Review

Mitigating N$_2$O emissions from agriculture: A review of the current knowledge on soil system modelling, environmental factors and management practices influencing emissions

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The AgriGES project is a Concerted Research Action lead by Gembloux Agro-Bio Tech, aiming to quantify methane and nitrous oxide emission from pastures and crop fields, respectively. Besides quantification, a second important goal of the AgriGES project is to study the flux dynamics and to gain a better understanding of the biophysical processes coming into play. We focus here on N$_2$O fluxes and first propose an overview of the current modelling efforts. Two main weaknesses of these models have been identified, and potential new developments are suggested to mitigate these issues, with an emphasis on the denitrification process. Secondly, we propose a review of the current knowledge on the main environmental factors influencing nitrous oxide emission. Several mitigation options are also explored.

Key words: Nitrous oxide emission, environmental factors, modeling.

INTRODUCTION

According to the Intergovernmental Panel on Climate Change (IPCC) 5th assessment report (Stocker et al., 2013b), nitrous oxide (N$_2$O) concentration in the atmosphere was 324 ppb in 2011, being 20% higher than in the pre-industrial period. This greenhouse gas (GHG) concentration is still increasing at an approximately constant rate of 0.8 ppb/year since the 1980s. N$_2$O concentrations are far below other GHG concentrations, e.g. CO$_2$ but it has a global warming potential 298 times higher than CO$_2$ over 100 years. N$_2$O is thought to be responsible for around 7.5% of the total radiative forcing induced by GHG (Stocker et al., 2013b). Nitrous oxide emissions caused by human activities represent more than two thirds of the total emissions. N$_2$O emitted from agricultural soil is known to be a major source (about 60%) of these

anthropogenic emissions (Mosier et al., 1998; Syakila and Kroeze, 2011).

The physical and biological processes responsible for nitrous oxide production in soil and its emission to the atmosphere lead to an extreme spatial and temporal variability of fluxes. Concerning the latter, it is commonly assumed that N_2O emission is composed of a background flux mainly due to the nitrification process, while high emission peaks occur due to denitrification in times of anaerobiosis e.g. after a rainfall. On an annual basis, these peaks generally account for about 50% of the total N_2O emissions, while representing only a minor part (about 10%) of the time (Groffman et al., 2009; Molodovskaya et al., 2012). The occurrence and magnitude of these peaks are also dependent on other variables influenced by human activities (mainly N-availability) enhanced by fertilization practices.

In this context, more and more attention is being given to the identification of the main environmental factors driving nitrous oxide emission, as well as to mitigation strategies to be applied in agriculture. Any mitigation options proposed to farmers need to be based on scientific evidence. This includes experimental tests at field scale, but this may also be done via the soil system modelling, provided that the model has been validated on experimental data.

Several models have been developed to predict N_2O emissions from soils. This paper first aims to identify why so many models have been proposed. Secondly, special attention will be paid to denitrification (NO_3-reduction to N_2) as it is the main biogeochemical process responsible for nitrous oxide emission from soils. Finally, a review of the current knowledge on environmental factors (soil pH, moisture, temperature, nitrogen (N) and carbon (C) content) and, to a lesser extent, a review of the explored mitigation strategies (e.g. reduced tillage, cover crop, lower fertilizer input) is also proposed.

Two main bottlenecks have been identified in current modelling: competition between the different steps of denitrification is never taken into account; and soil physics - particularly gaseous transports in soils - is poorly represented in most of the models. Some models assume direct emission of the produced gases to the atmosphere, thus bypassing all other reactions that may occur during the gas transport. Implementing new developments would be a great contribution to improving existing models.

PROPOSED MODELS

Despite the existence of well-known concurrent models (Blagodatsky and Smith, 2012) such as (this list is non-exhaustive) DNDC (Li et al., 1992a, b), ECOSYS (Grant et al., 1993a, b), STICS (Brisson et al., 2003, 1998,

2002), and DAYCENT (Parton et al., 1998, 2001), many different groups continue to develop new ones (e.g. MiCNiT (Blagodatsky et al., 2011), TOUGHREACT-N (Maggi et al., 2008)) or try to refine existing approaches, such as NLOSS which is based on DNDC (Riley and Matson, 2000).

