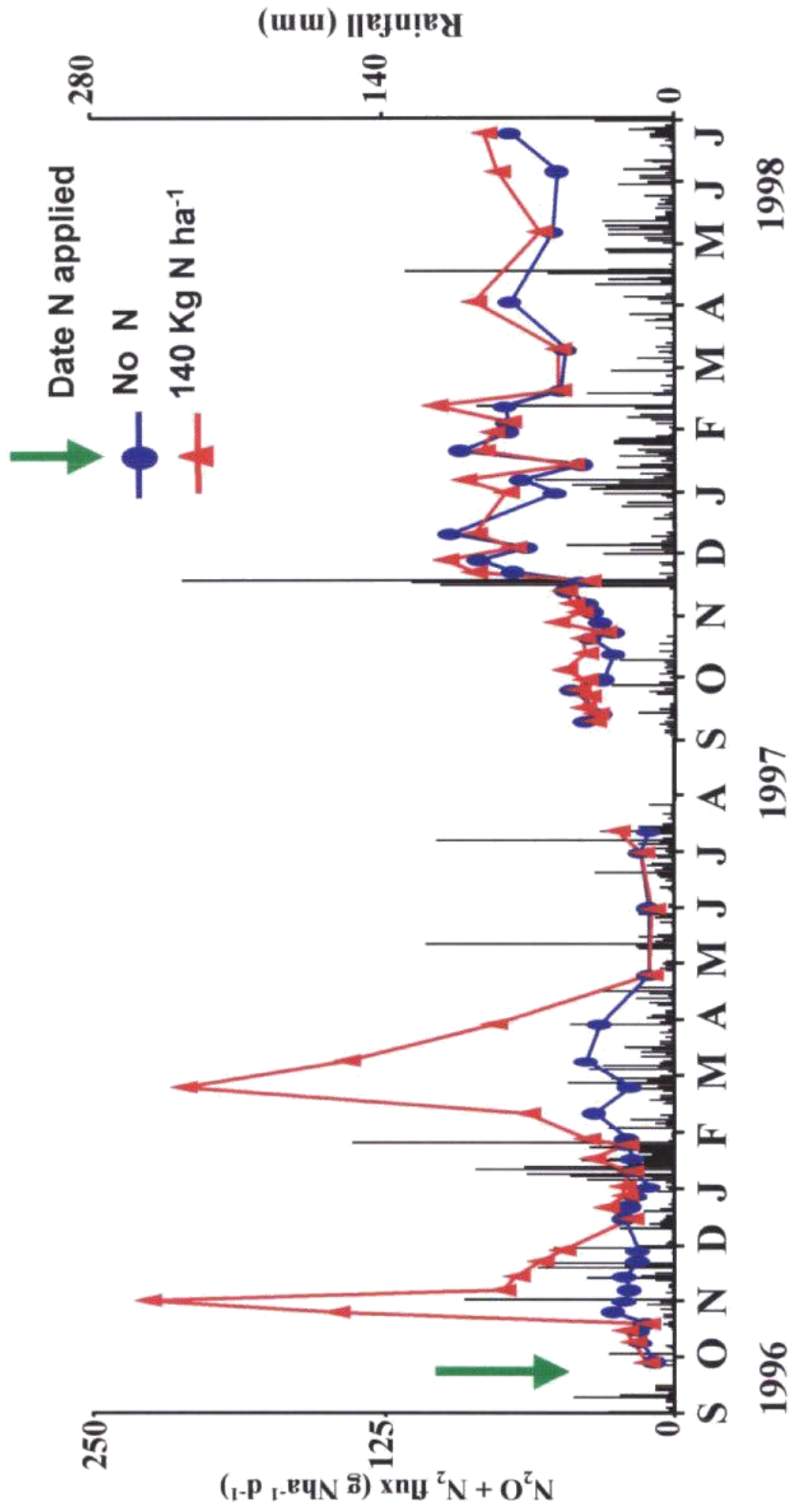


Fig 2d Influence of nitrogen fertilization (140 kgNha⁻¹) on N₂O + N₂ emission from soils under sugarcane at Union Park