There are several reasons for this:

1. From a historical point of view, some models have been designed for specific goals (e.g. DAYCENT for CO_2 emissions, STICS for crop yield), and later applied to other subsidiary outputs such as N_2O emissions.
2. A lack of cooperation between research groups attached to different research fields such as soil biology, soil physics, or agronomy (Blagodatsky and Smith, 2012; Sutton et al., 2011). The multidisciplinary nature of process-based approaches makes communication between researchers difficult. They may use different terminologies to refer to a same process or event, e.g. "dissimilatory nitrate reduction to ammonia" and "nitrate ammonification".
3. A lack of modular implementation, which makes it difficult to insert new developments into old models. Rubol et al. (2013), for instance, have implemented a new model to take nitrate ammonification into account because of growing experimental evidence of the importance of this process.
4. The need for models applicable to a wide range of temporal and spatial scales (Manzoni and Porporato, 2009). The scale of models ranges from days to decades and from experimental plots to the whole planet. At large regional or national scales, simple experimental models with few parameters are preferable to more complex mechanistic ones, because the number of parameters tends to increase with the model complexity. Lots of these parameters in mechanistic models are site and climate specific (Saggar et al., 2013), and need to be initialized from experimental measurements.

The up-scaling of process-based models is still problematic, but they nevertheless have a great advantage over simpler ones in that their modelling scheme is closer to reality, thus giving the opportunity for a deeper understanding of the biophysical processes influencing N2O emissions, and a better simulation of mitigation options at the farm scale.

Heinen (2006) makes a distinction between two different types of process-based models: microbial growth models and soil structural models. The former models focus on the dynamics of the microbial organisms for the N-cycling processes and often assume immediate transfer of the produced gases from the soil to the atmosphere. The latter consider gaseous transport in soils in more detail, simulating the anoxic fraction of the pore volumes in which denitrification occurs (Blagodatsky and Smith, 2012). Meanwhile,

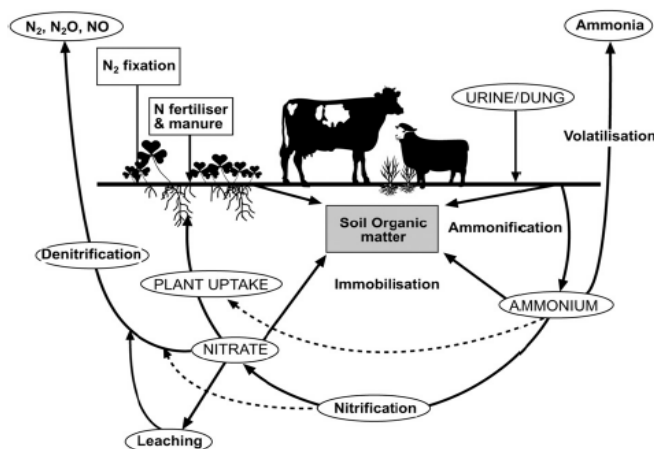## NITROGEN DYNAMICS IN CROPS AND PASTURE
(adapted from Saggar et al, 2013)

Figure 1. Basic N-cycle.

REACTION STEPS OF DENITRICATION
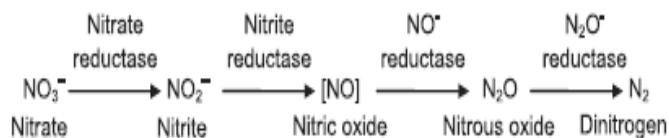(adapted from Saggar et al, 2013)

Figure 2. Reaction steps of denitrification.

most models consider simple dependencies of environmental variables (temperature, moisture) on the production of gases, and completely neglect the microbial nature of the C and N cycles.

Chen et al. (2008) provided a very comprehensive comparison of the most widely used models. The vast majority of these models are to be classified as microbial growth models (e.g. DNDC, DAYCENT, STICS). Most of the soil structural models have been tested in laboratory experiments, such as that of Arah and Vinten (1995), while a few attempts to up-scale them are worth noting (Langeveld and Leffelaar, 2002). There is probably a huge knowledge gap to fill, that is coupling a microbial growth model and a soil structural model into one. Since both mechanisms occur over the same timescale, their interaction cannot be neglected (Blagodatsky and Smith, 2012). This would be difficult to insert into the implementation of older models,

justifying some new attempts such as MiCNit (Blagodatsky et al., 2011) and PASTIS-CANTIS (Cannavo et al., 2006) to combine these two aspects.

DENITRIFICATION

Denitrification is a natural microbial process and is an essential part of the nitrogen cycle, briefly illustrated in Figure 1. Denitrification is the stepwise reduction of nitrate (NO_3^-) to nitrogen (N_2) via nitrite (NO_2^-), nitric oxide (NO) and nitrous oxide (N_2O) (Figure 2). Denitrification is performed by facultative anaerobic bacteria. In conditions of oxygen (O_2) depletion, these bacteria use nitrate as a substitute electron acceptor for adenosine triphosphate (ATP) generation, which is the energy source for cell processes. Each of the four steps of denitrification is catalyzed by a specific enzyme (nitrate reductase: Nar; nitrite reductase: Nir; nitric oxide reductase: Nor; nitrous oxide reductase: Nos). Since NO is highly cytotoxic, all denitrifiers have the gene to code Nor. On the other hand, some denitrifiers lack the gene to code Nos, therefore N_2O is their final product of denitrification. The complete denitrification cycle is mainly a symbiotic process, with bacteria coding Nos performing the last step.

There is growing evidence for competition among the different steps of denitrification. The experiment from Pan et al. (2013a) strongly suggests that this competition plays on the available electron donors (labile carbon). Whereas it is commonly thought that this competition only occurs under carbon-limiting conditions, their results tend to prove that it also occurs with high available carbon content.

According to them, the limiting step that triggers the competition is that the Nir, Nor and Nos use the same pathway to receive electrons, that is they all use the cytochrome c550 coenzyme, while Nar uses the ubiquinone/ubiquinol pool. This may explain their observations, showing a greater competition between Nir and Nos than between Nar and Nos or Nar and Nir. Under carbon-limiting conditions, Nir is highly favoured over Nos, leading to high nitrous oxide emissions.

Other experimental studies have indicated that an increasing nitrate concentration tends to enhance N_2O emission, provided that there is enough available labile carbon (Senbayram et al., 2012). Other molecules are also known to regulate enzyme activity and production, including intermediates from denitrification. Indeed, an increase in NO concentration favours NO-reductase production (Thomson et al., 2012; Zumft, 1997). This makes great sense in terms of biological evolution, since bacteria produce this enzyme to reduce a lethal component (NO) in response to its increase in concentration. It is also worth noting that N_2O reduction to N_2 is energetically less favourable compared to the other steps in denitrification (Senbayram et al., 2012),

which may also explain why bacteria tend to favour the previous steps in denitrification, resulting in greater N_2O emission and comparatively less N_2 produced.

Denitrification can also be performed by some fungi. The mechanisms and the relative importance of fungal denitrification have not yet been fully addressed, even though a few authors have found experimental evidence that fungi comprise the majority of denitrifying organisms in grassland soils (Laughlin and Stevens, 2002). The reaction steps are quite similar to bacterial denitrification, but the enzymes involved are different and are not inhibited by O_2. Another important difference is that none of these fungi have the gene to code N_2O-reductase.

An overview by Heinen (2006) assesses more than fifty simplified denitrification models. The majority of these models are to be used at a regional scale. Denitrification is considered as a first order decay process, which is inadequate to explain observed non-linearities in soil N-dynamics (e.g. non-linear response of N_2O emission on quantity of N-fertilizer input (Kim et al., 2013).

Following the pioneering work of Leffelaar and Wessel (1988), most process-based models (e.g. DNDC and TOUGHREACT-N) consider denitrification as a 4-step chain reaction, each step being independent of the others, except that the product of a previous reaction is the reactant of the next one. It is assumed that each reaction follows Michaelis-Menten kinetics. In this regard, STICS constitutes an exception by using a simplified sub-model of denitrification, namely NOE (Bessou et al., 2010; Henault et al., 2005). Very few attempts have been made so far to model the competition between the different reaction steps. As another example of the multidisciplinary nature of this topic, efforts have been made by researchers in waste water treatment, where denitrification is a crucial process.