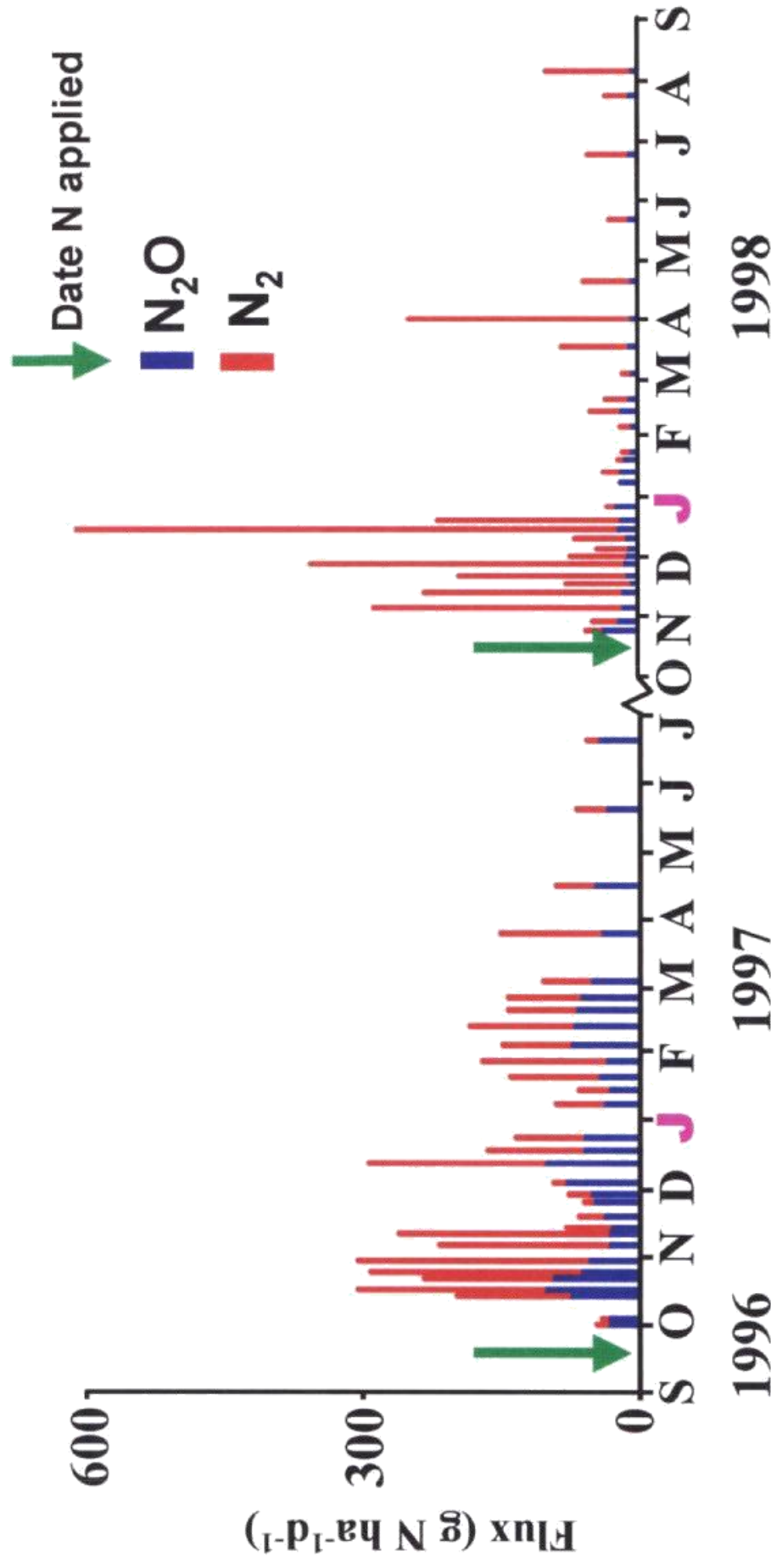


Fig 3a .Composition of N_2O and N_2 in the gases emitted to the atmosphere from soils under sugar cane receiving 140 kgN ha^{-1} at Belle Rive