Based on their previous experimental results (2013a), Pan et al. (2013b) have proposed a new model in which denitrification steps are thought to be mediated by electron carriers going through cell membranes, and competition among the different steps is modelled as a competition for these electron carriers. Each reaction rate is multiplied by a Michaelis-Menten term regarding the carrier concentration, with different affinity constants for each reaction. This seems to be a promising approach to account for the inhibition of nitrous oxide reduction by nitrite and nitrate.

A MODEL OF SOIL STRUCTURE

Directly representing the soil pore space structure in current models might be a tricky task, because it would involve a review of all simulated processes, changing their implementation to make them fit in to the geometric structure. Moreover, including a 3-dimensional (3D) representation of the soil structure in a model is likely to incur time and computer memory consumption issues (Blagodatsky and Smith, 2012).

Another approach involves using soil structural models as a pre-process tool to provide pedotransfer functions to be later used in a more generic model. In this regard, the pore-solid-fractal approach (PSF) derived by Perrier et al. (1999) has proved to be a powerful tool that is able to account for a wide variety of soil physical properties (Ghanbarian-Alavijeh et al., 2011; Perfect and Kay, 1995), such as the theoretical water retention curve derived by Wang and Zhang (2011).

Rappoldt and Crawford (1999) used the 3D-soil space resulting from the PSF model in order to solve the oxygen diffusion-respiration equation in soils. In relation to denitrification, their results can be used to simulate the soil anoxic fraction as a function of its water content in a given depth layer. This, together with a model of denitrification rate as a function of the soil depth (most likely decreasing with depth because of the decrease of the microbial community), may potentially lead to a better simulation of the N_2O emission peak dynamics, both in magnitude and temporal occurrence.

MAIN DRIVERS

Several environmental factors are known to play a key role in greenhouse gas emission from soils. In general, complete denitrification is favoured by high water content, neutral to slightly basic pH, high temperature, low O_2 diffusion, and labile C availability. While each individual influence is quite well researched, there is still a lack of complete comprehension of some mechanisms by which these factors (e.g. pH) act on emissions, and a fully comprehensive scheme of their interactions is thought to be unrealistic (Butterbach-Bahl et al., 2013). Several authors have provided reviews of the current knowledge on these driving factors. Of particular interest are: Butterbach-Bahl et al. (2013), Giles et al. (2012), and Saggar et al. (2013).

The timescales at which these factors influence N_2O emission from soils vary in a very wide range, from hours to decades. For instance, water content is known to regulate the anoxic volume of soils in a very direct way, while soil pH may have a long-term effect via microbial community adaptation. Long-term feedback effects are also reported, e.g. the increase of soil temperature due to the global warming will probably enhance greenhouse gas emission from soils in the coming decades.

Soil pH

Soil pH is known to be a key variable in soil biogeochemical processes, although its influence is not

yet well understood (Liu et al., 2010; Simek and Cooper, 2002; Van den Heuvel et al., 2011). Regarding nitrous oxide emissions, it is globally accepted that acidic conditions (lower pH) tend to increase $N_2O:N_2$ emission ratio while decreasing total N_2O and N_2 emission (Liu et al., 2010). To minimize N_2O emission, a neutral to slightly alkaline soil pH seems to be optimal (Giles et al., 2012), dependent on other soil characteristics.

Soil pH affects denitrification in many different ways. Both direct and indirect effects of pH on denitrification rates, denitrification end product, and denitrifier community have been reported. Liu et al. (2010) have shown that a pH-dependent effect on denitrification enzyme activity occurs at a post-transcriptional level. They suggested that pH may disable the protein assembly, or influence its shape, leading to unusable active site. Bakken et al. (2012) working with a specific bacteria (*Paracoccus denitrificans*), observed that at pH 7, nearly no N_2O was emitted from batch cultures, while at pH 6, N_2O-reductase activity was drastically reduced, leading to high N_2O emissions.

Indirect effects of soil pH on denitrification may include changes in organic carbon availability and nitrogen mineralization rates (Simek and Cooper, 2002). These two variables tend to decrease under acidic conditions, leading to a smaller microbial community, which in turn leads to lower denitrification rates in soils (Van den Heuvel et al., 2011). Meanwhile, this effect could be counterbalanced by a long term adaptation of the microbial community.