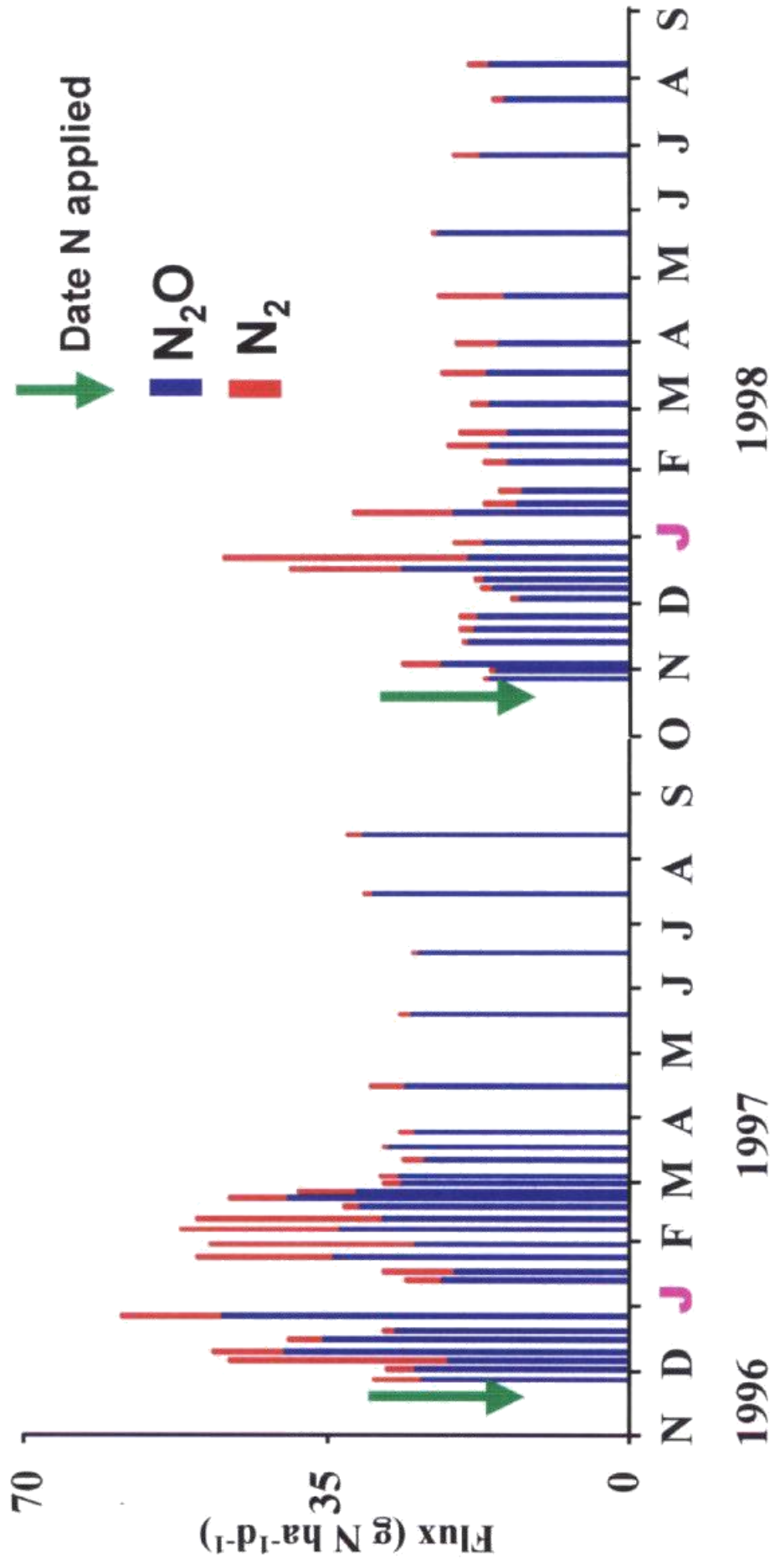


Fig 3b. Composition of N_2O and N_2 in the gases emitted to the atmosphere from soils under sugar cane receiving 140 kgN ha^{-1} at Pamplémousses

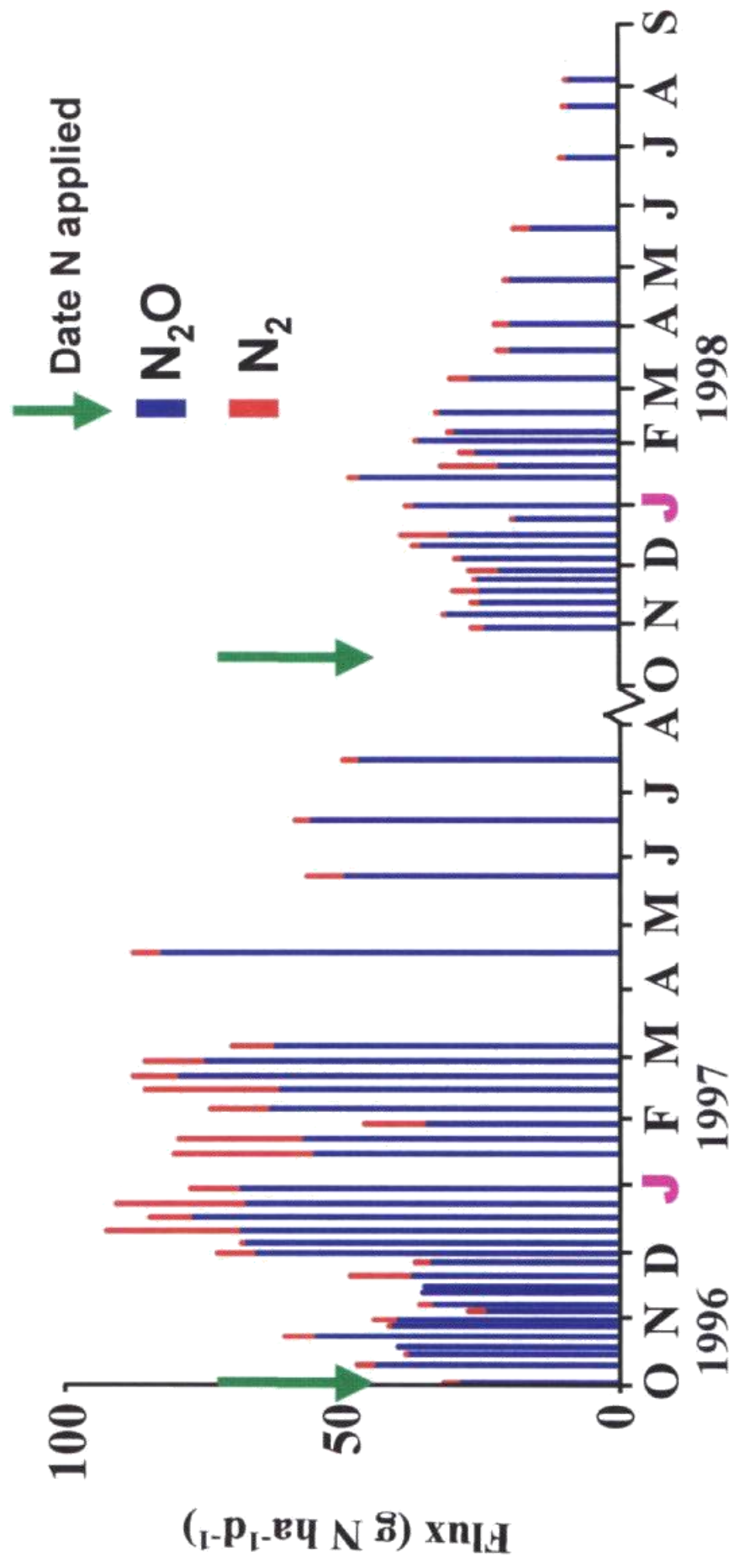


Fig 3c. Composition of N_2O and N_2 in the gases emitted to the atmosphere from soils under sugar cane receiving 140 kgN ha^{-1} at Réduit