Soil moisture and oxygen availability

Soil water content may influence gaseous emissions from soils in many different ways. For instance, water presence in soil is necessary for plant and microbial growth, which in turn can influence biochemical reaction rates and enhance nitrate uptake by crops. However, soil water content is particularly studied for its key role in the development of anaerobic conditions in soils. Indeed, most of the models (e.g. DNDC) use the water filled pore space (WFPS) of soils as a proxy to define periods of activation and inhibition of the denitrification process.

Several experimental studies, e.g. Bateman and Baggs (2005), have shown the existence of a WFPS threshold value above which denitrification rates increase sharply with soil water content, and under which denitrification rates are low and seem unrelated to WFPS. DNDC uses a threshold value of 60%, and this value differs according to soil type. After reaching a maximum around 70 to 80% WFPS, nitrous oxide emission generally tends to decrease. This is thought to be caused by a lower gas diffusion into soils, thus giving more time for denitrifiers to reduce N_2O (Smith et al., 2003).

De Klein and Van Logtestijn (1996) suggest that the WFPS threshold value is equivalent to or slightly higher than field capacity. This makes sense in several ways: at field capacity, micro-pores are still filled with water; whereas these pores are also thought to be the privileged location of microorganisms (including denitrifiers) in soils (Or et al., 2007). In addition, Saggar et al. (2013) report a duration of 24 to 48 h after a rainfall for N_2O emissions to return to their background level. The experimental definition of the field capacity is indeed the soil water content 48 h after its saturation.

This threshold value may also be related to several models describing the anoxic fraction of soils as a function of its water content (Arah and Vinten, 1995; Rappoldt and Crawford, 1999; Schurgers et al., 2006). These models show a highly non-linear response of soil anoxic fraction to soil WFPS, with the anoxic fraction notably increasing above a certain threshold value which may be compared to the observed threshold value for the denitrification activity. The threshold values computed by Rappoldt and Crawford (1999) and Schurgers et al. (2006) are in accordance with the WFPS threshold range given by Bateman and Baggs (2005).

Soil temperature

Soil temperature most significantly influences gas emission via its role in microbial growth and activity (Braker et al., 2010). Denitrification occurs across a wide temperature range (near 0 to 75°C), and is limited by water availability (freezing below 0°C) and microbial death at too high a temperature (Saggar et al., 2013). Optimal temperatures for denitrification rates have been reported from 25 to 30°C (Braker et al., 2010).

Of particular interest are fluctuations of temperature around 00, which lead to freeze-thaw cycles with high emission peaks reported in numerous studies, e.g. (Mørkved et al., 2006). Several explanations have been proposed for these peaks (de Bruijn et al., 2009). N_2O may accumulate in frozen phases due to lower gas diffusion and be released on thawing. It has also been suggested that substrate availability may be more important in winter because of low plant uptake during this period.

Increasing temperature may also cause higher oxygen consumption, thus leading to an increase in the anoxic fraction of the soil pores as less oxygen is available (Smith et al., 2003).

At a seasonal scale, Wolf and Brumme (2002) reported a linear relation between temperature and nitrous oxide emission on a bare soil with WFPS kept constant at field capacity. On a larger time scale, global warming (and consequent soil warming) is likely to have an important positive feedback on CO_2, CH_4 and N_2O

emission from soils in the coming decades (Arneth et al., 2010; Stocker et al., 2013a; van Groenigen et al., 2011), mainly resulting from an enhanced microbial activity.

Soil carbon and nitrogen content

As electron donors and electron acceptors for denitrification respectively, both labile organic carbon and nitrate play a key role in nitrous oxide emissions. Besides the direct effect of providing supplies for the denitrifiers, other indirect effects have also been reported (Giles et al., 2012). For instance, the presence of carbon stimulates heterotrophic respiration, hence favouring anaerobic conditions in soils.

Several studies focus on the C:N ratio of fertilizers as well as on the applied rate to reach maximum crop yield while limiting N_2O emission. In their meta-analysis, Van Groenigen et al. (2010) reported a sharp increase in nitrous oxide emission when the N-fertilizer input leads to nitrogen saturation (excess of N compared to plant and microbial maximum demand) of soils, which may indicate that a precise control of N-input is a straightforward and effective option to reduce nitrous oxide emission.

The incubation experiment led by Senbayram et al. (2012) found that, for soils with low nitrate content, the nitrous oxide emission can be substantially lowered by the addition of organic matter with high content of labile C, by promoting the reduction of N_2O to N_2. Alternatively, for soils with high nitrate content, labile organic C enhanced nitrous oxide emission, most likely due to an inhibitory effect of NO_3- on the N_2O-reductase.

Explored mitigation options

Strategies to reduce greenhouse gas emissions from agriculture mainly rely on our knowledge of the main drivers, with the aim to influence these factors (e.g. pH and liming, soil carbon and nitrogen content and fertilization practices) in order to decrease the emission levels as much as possible. These strategies have to be placed in a wider context, considering other constraints that best management practice assessments should also acknowledge:

1. Productivity must be kept at a sustainable level in order to ensure food security.

2. Avoidance of nitrate leaching to maintain water quality. Moreover, this nitrate will eventually end in waste water treatment plants where denitrifiers are used to remove it from water, also leading to nitrous oxide emissions (Cui et al., 2014; Pan et al., 2013b).

3. The greenhouse gas budget of an agricultural farm should also take into account GHG emission from agricultural machinery, GHG emission for the production and transport of inputs (pesticides, fertilizers, lime, etc.). Promoting less intensive crop systems and reducing inputs to the field results in the saving of a considerable amount of GHG emissions upstream of the cropping itself (Smith et al., 2008).

4. The need to compute greenhouse gas emissions for a complete sequence of crops, which is more relevant than for a single crop as a previous crop affects the following one because of its influence on soil nutrients (Lehuger et al., 2011).

Several mitigation options for direct nitrous oxide emissions from the field are discussed hereafter. It is most likely that the best way to reduce these emissions involve a mix of these different practices, as they influence each other.

Fertilizers

As shown experimentally by Van Groenigen et al. (2010), nitrous oxide emission increases exponentially with excess N-fertilizer compared to plant demand. According to their study, lowering N-input to match the crop demand would be a straightforward and effective way to reduce N_2O emission per crop yield. Once this is done, if soil nitrate content remains at a relatively low level, addition of labile organic carbon may also be a way to enhance reduction of N_2O to N_2, thus also leading to a reduction of nitrous oxide emission. In this regard, combining both organic and synthetic fertilizers would be an efficient mitigation option (Senbayram et al., 2012).

Changing spatial and temporal location of N-input may also be an important factor in improving their efficiency. For instance, in the UK, avoiding manure application during autumn and early spring may be an effective way to reduce yearly emissions, since it often rains a lot during these periods leading to high denitrification rates in soils. This mitigation scenario has been tested with DNDC and shows promising results.

Technical progress has (Cardenas et al., 2013) also been made in the manufacture of slow- (or "controlled-") release fertilizers, that is, fertilizers which decompose more slowly than traditional ones, thus providing nutrients more homogeneously over time (Azeem et al., 2014). The use of slow-release fertilizers seems to be a promising way to increase N-use efficiency, but their cost remains prohibitive.

Enzymatic regulations

While all of the enzymes involved in the denitrification process are influenced by environmental factors,

explored mitigation options focus mainly on the N_2O-reductase. Enhancing N_2O-reductase activity is expected to decrease the $N_2O:N_2$ ratio. Liming of acidic soil is a way to achieve this, but it also leads to higher total N_2O+N_2 emission (Bakken et al., 2012).

Meanwhile, this additional direct loss of N-input is thought to be largely compensated by the decrease of nitrate leaching, since nitrate is consumed by denitrifiers. Another key variable for the N_2O-reductase enzyme is the soil copper content. Indeed, copper (Cu) is a mandatory and irreplaceable cofactor in order for this enzyme to be active. Thus, addition of Cu in copper-poor soil may also decrease $N_2O:N_2$ ratio, without disturbing the total N_2O+N_2 emission (Richardson et al., 2009; Thomson et al., 2012)

Regarding other biogeochemical processes, the use of nitrification (oxidation of ammonia to nitrate) inhibitors together with ammonium-based fertilizers has also been addressed (Liu et al., 2013). Inhibiting nitrification is expected to reduce nitrate losses by leaching and denitrification, while still leaving a usable form of N (rhat is, ammonium) for plant uptake. Liu et al. (2013) reported an increase of crop yield, and a better nitrogen use efficiency, as well as a reduction of N_2O emissions. Meanwhile, a phytotoxic effect associated with excessive inhibitor application rates was also reported, as well as a subsequent yield reduction.