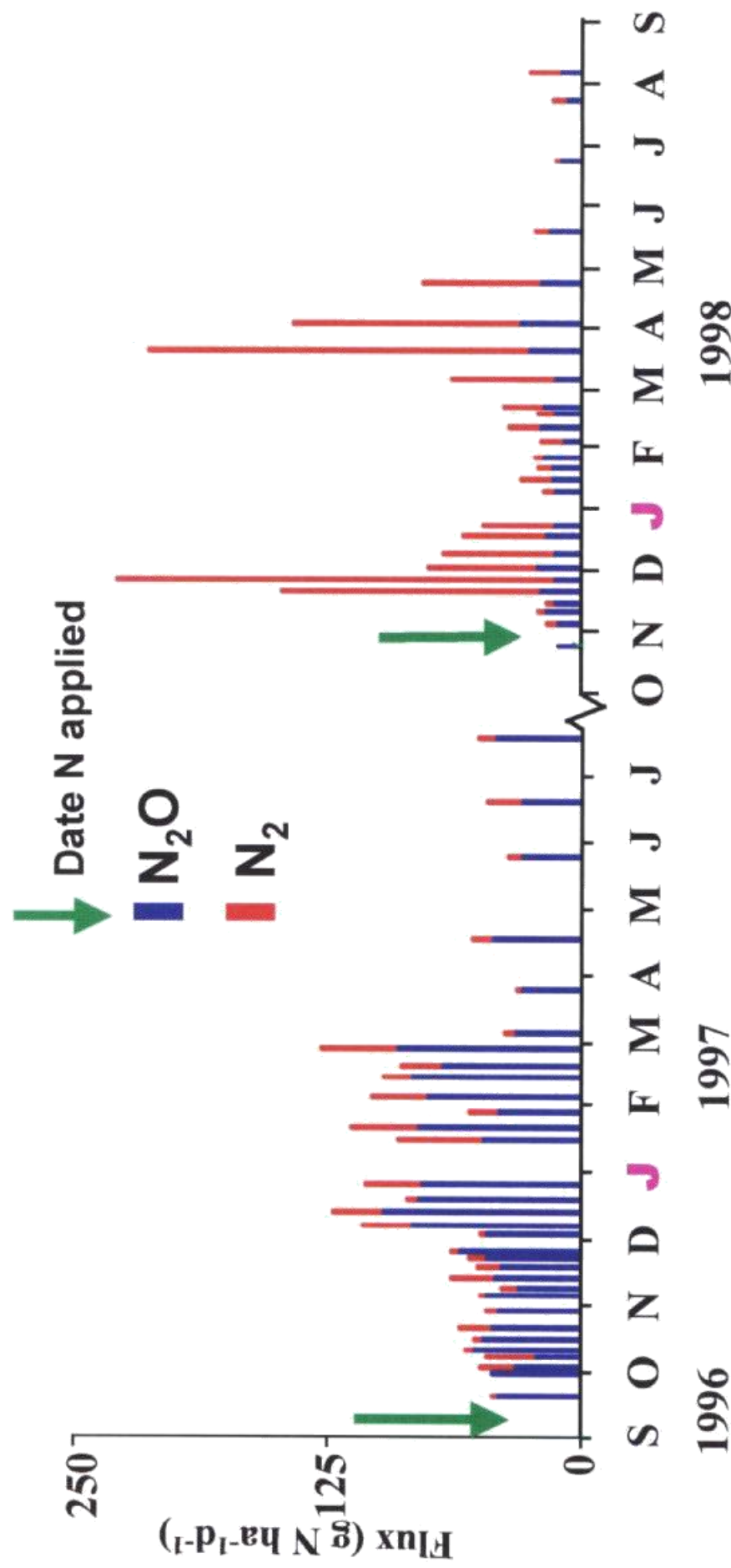


Fig 3d. Composition of N_2O and N_2 in the gases emitted to the atmosphere from soils under sugar cane receiving 140 kgN ha^{-1} at Union Park