Moreover, several key points have not been addressed yet concerning the use of nitrification inhibitors, e.g. the GHG emission resulting from their manufacture and their long-term impact on the soil microbial community.

Cover crops

Cover crops, also known as catch crops, are mainly used during winter to avoid soil erosion by the structural effect of roots preventing soil loss, and nitrate leaching by plant uptake. N sequestered by cover crops is then made available for the main crops during the growth season, which enhances the yearly N-use efficiency of the crop rotation. Given this dual effect, cover crops are often considered as a win-win practice for N-management (Constantin et al., 2010), and their use is thought to reduce reliance on fertilizer input. In drier lands the positive impact of cover crops is less pronounced, since there is naturally less nitrate leaching in winter in these areas.

There are also concerns about water and nitrogen stress induced by cover crops, resulting in yield reduction issues for the main crop (Celette and Gary, 2013; Celette et al., 2008).

Conservation tillage

It is widely thought that conventional tillage is the primary cause of the decrease in soil organic carbon content. Many soils have lost up to 40% of the C they contained before cultivation (Baker et al., 2007). The first intent of reduced or no-till practices is to promote soil C sequestration, thus allowing soils to refill in order to act as a reservoir. In addition to this potential for sequestration, an increased soil C content can also promote the reduction of N_2O to N_2, provided that soil nitrate remains at a relatively low level. In order to control nitrate content, conservation tillage is often used in combination with other management practices, such as lower N-fertilizer application rates and the use of winter cover crops (Constantin et al., 2010; Petersen et al., 2011). A higher soil organic carbon content is also thought to enhance plant uptake, but this effect may be counterbalanced by a greater soil compaction, leading to a difficult root growth.

Nevertheless, a few authors have pointed out that evidence of soil C sequestration induced by reduced tillage is not that obvious. Baker et al. (2007) argued that the sampling depth in most of the experimental studies is not deep enough. Conservation tillage may change the C distribution in soils, thus increasing the C content in the first 30 cm layer, but at deeper depths some studies show a decrease in soil C content.

CONCLUSIONS AND PERSPECTIVES

In this paper, the current knowledge on the main driving factors influencing nitrous oxide emissions from cultivated fields has been reviewed. While many environmental factors have been studied (such as soil pH, water content, temperature), there is still progress to be made to gain a complete comprehension of the influence of soil physical properties on GHG production and emission.

In this regard, a fractal model of soils seems a promising approach to give a consistent theoretical basis that is able to account for a wide variety of soil physical properties (Ghanbarian-Alavijeh et al., 2011; Perfect and Kay, 1995). In particular, the pore-solid-fractal approach (Perrier et al., 1999) allows derivation of a theoretical water retention curve (Wang and Zhang, 2011) and provides a 3D-space in which to simulate several key processes, including the temporal dynamics of the anoxic fraction of the pore volume (Rappoldt and Crawford, 1999).

Another key process which is not taken into account in current modelling efforts is the competition among the different steps of denitrification. The approach from Pan et al. (2013b) seems promising, but needs to be tested in a wider range of anaerobic conditions.

These potential developments need to be incorporated in a more general process-based model of soil C and N cycles which will need to be chosen from the different existing models, based on their current performance in agricultural and meteorological conditions encountered in our region.

Abbreviations: ATP, adenine triphosphate; C, carbon; CANTIS, carbon and nitrogen transport in soil; CH4, methane; CO2, carbon dioxide; Cu, copper; DNDC, denitrification-decomposition; GHG, greenhouse gas; IPCC, intergovernmental panel on climate change; MiC-NiT, microbial carbon and nitrogen turnover; N, nitrogen; N2, dinitrogen; Nar, nitrate reductase; Nir, nitrite reductase; N2O, nitrous oxide; NO2-, nitric oxide; NO3-, nitrate; NOE, nitrous oxide emissions; Nor, nitric oxide reductase; Nos, nitrous oxide reductase; PASTIS, predicting agricultural solute transport in soils; PSF, pore-solid-fractal; STICS, simulateur multidisciplinaire pour les cultures standard ; UK, United Kingdom; WFPS, water filled pore space.

Conflict of Interest

The authors have not declared any conflict of interest.

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