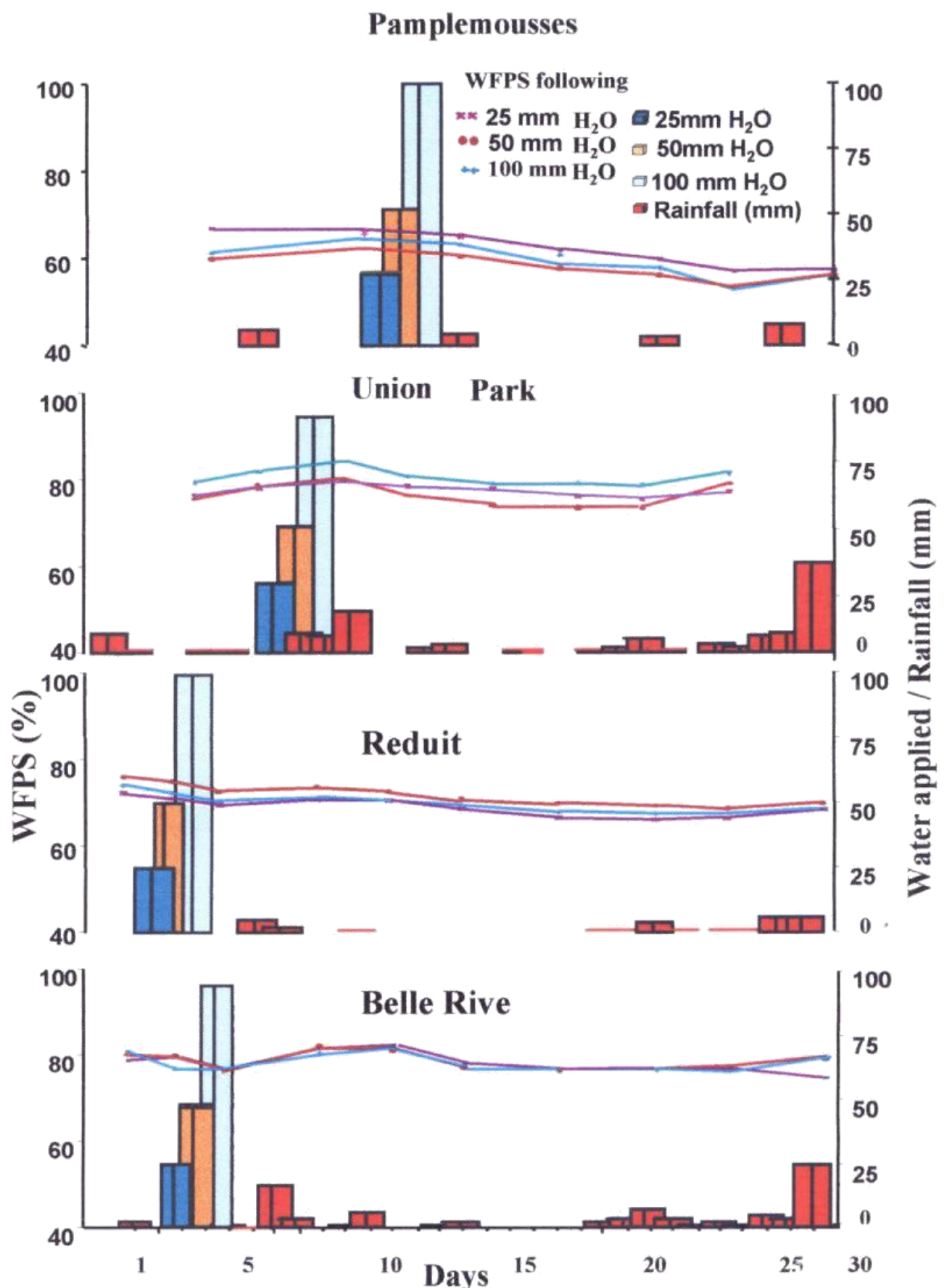


Fig 4 Evolution of water filled pore space (WFPS)* with time after the addition of 25, 50 and 100 mm water to soils at Pamplémousses, Union Park, Réduit and Belle Rive

* determined by Gamma ray attenuation as outlined by Gardner (1986) .